National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Natural Resource Condition Assessment

Glacier Bay National Park and Preserve

Natural Resource Report NPS/GLBA/NRR—2017/1473

ON THE COVER
Photograph of Steller sea lions and birds on islet adjacent to South Marble Island.
Photograph courtesy of the NPS.

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Executive Summary

The Natural Resource Condition Assessment (NRCA) Program aims to provide documentation about the current conditions of important park natural resources through a spatially explicit, multi-disciplinary synthesis of existing scientific data and knowledge. Findings from the NRCA will help Glacier Bay National Park and Preserve (GLBA) managers to develop near-term management priorities, engage in watershed or landscape scale partnership and education efforts, conduct park planning, and report program performance (e.g., Department of the Interior's Strategic Plan "land health" goals, Government Performance and Results Act).

The objectives of this assessment are to evaluate and report on current conditions of key park resources, to evaluate critical data and knowledge gaps, and to highlight selected existing stressors and emerging threats to resources or processes. For the purpose of this NRCA, staff from the National Park Service (NPS) and Saint Mary's University of Minnesota – Geospatial Services (SMUMN GSS) identified key resources, referred to as "components" in the project. The selected components include natural resources and processes that are currently of the greatest concern to park management at GLBA. The final project framework contains 24 resource components, each featuring discussions of measures, stressors, and reference conditions.

This study involved reviewing existing literature and, where appropriate, analyzing data for each natural resource component in the framework to provide summaries of current condition and trends in selected resources. When possible, existing data for the established measures of each component were analyzed and compared to designated reference conditions. A weighted scoring system was applied to calculate the current condition of each component. Weighted Condition Scores, ranging from zero to one, were divided into three categories of condition: good condition, moderate concern, and significant concern. These scores help to determine the current overall condition of each resource. The discussions for each component, found in Chapter 4 of this report, represent a comprehensive summary of current available data and information for these resources, including unpublished park information and perspectives of park resource managers, and present a current condition designation when appropriate. Each component assessment was reviewed by GLBA resource managers, NPS Southeast Alaska Network staff, or outside experts.

Existing literature, short- and long-term datasets, and input from NPS and other outside agency scientists support condition designations for components in this assessment. However, in some cases, data were unavailable or insufficient for several of the measures of the featured components. In other instances, data establishing reference condition were limited or unavailable for components, making comparisons with current information inappropriate or invalid. In these cases, it was not possible to assign condition for the components. Current condition was not able to be determined for four (sea ducks, breeding landbirds, mid-trophic level marine forage community, and air quality) of the 24 components (16.7%) due to these data gaps.

For those components with sufficient available data, the overall condition varied. Thirteen components were determined to be in good condition (marine shoreline, murrelets, glaucous-winged gulls, moose, bears, mountain goat, humpback whales, sea otters, harbor porpoise, Steller sea lions, anadromous fish, water quality, and wilderness). Five components (harvested marine invertebrates – Tanner crabs, Pacific halibut, western toad, plants, and underwater soundscape) were of moderate concern. Three components were determined to be of significant concern (glaciers, harvested marine invertebrates – weathervane scallops, and harbor seals). Detailed discussion of these designations is presented in Chapters 4 and 5 of this report.

Many of the data gaps identified for the components involved the continuation or expansion of existing monitoring efforts. Components such as glaciers and water quality are in need of continuation of data collection in order to accumulate enough data over time to identify trends. Other components, such as humpback whales and murrelets, are in need of expansion of their existing monitoring in order to see how that component's prey community is structured in GLBA. While components such as Pacific halibut and harvested marine invertebrates are in need of expanded knowledge regarding how they interact with their ecosystem. There are a few components in this assessment that are in need of an established monitoring effort (e.g., breeding landbirds, sea ducks), as there is not currently enough data to determine trend or condition in the park.

Several park-wide threats and stressors influence the condition of priority resources in GLBA. Those of primary concern include climate change, vessel traffic, and ecological succession patterns. Understanding these threats, and how they relate to the condition of park resources, can help the NPS prioritize management objectives and better focus their efforts to maintain the health and integrity of the park ecosystem.

Acknowledgments

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Acronyms and Abbreviations

AAF – Areas as Fleets

AAR – Accumulation Area Ratio

ACL – Annual Catch Limit

ADFG – Alaska Department of Fish and Game

AGAP – Alaska Gap Analysis Project

AIS – Automated Identification System

AK DEC – Alaska Department of Environmental Conservation

ANCOVA – Analysis of Covariance

ANILCA – Alaska National Interest Lands Conservation Act

ANOVA – Analysis of Variation

ARD – Air Resources Division

AVHRR – Advanced Very High Resolution Radiometer

BBP – Breeding Bird Phenology Program

BBS – Breeding Bird Survey

BCC – Bird Species of Conservation Concern

BCR – Bird Conservation Region

BCS – Bitter Crab Syndrome

BHIMS – Bear Human Information Management System

BLM – Bureau of Land Management

BLTR – Bartlett Lake Trail

BRFC – Bear Resistant Food Canisters

CAA – Clean Air Act of 1977

CBC – Christmas Bird Count

CL – Condition Level

CPUE – Catch per Unit Effort

Acronyms and Abbreviations (continued)

CS – Catch Survey

CSA – Catch Survey Analysis

CSP – Catch Sharing Plan

CTD – Conductivity-Temperature-Depth

CWD – Chronic Wasting Disease

DEM – Digital Elevation Model

DEVA – Death Valley National Park

DLP – Defense of Life or Property

DPS – Distinct Population Segments

eBio – Exploitable Biomass

EEZ – Economic Exclusion Zone

EIS – Environmental Impact Statement

ELA – Equilibrium Line Altitude

EPA – Environmental Protection Agency

EPMT – Alaska Exotic Plant Management Team

EVOS – Exxon Valdez Oil Spill

EXRI – Excursion Ridge

GAAR – Gates of the Arctic National Park and Preserve

GBP – Glacier Bay Proper

GHL – Guideline Harvest Level

GIS – Geographic Information System

GLBA – Glacier Bay National Park and Preserve

GMU – Game Management Unit

GPRA – Government Performance and Results Act

GPS – Global Positioning System

Acronyms and Abbreviations (continued)

GRTS – Generalized Random Tessellation Stratified Sampling

HCH – Hexachlorocyclohexanes

HIA – Hoonah Indian Association

HPUE – Harvest Rate per Unit Effort

HUC – Hydrologic Unit Codes

I&M – Inventory and Monitoring

IMPROVE – Interagency Monitoring of Protected Visual Environments Program

IPHC – International Pacific Halibut Commission

IRMA – Integrated Resource Management Application

IUCN – International Union for the Conservation of Nature

LAP – Lifetime Access Permit

MDN – Mercury Deposition Network

MLLW – Mean Lower Low Water

MMPA – Marine Mammal Protection Act

NAAQS – National Ambient Air Quality Standards

NALCP – North American Landbird Conservation Plan

NMFS – National Marine Fisheries Service

NMML – National Marine Mammal Laboratory

NMS – Non-metric Multidimensional Scaling

NOAA – National Oceanic and Atmospheric Administration

NOER – No-Effects-Residue

NPFMC – North Pacific Fishery Management Council

NPIW – North Pacific Intermediate Waters

NPS – National Park Service

NRCA – Natural Resource Condition Assessment

Acronyms and Abbreviations (continued)

NSWCD – Naval Surface Warfare Center Detachment

NWPS – National Wilderness Preservation System

OA – Ocean Acidification

PAH – Polycyclic Aromatic Hydrocarbons

PBDE – Polybrominated Biphenyl ether

PCB – Polychlorinated Biphenyls

PDO – Pacific Decadal Oscillation

PDV – Phocine Distemper Virus

PIF – Partners in Flight

PM – Particulate Matter

POP – Persistent Organic Pollutants

PRB – Policy Relevant Background

PSAT – Pop-up Satellite Archival Tag

PST – Pacific Salmon treaty

ROD – Record of Decision

SAFE – Stock Assessment and Fishery Evaluations

Sbio – Female Spawning Biomass

SEAN – Southeast Alaska Network

SEBE – Sealer's Beach

SL – Significance Level

SMUMN GSS – Saint Mary's University of Minnesota Geospatial Services

SOS – Saving our Shared Birds

SPL – Sound Pressure Levels

SPT – Scallop Plan Team

SSB – Spawning Stock Biomass

Acronyms and Abbreviations (continued)

SSF – Small Schooling Fish

SST – Sea Surface Temperature

TA – Total Alkalinity

TCEY – Total Constant Exploitation Yield

TPAH – Total Polycyclic Aromatic Hydrocarbons

UAF – University of Alaska Fairbanks

UNEP – United Nations Environment Programme

USFWS – U.S. Fish and Wildlife Service

USFS – U.S. Forest Service

USGS – U.S. Geological Survey

UW – University of Washington

VAHO – Van Horn

WCS – Weighted Condition Score

WMS – Weak-meated Scallop

WRF – Weather Research and Forecasting Model

Chapter 1. NRCA Background Information

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units, hereafter "parks." NRCAs also report on trends in resource condition (when possible), identify critical data gaps, and characterize a general level of confidence for study findings. The resources and indicators emphasized in a given project depend on the park's resource setting, status of resource stewardship planning and science in identifying high-priority indicators, and availability of data and expertise to assess current conditions for a variety of potential study resources and indicators.

> ***NRCAs Strive to Provide…***
>
> - *Credible condition reporting for a subset of important park natural resources and indicators*
> - *Useful condition summaries by broader resource categories or topics, and by park areas*

NRCAs represent a relatively new approach to assessing and reporting on park resource conditions. They are meant to complement—not replace—traditional issue-and threat-based resource assessments. As distinguishing characteristics, all NRCAs:

- Are multi-disciplinary in scope;[1]
- Employ hierarchical indicator frameworks;[2]
- Identify or develop reference conditions/values for comparison against current conditions;[3]
- Emphasize spatial evaluation of conditions and Geographic Information System (GIS) products;[4]
- Summarize key findings by park areas;[5] and
- Follow national NRCA guidelines and standards for study design and reporting products.

Although the primary objective of NRCAs is to report on current conditions relative to logical forms of reference conditions and values, NRCAs also report on trends, when appropriate (i.e., when the underlying data and methods support such reporting), as well as influences on resource conditions. These influences may include past activities or conditions that provide a helpful context for

[1] The breadth of natural resources and number/type of indicators evaluated will vary by park.

[2] Frameworks help guide a multi-disciplinary selection of indicators and subsequent "roll up" and reporting of data for measures ⇨ conditions for indicators ⇨ condition summaries by broader topics and park areas

[3] NRCAs must consider ecologically-based reference conditions, must also consider applicable legal and regulatory standards, and can consider other management-specified condition objectives or targets; each study indicator can be evaluated against one or more types of logical reference conditions. Reference values can be expressed in qualitative to quantitative terms, as a single value or range of values; they represent desirable resource conditions or, alternatively, condition states that we wish to avoid or that require a follow-up response (e.g., ecological thresholds or management "triggers").

[4] As possible and appropriate, NRCAs describe condition gradients or differences across a park for important natural resources and study indicators through a set of GIS coverages and map products.

[5] In addition to reporting on indicator-level conditions, investigators are asked to take a bigger picture (more holistic) view and summarize overall findings and provide suggestions to managers on an area-by-area basis: 1) by park ecosystem/habitat types or watersheds, and 2) for other park areas as requested.

understanding current conditions, and/or present-day threats and stressors that are best interpreted at park, watershed, or landscape scales (though NRCAs do not report on condition status for land areas and natural resources beyond park boundaries). Intensive cause-and-effect analyses of threats and stressors, and development of detailed treatment options, are outside the scope of NRCAs.
Due to their modest funding, relatively quick timeframe for completion, and reliance on existing data and information, NRCAs are not intended to be exhaustive. Their methodology typically involves an informal synthesis of scientific data and information from multiple and diverse sources. Level of rigor and statistical repeatability will vary by resource or indicator, reflecting differences in existing data and knowledge bases across the varied study components.

The credibility of NRCA results is derived from the data, methods, and reference values used in the project work, which are designed to be appropriate for the stated purpose of the project, as well as adequately documented. For each study indicator for which current condition or trend is reported, we will identify critical data gaps and describe the level of confidence in at least qualitative terms. Involvement of park staff and National Park Service (NPS) subject-matter experts at critical points during the project timeline is also important. These staff will be asked to assist with the selection of study indicators; recommend data sets, methods, and reference conditions and values; and help provide a multi-disciplinary review of draft study findings and products.

NRCAs can yield new insights about current park resource conditions, but, in many cases, their greatest value may be the development of useful documentation regarding known or suspected resource conditions within parks. Reporting products can help park managers as they think about near-term workload priorities, frame data and study needs for important park resources, and communicate messages about current park resource conditions to various audiences. A successful NRCA delivers science-based information that is both credible and has practical uses for a variety of park decision making, planning, and partnership activities.

Important NRCA Success Factors

- *Obtaining good input from park staff and other NPS subject-matter experts at critical points in the project timeline*
- *Using study frameworks that accommodate meaningful condition reporting at multiple levels (measures ⇨ indicators ⇨ broader resource topics and park areas)*
- *Building credibility by clearly documenting the data and methods used, critical data gaps, and level of confidence for indicator-level condition findings*

However, it is important to note that NRCAs do not establish management targets for study indicators. That process must occur through park planning and management activities. What an NRCA can do is deliver science-based information that will assist park managers in their ongoing, long-term efforts to describe and quantify a park’s desired resource conditions and management targets. In the near term, NRCA findings assist strategic park resource planning[6] and help parks to report on government accountability measures.[7] In addition, although in-depth analysis of the effects of climate change on park natural resources is outside the scope of NRCAs, the condition analyses and data sets developed for NRCAs will be useful for park-level climate-change studies and planning efforts.

NRCAs also provide a useful complement to rigorous NPS science support programs, such as the NPS Natural Resources Inventory & Monitoring (I&M) Program.[8] For example, NRCAs can provide current condition estimates and help establish reference conditions, or baseline values, for some of a park's vital signs monitoring indicators. They can also draw upon non-NPS data to help evaluate current conditions for those same vital signs. In some cases, I&M data sets are incorporated into NRCA analyses and reporting products.

NRCA Reporting Products…

Provide a credible, snapshot-in-time evaluation for a subset of important park natural resources and indicators, to help park managers:

- *Direct limited staff and funding resources to park areas and natural resources that represent high need and/or high opportunity situations* ***(near-term operational planning and management)***
- *Improve understanding and quantification for desired conditions for the park's "fundamental" and "other important" natural resources and values* ***(longer-term strategic planning)***
- *Communicate succinct messages regarding current resource conditions to government program managers, to Congress, and to the general public* ***("resource condition status" reporting)***

Over the next several years, the NPS plans to fund an NRCA project for each of the approximately 270 parks served by the NPS I&M Program. For more information visit the NRCA Program website.

[6] An NRCA can be useful during the development of a park's Resource Stewardship Strategy (RSS) and can also be tailored to act as a post-RSS project.

[7] While accountability reporting measures are subject to change, the spatial and reference-based condition data provided by NRCAs will be useful for most forms of "resource condition status" reporting as may be required by the NPS, the Department of the Interior, or the Office of Management and Budget.

[8] The I&M program consists of 32 networks nationwide that are implementing "vital signs" monitoring in order to assess the condition of park ecosystems and develop a stronger scientific basis for stewardship and management of natural resources across the National Park System. "Vital signs" are a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values.

Chapter 2. Introduction and Resource Setting

2.1. Introduction

2.1.1. Enabling Legislation

Glacier Bay National Park and Preserve (GLBA) was originally established as Glacier Bay National Monument on 25 February 1925, with 1.1 million ha (2.8 million ac) of land designated for protection. Presidential Proclamation 1733 was signed by Calvin Coolidge and aimed to preserve an area with

> *…a number of tidewater glaciers of the first rank in a magnificent setting of lofty peaks, and more accessible to ordinary travel than other similar regions of Alaska….a great variety of forest covering consisting of mature areas, bodies of youthful trees which have become established since the retreat of the ice which should be preserved in absolutely natural condition, and great stretches now bare that will become forested in the next course of the next century….a unique opportunity for the scientific study of glacial behavior and of resulting movements and developments of flora and fauna and of certain valuable relics of interglacial forests….historic interest, having been visited by explorers and scientists since the early voyages of Vancouver in 1794 who left valuable records of such visits and explorations (Proc. No. 1733, 43 Stat. 1988).*

In 1939, adjacent lands of the Tongass National Forest were added to the monument (Proc. No. 2330, 53 Stat. 2534), and in 1955 the total area of the monument was again enlarged when a portion of Excursion Inlet was added to the monument (Proc. No. 3089, 69 Stat. c27). President Jimmy Carter signed an additional Presidential Proclamation in 1978 that added approximately 550,000 ac to the park, primarily around Mount Fairweather, the Alsek River, and the Grand Plateau Glacier and Deception Hills areas (Proc. No. 4618, 93 Stat. 1458).

Glacier Bay National Monument transitioned to Glacier Bay National Park and Preserve on 2 December 1980 under the authority of the Alaska National Interest Lands Conservation Act (ANILCA) Section 202 (Public Law 96-487). During this transition, an additional 211,651 ha (523,000 ac) of land were added to the national park, and 23,067 ha (57,000 ac) of land near the Alsek River were set aside as national preserve lands. ANILCA also designated approximately 1.12 million ha (2.77 million ac) within the park as wilderness, under the conditions of the 1964 Wilderness Act. Following these acquisitions, the total area of GLBA reached nearly 1.3 million ha (3.3 million ac).

2.1.2. Geographic Setting

Glacier Bay National Park and Preserve is located in the panhandle of Southeast Alaska and includes a wide expanse of both terrestrial and marine areas (Figure 1). The total terrestrial area of GLBA falls just under 1.2 million ha (2.9 million ac) and consists primarily of mountains carved by glaciers with narrow beaches along the coast. Mountain ranges in GLBA include: the Fairweather Range, the Saint Elias Mountains, the Takhinsha Mountains, the Beartrack Mountains, and portions of the Alsek Range and Chilkat Range. At their highest, the mountains in the park reach approximately 4,663 m (15,298 ft) above sea level (Boggs et al. 2008). GLBA is considered to have significant glacial coverage, and according to Loso et al. (2014), glacial ice covered 5,323 km^2 (2,055 mi^2) of GLBA in 2010, an estimated 40.1% of the total park area.

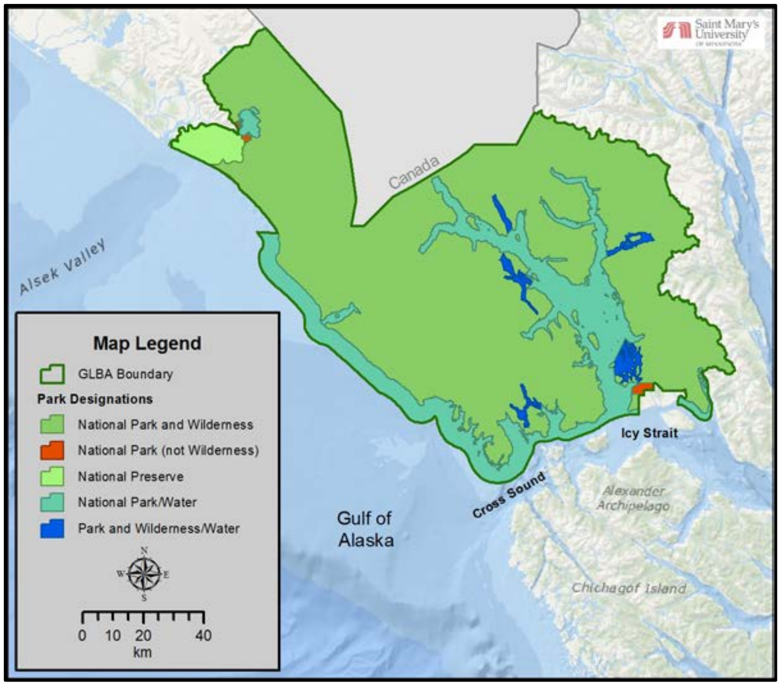

Figure 1. Map of the GLBA area showing terrestrial and marine park, wilderness, and preserve areas.

The marine areas of GLBA are dominated by Glacier Bay Proper (GBP) in the center of the park, Icy Strait and Cross Sound along the southern border of the park, and the Gulf of Alaska along the western boundary (Figure 1). Glacier Bay Proper is approximately 100 km (62 mi) in length, with widths varying from 15 km (9 mi) at mid-bay to 4-8 km (2.5-5 mi) in the lower and upper portions (Robards et al. 2003). Icy Strait and Cross Sound connect to the Gulf of Alaska; the gulf processes heavily influence lower Glacier Bay by way of this connectivity (Robards et al. 2003).

Data regarding the climate of the GLBA area are largely limited to only Bartlett Cove. NPS (1984) describes that GLBA can be divided into three climate “zones” (the outer coast along the Gulf of Alaska, the upper GBP region north of Tidal Inlet, and Bartlett Cove); however, this leaves much of the park unassigned to a climate “zone” (e.g., Brady Icefield, Beartrack Mountains). The weather zones of GLBA remains a significant data gap that will be updated in the future by the installation of climate monitoring stations throughout the park.

Bartlett Cove in the lower bay experiences temperatures in the summer around 16-18°C (60-65°F) and temperatures below freezing for several winter months (Table 1; NCDC 2015). Average annual precipitation at Bartlett Cove is nearly 200 cm (79 in) (NCDC 2015). The town of Gustavus, which lies just south of GLBA along Icy Strait, shows a climate similar to Bartlett Cove, but with slightly cooler low temperatures during the winter months and less annual average precipitation (Table 2; NCDC 2015).

Table 1. 30-year climate normals (1981-2010) for the Glacier Bay weather station at Bartlett Cove (NCDC 2015).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Average Temperature (°C)													
Max	0.6	1.7	3.8	8.6	13.2	16.3	17.6	16.8	12.9	8.2	3.3	1.4	8.7
Min	-4.0	-3.7	-2.6	0.0	3.4	6.5	8.3	8.1	5.6	2.3	-1.7	-3.2	1.6
Average Precipitation (cm)													
Total	18.0	16.6	11.3	8.6	11.4	7.7	12.1	16.4	27.5	29.3	20.7	19.9	199.7

Table 2. 30-year climate normals (1981-2010) for the Gustavus weather station southeast of GLBA (NCDC 2015).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Average Temperature (°C)													
Max	0.5	2.2	4.9	9.4	13.3	16.5	17.7	17.3	13.8	8.6	3.6	1.2	9.1
Min	-5.5	-4.9	-3.6	-0.3	3.2	6.4	8.9	8.3	5.0	1.4	-2.7	-3.9	1.1
Average Precipitation (cm)													
Total	13.7	11.7	8.2	6.8	7.6	7.2	10.5	13.3	19.4	20.8	16.3	15.1	150.8

2.1.3. Visitation Statistics

Despite the fact that GLBA is only accessible by watercraft or aircraft, the park is one of the most visited national park units in Alaska, and was behind only Klondike Gold Rush National Historic Park and Denali National Park and Preserve in visitation in 2014. The remote and undeveloped nature of GLBA attracts many tourists, recreational visitors, and scientists. The majority of the park's visitors arrive via large cruise ships that can carry thousands of passengers. In an effort to protect the park's critical glacially influenced ecosystems, the park finalized an Environmental Impact Statement (EIS) and Record of Decision (ROD) in 2003 (NPS 2003). Part of this EIS established a quota for vessels operating in the park. The superintendent annually determines the cruise ship quota (up to a maximum of 184 cruise ships during the peak season from June-August), based upon public comment and available scientific and other information, with no more than two cruise ships permitted to enter on a given day.

The only major road in the park runs between the town of Gustavus, AK and Bartlett Cove, where park headquarters and the visitor center are located; some dirt track roads exist near the Dry Bay portion of the national preserve as well. Annual visitation to GLBA has increased in almost every year since the 1950s, when record keeping began for the park. The average annual visitation over the past decade (2005-2014) has been just over 440,000 people, with an all-time high 500,727 visitors in 2014 (NPS 2015a).

2.2. Natural Resources

2.2.1. Ecological Units and Watersheds

GLBA includes portions of two Environmental Protection Agency (EPA) Level III Ecoregions: the Pacific Coastal Mountains and the Coastal Western Hemlock-Sitka Spruce Forest (Figure 2). The steep and rugged mountains along the Pacific Coast in Southeast Alaska receive more annual precipitation than other mountainous ecoregions of Alaska (EPA 2013). Much of the Pacific Coastal Mountains Ecoregion is covered by glaciers or icefields or is barren of vegetation; low scrub communities dominate where plants do occur (EPA 2013).

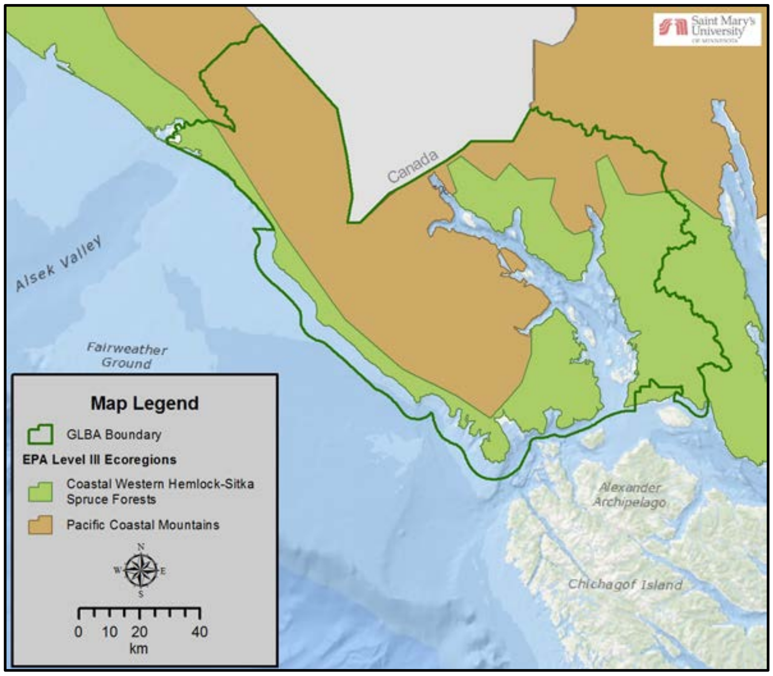

Figure 2. EPA Level III Ecoregions within GLBA (EPA 2011).

The irregular coastline, narrow bays, and steep valley walls of the Western Hemlock-Sitka Spruce Forest Ecoregion were shaped by glaciation (EPA 2013). This ecoregion experiences the mildest winter temperatures in Alaska and has substantial precipitation, which supports the forests of western hemlock (*Tsuga heterophylla*) and Sitka spruce (*Picea sitchensis*) (EPA 2013).

Based on U.S. Geological Survey (USGS) hydrologic unit codes (HUCs; 8-digit codes) (USGS 1994), GLBA can be divided into four watersheds. The entire eastern portion of the park and the coastline along Icy Strait is part of the Glacier Bay Watershed (Figure 3). Most of GLBA's outer coastal area falls within the Palma Bay Watershed, while the furthest northwest portion of GLBA falls within the Alsek River watershed (USGS 1994). A small section in the northwest portion of GLBA is considered part of the Tatshenshini River watershed, although this river eventually drains into the Alsek.

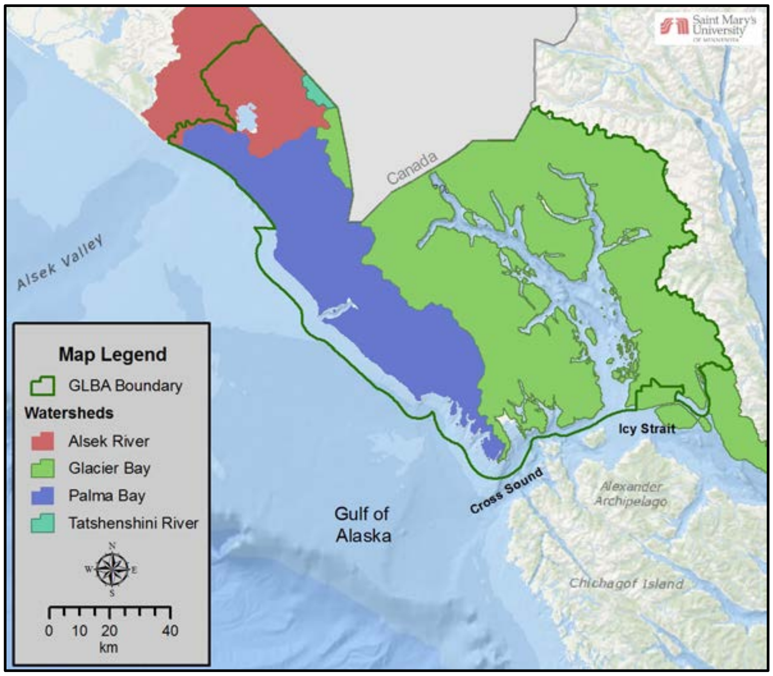

Figure 3. Watersheds (HUC 8) of GLBA (USGS 1994).

2.2.2. Resource Descriptions

An exceptional feature of GLBA is the almost entirely intact terrestrial ecosystem that functions largely unimpeded by anthropogenic activity and thus preserves a stunning example of pristine wilderness (NPS 2010). There is a tremendous diversity of flora and fauna supported by GLBA. Boggs et al. (2008) identified 42 land cover classes, from needleleaf forest to dwarf shrub and bare ground/rock, and 104 distinct plant associations (i.e., communities) (Photo 1). The most prevalent land cover classes are tall shrubs (e.g., alder and willow) and coniferous forest, comprising 29% and 25% of park area, respectively (Boggs et al. 2008). Over 500 vascular plant species have been confirmed in the park, along with more than 875 species of lichen (Spribille and Fryday 2012, USFS and NPS 2013, NPS 2015b, Spribille *In Prep.*).

Photo 1. Herbaceous and forested vegetation along Beartrack Cove (NPS photo from Carlson et al. 2004).

The park supports approximately 50 terrestrial and marine mammal species (NPS 2015b). Common terrestrial mammals (as defined by NPS 2015b) include bears (brown [*Ursus arctos*] and black [*U. americanus*]), wolves (*Canis lupus*), coyotes (*C. latrans*), moose (*Alces alces*), mountain goats (*Oreamnos americanus*), river otters (*Lontra canadensis*), and many small mammals (e.g., voles, shrews) (NPS 1984, 2015b). Among marine mammals, humpback whales (*Megaptera novaeangliae*), orca (*Orcinus orca*; Photo 2), sea otters (*Enhydra lutris*), and harbor porpoise (*Phocoena phoecona*) are common. Humpback whales were first reported in GLBA in 1899 (during the Harriman Expedition), but were not commonly reported in the park until the 1950s (Unpublished NPS Ranger Logs). Individually identified whales have been documented using GLBA as a feeding ground since the 1970s (Chris Gabriele, GLBA Wildlife Biologist, written communication, 7 August 2014). Humpback whales were identified as a Vital Sign within SEAN and have been monitored by the NPS in the park and surrounding waters since 1985 (Neilson et al. 2014).

Photo 2. Orcas in Glacier Bay (NPS photo).

GLBA supports over 250 species of birds, although some are present only during the summer or just pass through during migration (NPS 2015b). Large aggregations of seabirds (e.g., glaucous-winged gull [*Larus glaucescens*]) nest on the park's rocky shores, cliffs, and islands. Other prominent seabirds such as cormorants (*Phalacrocorax* spp.) and puffins (*Fratercula* spp.) are frequently observed in GBP. Glacier Bay is also home to a large proportion of the worlds marbled and Kittlitz's murrelet (*Brachyrampus marmoratus* and *B. brevirostrus*) populations, especially during the molting period. Sea ducks are common in the park, with large aggregations of scoters (*Melanitta* spp.), mergansers (*Mergus* spp.), and Barrow's goldeneyes (*Bucephala islandica*) common in the winter.

No reptile species are known to inhabit GLBA, but the park does support one native amphibian, the western toad (*Bufo boreas*) (NPS 2015b). This toad breeds in some of the park's shallow ponds and has even been documented in saltwater intertidal areas (Taylor 1983). A lack of regional knowledge regarding western toad distribution, population abundance, and habitat range prompted recent monitoring efforts in the SEAN (Klondike Gold Rush National Historic Park) (Wetherbee 2009). The IUCN Red List of Threatened Species lists the western toad as near threatened with a decreasing population trend; the significant rate of decline is likely due to a myriad of threats, including habitat loss, disease, predation, and other synergistic effects (Hammerson et al. 2004).

The NPS Certified Species List (NPS 2015b) confirms the presence of approximately 100 fish species in GLBA waters and identifies an additional 225 which are considered “probably present.” Nine anadromous fish species (i.e., fish born in freshwater that live in the marine environment and return to freshwater to spawn) occur in the waters of GLBA, including five species of salmon (*Oncorhynchus* spp.), steelhead trout (*O. mykiss*), sea run cutthroat trout (*O. clarki*), Dolly Varden char (*Salvelinus malma*), and eulachon smelt (*Thaleichthys pacificus*). Other fish in GLBA waters include small forage fishes and many species commonly associated with commercial fisheries (e.g., Pacific halibut [*Hippoglossus stenolepis*], rockfish [*Sebastes* spp.], and Pacific cod [*Gadus macrocephalus*]).

The park’s waters and tidal areas also support numerous marine invertebrates, such as barnacles, mussels, snails, crabs, and sea urchins. These invertebrates, as well as some fish, are a key prey resource for many of the larger birds and mammals in the park (Springer and Speckman 1997, Robards et al. 2003). Additionally several marine invertebrates are harvested in the GLBA area, with species such as the Tanner crab (*Chionoecetes bairdi*) and weathervane scallop (*Patinopecten caurinus*) representing species currently being harvested that are of significant interest to park managers.

The glaciers of GLBA are considered one of the park’s most important natural resources and are a sight that attracts many visitors annually (Photo 3). There are two types of glaciers in GLBA: tidewater and terrestrial glaciers. Tidewater glaciers are those that terminate in water, whereas terrestrial glaciers terminate on land. Tidewater glacial calving is a common sight-seeing attraction; calving is a natural process that occurs as pieces of glacial ice break off from the main glacier body and fall into the ocean, often creating large waves and a rushing sound as they drop into the water. Tidewater glaciers can be viewed in the upper west and east arms of Glacier Bay (Figure 4).

Photo 3. A tour boat approaching a glacier in Glacier Bay (NPS photo).

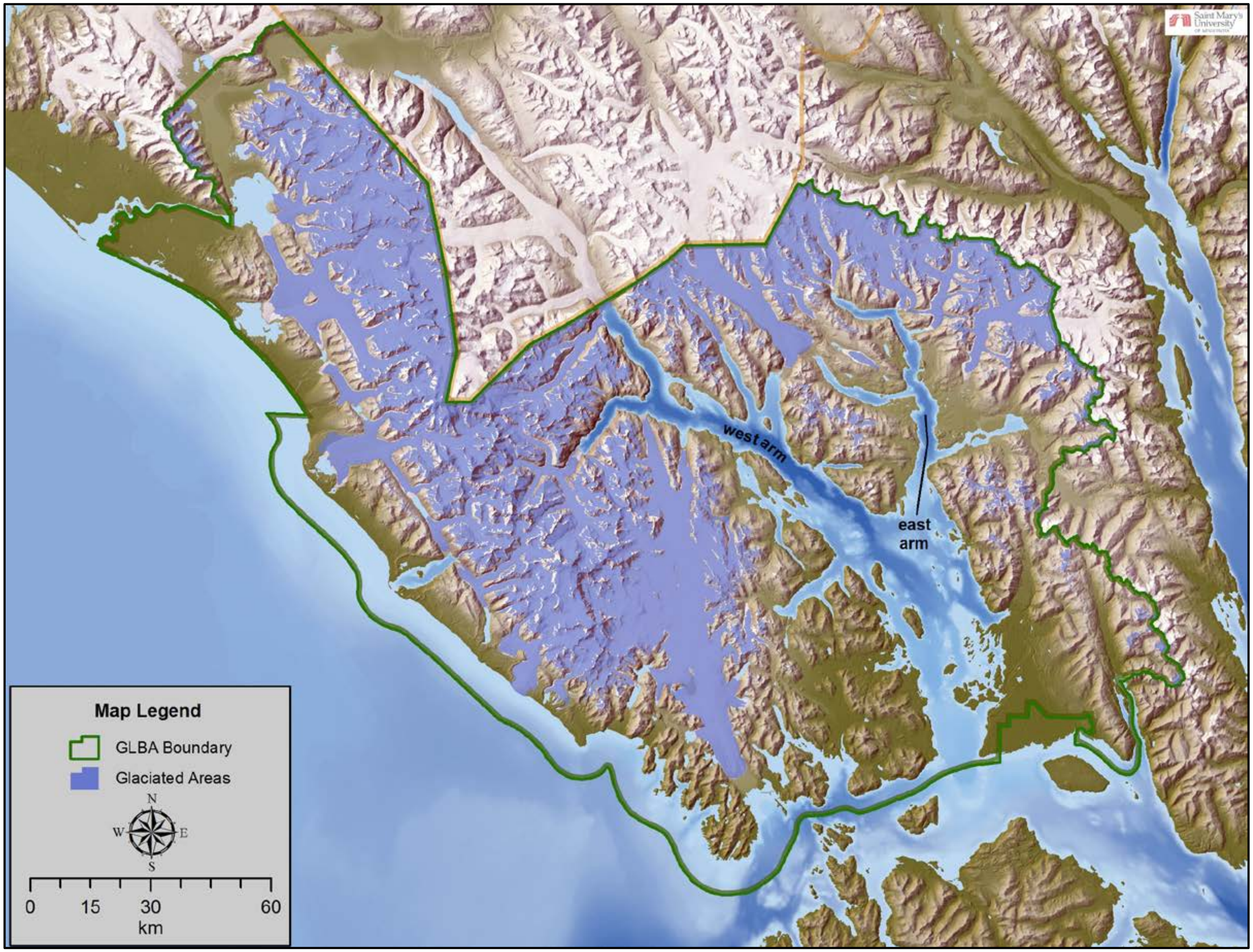

Figure 4. Glacial coverage within GLBA (NPS 2013).

Ecological Succession

GLBA is widely recognized as one of the best documented examples of terrestrial primary succession following deglaciation (Chapin et al. 1993). Most of GBP was filled by a glacier terminating in Icy Strait as recently as 200 years ago. As a recently deglaciated fjord, Glacier Bay provides scientists with a rare opportunity to observe natural succession as it unfolds in front of them. Very few flora or fauna existed until the glacier retreated and plant succession began. The process continues today and the many years of this phenomenon recorded within GLBA represents perhaps the most studied area of ecological succession in the world (Cooper 1923, 1931, 1939; Lawrence 1958; Reiners et al. 1971; Walker 1999).

Glacial retreat and isostatic rebound continually creates and removes habitat throughout the park; because of this, biotic and abiotic communities are constantly adjusting and moving. While these ecological communities in the park are transitioning due to succession, this is by no means to be viewed as a detriment to the overall health of the ecosystem. According to NPS (2010, p. 10),

> *Agents and processes of disturbance (e.g., glaciers, earthquakes, floods, erosion, insect irruptions), along with those of ecosystem recovery (e.g., biological succession, landform evolution) are allowed to exist and proceed free of human influence. Fluctuations in animal and plant populations (driven by any natural cause) are allowed to occur and reach their own states of equilibrium.*

Four primary stages of vegetation succession are generally recognized within the park (Chapin et al. 1993). The first is a "pioneer community" of blue-green algae that form a crust on the newly exposed surface. The second stage consists of a mat of mountain avens (*Dryas* spp.) with scattered alders (*Alnus viridis*), willows (*Salix* spp.), and cottonwood (*Populus balsamifera* ssp. *trichocarpa*). This is followed by a stage with dense thickets of alder, and finally a mature spruce forest dominated by Sitka spruce (*Picea sitchensis*) (Cooper 1923, Chapin et al. 1993).

Similar to plant succession, glacial retreat is a natural phenomenon and the rate at which glaciers retreat can be highly variable. In the case of GLBA, glacial advancement and retreat has occurred rapidly in geologic terms (Figure 5). The 1750 pane below represents the maximum extent of glacial coverage in Glacier Bay, and just over 100 years later the glacier had retreated revealing all of GBP and much of the west arm (Figure 5, 1880 pane). While some of the glaciers in the park are currently advancing, many of the park's remaining glaciers continue to retreat, as the total extent of glacial coverage in the park has decreased approximately 15% since 1950 (Loso et al. 2014).

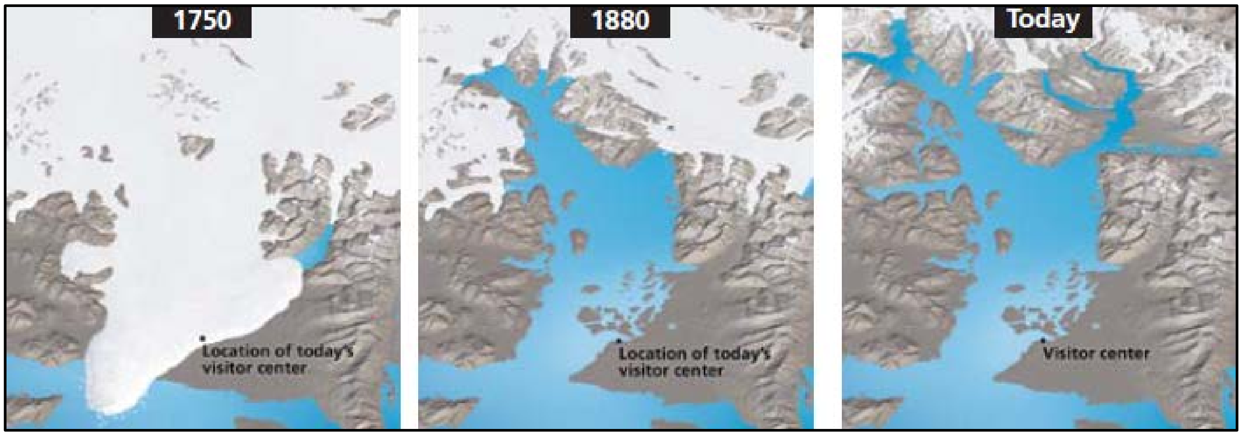

Figure 5. A visualization of glacial advancement and retreat in the GLBA area. Figure reproduced from NPS (no date).

2.2.3. Resource Issues Overview

The remoteness and protected status of GLBA may give the impression that its resources are largely undisturbed and in a pristine, natural condition. While this is true of some areas, others are threatened by anthropogenic stressors. For example, surveys of limnic, biotic, and tropospheric systems throughout Alaska's wild lands have shown detectable levels of industrial or agricultural contaminants (Shirokauer and Moynahan 2010). Industrial air pollution sources are rare in Alaska, but airborne contaminants can be transported long distances, even over the Pacific Ocean from Asia to Southeast Alaska (Landers et al. 2008). The presence of these contaminants poses a threat to GLBA's natural resources and human health. Persistent organic pollutants (POPs) such as pesticides and mercury can accumulate in organisms with long life-histories and/or at upper trophic levels, eventually reaching concentrations that can impact reproduction and/or survival (Shirokauer and Moynahan 2010). Contaminants are not limited to airborne sources, however, as several water borne sources of contaminants exist in the region as well (e.g., marine debris that carries POPs, microplastics). The current distribution, density, or total amount of contaminant loads in marine debris and microplastics represents an area that is in need of further research, but is a threat to the area regardless. The continual washing ashore of marine debris is an issue for GLBA along both the outer coast and in GBP, and when originating from foreign sources marine debris may serve as a vector for contaminants or invasive species.

Climate change is an issue of concern for GLBA, as climate is one of the most important factors influencing ecosystems, particularly at high latitudes. Climate-related changes in GLBA could: cause sea level rise (although isostatic rebound in GLBA may keep up with sea level rises, resulting in no net gain), cause glacier loss, contribute directly and indirectly to ocean acidification, increase disease and pest outbreaks, and cause shifts in plant and wildlife distribution and phenology (Hitch and Leberg 2007, Scenarios Network for Alaska Planning 2009, Sharman 2010).

Alaska's climate fluctuates on multiple temporal scales, including several natural cycles. One climate fluctuation of particular importance in Alaska is the Pacific Decadal Oscillation (PDO) (Lindsay

2011). The PDO, which is related to sea surface temperatures in the northern Pacific Ocean, affects atmospheric circulation patterns and alternates between positive and negative phases (Wendler and Shulski 2009). A positive phase is associated with a relatively strong low pressure center over the Aleutian Islands which moves warmer air into the state, particularly during the winter (Wendler and Shulski 2009). Hartmann and Wendler (2005) found that much of the warming that occurred in Alaska during the last half of the twentieth century was likely due to the PDO shift in 1976-77. However, there is a scientific consensus that human activities, particularly those that produce greenhouse gases, have contributed to a general warming trend in global climate (IPCC 2007). The Scenarios Network for Alaska Planning (2009) provided some projected estimates regarding changes in temperature through 2080. Currently, the average annual temperature within GLBA is 1.9°C (35.4°F); this average is expected to rise to 5.3°C (41.5°F) by 2080, an increase of about 0.6°C (1°F) per decade. Mean winter temperatures could rise more dramatically, by as much as 3.9°C (7°F) by 2080.

The park is somewhat protected from invasive plant species since many of the annual visitors arrive and depart on cruise ships, so there is a limited amount of ground traffic in the area (Fisk 2011). However, on-the-ground visitor use (e.g., kayaking and backcountry camping) can lead to invasive species introduction. Invasive species infestations have primarily been documented in or near developed or frequently visited areas including Bartlett Cove, Dry Bay, and Strawberry Island (Fisk 2011). No invasive fauna of particular concern have been detected at GLBA, although several non-native species are known or thought to occur (e.g., Eurasian collared-doves [*Streptopelia decaocto*], Atlantic salmon [*Salmo salar*], and brown-headed cowbirds [*Molothrus ater*]) (Fisk 2011, NPS 2015b). Monitoring has occurred for several potentially damaging invasive invertebrates (European green crab [*Carcinus maenas*], tunicates [e.g., *Didemnum vexillum, Botrylloides violaceous, Botryllus schlosseri*], gypsy moth [*Lymantria dispar*]), but to date these organisms have not been detected in the park (Fisk 2011). It's also important to note that invasive species vectors are not limited to ground traffic. Wind traffic is one non-ground dispersal method, and several bird species have been shown to be vectors for invasive plants (La Rue 1994, Renne et al. 2002, Bartuszevige and Gorchov 2006).

Cruise ship traffic in GBP represents a threat to many biological communities within the park. Air pollution from cruise ships associated with increased tourism throughout Southeast Alaska is a concern in the park. Plumes of exhaust are often observed from ships travelling through the fjords of GLBA, and nitrogen and sulfur deposition are known to be elevated at Sitka National Historical Park south of GLBA, another popular cruise ship destination (Geiser et al. 2010). According to Vequist (1993), air pollution in GLBA dates back to the 1970s and has potentially increased as a result of increased cruise ship visits. Boating activity, such as cruise liners, create visual blue-grey emissions, which results in impaired scenic vistas (Vequist 1993). Vessel noise represents a threat to the natural underwater soundscape inside the park because it has the potential to negatively affect a variety of marine wildlife (e.g., larval fish, invertebrates, harbor seals [*Phoca vitulina*], and whales) in a variety of ways.

It is likely that many of the animals that frequent GLBA are affected by vessel traffic in some manner. Disturbance during nesting, foraging, or molting is likely a threat for birds, including shore-nesting seabirds, murrelets, and sea ducks. Mammalian species such as harbor seals, humpback whales, and sea otters also have the potential to be disturbed by vessels while foraging, resting, and transitioning. There also exists the possibility of accidental mammalian or avian strike by cruise ships or other vessels. All types of vessels (motorized and non-motorized) performing various activities have been known to hit humpback whales (Neilson et al. 2012). According to Neilson et al. (2012), small, medium, and large vessels all strike whales, but small vessel strikes are most common in Southeast Alaska.

2.3. Resource Stewardship

2.3.1. Management Directives and Planning Guidance

The Alaska National Interest Lands Conservation Act established several national park units in Alaska, and defined the general purpose of the conservation areas established by the Act as:

> *…to preserve for the benefit, use, education, and inspiration of present and future generations certain lands and waters in the State of Alaska that contain nationally significant natural, scenic, historic, archeological, geological, scientific, wilderness, cultural, recreational, and wildlife (ANILCA, PL 96-487Sec. 101 9[a]).*
>
> *…to preserve unrivaled scenic and geological values associated with natural landscapes; to provide for the maintenance of sound populations of, and habitat for, Wildlife species of inestimable value to the citizens of Alaska and the Nation, including those species dependent on vast relatively undeveloped areas; to preserve in their natural state extensive unaltered arctic tundra, boreal forest, and coastal rainforest ecosystems, to protect the resources related to subsistence needs; to protect and preserve historic and archeological sites, rivers, and lands, and to preserve wilderness resource values and related recreational opportunities including but not limited to hiking, canoeing fishing, and sport hunting, within large arctic and subarctic lands and on free flowing rivers; and to maintain opportunities for scientific research and undisturbed ecosystems (ANILCA, PL 96-487 Sec. 101 (b)).*
>
> *It is further the intent and purpose of this Act consistent with management of fish and wildlife in accordance with recognized scientific principles and the purposes for which each conservation system unit is established, designated, or expanded by or pursuant to this Act, to provide the opportunity for rural residents engaged in a subsistence way of life to continue to do so (ANILCA, PL 96-487 Sec. 101 (c)).*

The Act also contains specific management directives for GLBA, specifically that hunting and subsistence use shall be permitted in Glacier Bay National Preserve and that no fee shall be charged for admittance to the park (ANILCA, PL 96-487 Sec. 203). Additionally, ANILCA provides direction for the commercial fishery industry that exists in the GLBA region, stating that the Secretary can take

… no action to restrict unreasonably the exercise of valid commercial fishing rights or privileges obtained pursuant to existing law, including the use of public lands for campsites, cabins, motorized vehicles, and aircraft landings on existing airstrips, directly incident to the exercise of such rights or privileges, except that this prohibition shall not apply to activities which the Secretary, after conducting a public hearing in the affected locality, finds constitute a significant expansion of the use of park lands beyond the level of such use during 1979 (ANILCA PL 96-487 Sec. 205).

One of the provisions of ANILCA (Section 1301) was that all national park units established by the act were required to prepare a conservation and management plan. This general management plan was completed in 1984 (NPS 1984), and the natural resource management section of this plan describes that

… management programs will ensure the continuity of support for internationally important research efforts on glaciers, plant succession, marine biology, and wildlife. Support will be provided when available to further research efforts. Carrying capacities will be determined to prevent degradation of resources. Regulations to protect humpback whales in Glacier Bay are being revised in accordance with the biological opinion of the National Marine Fisheries Service, issued in June 1983.

Traditional commercial fishing practices will be allowed throughout non-wilderness park and preserve waters and will be subject to regulations by the NPS and the Alaska Department of Fish and Game (ADFG). Commercial fishing in wilderness waters will be prohibited, in accordance with ANILCA and the Wilderness Act. Sport fishing will be allowed in accordance with NPS and ADFG regulations. Motorized vessels may be eliminated from wilderness waters in phases.

The subsistence harvest of fish, wildlife, and related resources on federal lands and waters in Alaska is controlled by provisions of ANILCA. Consequently, such activities will be allowed in the national preserve, but not in the national park. Management activities will provide and protect the opportunity for rural residents engaged in a subsistence way of life to continue those activities basic to their well-being (NPS 1984, p. iii).

GLBA has a wilderness visitor use management plan that covers visitor use and activities within the designated wilderness areas (NPS 1989). Plans like these fulfill the NPS policy on wilderness preservation and management by developing and maintaining a wilderness stewardship plan to guide the preservation, management, and use of wilderness resources (NPS 1989). The specific purpose and objectives of this wilderness management plan was to:

- Allow ecological processes to continue unimpaired by visitor use activities and patterns;
- Preserve opportunities for outstanding aquatic and terrestrial wilderness experiences;
- Protect specific sensitive species of wildlife and vegetation from adverse effects of visitor use;

- Provide opportunity for visitors to gain a greater understanding of the park resources and values to help heighten their enjoyment of their visit;
- Minimize the effects that motorized uses such as aircraft and motor boats may have on wilderness values and experiences;
- Develop a greater understanding through research, of those issues that are important to the other objectives mentioned above;
- Monitor and evaluate the effects of the management program to provide information for modifications;
- Make adjustments/modifications in the management program as needed based on monitoring and research information and public review and comment (NPS 1989, p. 2-3).
- More recently, a purpose and significance statement was identified in GLBA's 2010 foundation statement (NPS 2010). NPS (2010, p. 4) states:

> *The purpose of GLBA is to protect a dynamic tidewater glacial landscape and associated natural successional processes for science and discovery in a wilderness setting.*
>
> *Significance statements:*
>
> 1. *GLBA fosters unique opportunities for scientific studies of tidewater glacial landscapes and associated natural successional processes.*
> 2. *GLBA gathers and protects records of exploration, scientific endeavor and human use, and provides for understanding of the landscape through the lens of human experience and study.*
> 3. *GLBA protects ecological integrity by preserving a diversity of large, contiguous, intact ecosystems (from the highest peaks of the Fairweather Range to the open Pacific Ocean and sheltered inland fjords) that are strongly dominated by natural processes.*
> 4. *GLBA protects a natural biophysical landscape that is continually changing through large-scale natural disturbance followed by the biological succession of plants and animals, and accompanied by an evolving physical environment.*
> 5. *Glacier Bay National Park preserves one of the largest units of the national wilderness preservation system, encompassing more that 2.7 million acres of glacially influenced marine, terrestrial, and freshwater ecosystems.*
> 6. *Glacier Bay National Park preserves one of the largest (nearly 600,000 acres) areas of federally protected marine ecosystems in Alaska (including submerged lands) against which other less protected marine ecosystems can be compared.*
> 7. *GLBA lies within two Tlingit ancestral homelands that are of cultural and spiritual significance to living communities today.*
> 8. *GLBA provides diverse opportunities for visitors to experience a dynamic tidewater glacial landscape.*

9. *GLBA protects the remote and wild character of the Alsek River as a significant route of discovery and migration through the coastal mountain range to the Pacific Ocean.*
10. *Glacier Bay National Preserve protects a productive, evolving, glacial outwash ecosystem at the terminus of the Alsek River and provides a setting for subsistence uses, commercial fishing activities, and hunting as outlined by the Alaska National Interest Lands Conservation Act (NPS 2010).*

2.3.2. Status of Supporting Science

The Southeast Alaska Network (SEAN) identifies key resources network-wide and for each of its parks that can be used to determine the overall health of the parks. These key resources are called Vital Signs. In 2008, the SEAN completed and released a Vital Signs Monitoring Plan (Moynahan et al. 2008). Table 3 shows the network vital signs selected for monitoring in GLBA. Monitoring vital signs could be important to GLBA to ensure early detection of environmental degradation; with the rapid increase in human population there will also be increased use of resources such as land, water, and energy as areas become more developed and increased industrial activities take a larger toll on natural systems (Shirokauer and Moynahan 2010).

Table 3. SEAN Vital Signs selected for monitoring in GLBA (Moynahan et al. 2008).

Category	SEAN Vital Signs
Air and Climate	*Airborne contaminants*[A], *weather and climate*[A]
Geology & Soils	*Glacial dynamics*[A], **streamflow**[B], *oceanography*[A]
Water	*Freshwater contaminants*[A], *freshwater water quality*[A], and *marine contamination*[A]
Biological Integrity	**Invasive/non-native plants**[B], *marine predators*[A], *Kittlitz's murrelets*[A], western toads[A], **humpback whales**[B], **harbor seals**[B]
Human Use	**Human uses and mode of access**[B], **underwater sound**[B]
Landscapes (Ecosystem Patterns and Processes)	*Landform and land cover*[A]

[A] *Italics* indicate Vital Signs for which the network will implement monitoring protocols in concert with other networks, using funding from the Vital Signs or water quality monitoring programs.

[B] **Bold** indicates Vital Signs being monitored by a network park, another NPS program, or another federal or state agency, using other funding. The network will collaborate with or supplement these efforts.

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Chapter 3. Study Scoping and Design

This NRCA is a collaborative project between the NPS and Saint Mary's University of Minnesota Geospatial Services (SMUMN GSS). Project stakeholders include the GLBA resource management team and SEAN Inventory and Monitoring Program staff. Before embarking on the project, it was necessary to identify the specific roles of the NPS and SMUMN GSS. Preliminary scoping meetings were held, and a task agreement and a scope of work document were created cooperatively between the NPS and SMUMN GSS.

3.1. Preliminary Scoping

A preliminary scoping meeting was held on 18-20 November 2013. At this meeting, SMUMN GSS and NPS staff confirmed that the purpose of the GLBA NRCA was to evaluate and report on current conditions, critical data and knowledge gaps, and selected existing and emerging resource condition influences of concern to GLBA managers. Certain constraints were placed on this NRCA, including the following:

- Condition assessments are conducted using existing data and information;
- Identification of data needs and gaps is driven by the project framework categories;
- The analysis of natural resource conditions includes a strong geospatial component;
- Resource focus and priorities are primarily driven by GLBA resource management.

This condition assessment provides a "snapshot-in-time" evaluation of the condition of a select set of park natural resources that were identified and agreed upon by the project team. Project findings will aid GLBA resource managers in the following objectives:

- Develop near-term management priorities (how to allocate limited staff and funding resources);
- Engage in watershed or landscape scale partnership and education efforts;
- Consider new park planning goals and take steps to further these;
- Report program performance (e.g., Department of Interior Strategic Plan "land health" goals, Government Performance and Results Act [GPRA]).

Specific project expectations and outcomes included the following:

- For key natural resource components, consolidate available data, reports, and spatial information from appropriate sources including: GLBA resource staff, the NPS Integrated Resource Management Application (IRMA) website, Inventory and Monitoring Vital Signs program, and available third-party sources. The SEAN report will provide a resource assessment and summary of pertinent data evaluated through this project.
- When appropriate, define a reference condition so that statements of current condition may be developed. The statements will describe the current state of a particular resource with respect to an agreed upon reference point.
- Clearly identify "management critical" data (i.e., those data relevant to the key resources). This will drive the data mining and gap definition process.

- Where applicable, develop geographic information system (GIS) products that provide spatial representation of resource data, ecological processes, resource stressors, trends, or other valuable information that can be better interpreted visually.
- Utilize “gray literature” and reports from third party research to the extent practicable.

3.2. Study Design

3.2.1. Indicator Framework, Focal Study Resources and Indicators

Selection of Resources and Measures

As defined by SMUMN GSS in the NRCA process, a “framework” is developed for a park or preserve. This framework is a way of organizing, in a hierarchical fashion, bio-geophysical resource topics considered important in park management efforts. The primary features in the framework are key resource components, measures, stressors, and reference conditions.

“Components” in this process are defined as natural resources (e.g., birds, plant communities), ecological processes or patterns (e.g., natural fire regime), or specific natural features or values (e.g., geological formations) that are considered important to current park management. Each key resource component has one or more “measures” that best define the current condition of a component being assessed in the NRCA. Measures are defined as those values or characterizations that evaluate and quantify the state of ecological health or integrity of a component. In addition to measures, current condition of components may be influenced by certain “stressors,” which are also considered during assessment. A “stressor” is defined as any agent that imposes adverse changes upon a component. These typically refer to anthropogenic factors that adversely affect natural ecosystems, but may also include natural processes or disturbances such as floods, fires, or predation (adapted from GLEI 2010).

During the GLBA NRCA scoping process, key resource components were identified by NPS staff and are represented as “components” in the NRCA framework. While this list of components is not a comprehensive list of all the resources in the park, it includes resources and processes that are unique to the park in some way, or are of greatest concern or highest management priority in GLBA. Several measures for each component, as well as known or potential stressors, were also identified in collaboration with NPS resource staff.

Selection of Reference Conditions

A “reference condition” is a benchmark to which current values of a given component’s measures can be compared to determine the condition of that component. A reference condition may be a historical condition (e.g., flood frequency prior to dam construction on a river), an established ecological threshold (e.g., EPA standards for air quality), or a targeted management goal/objective (e.g., a bison herd of at least 200 individuals) (adapted from Stoddard et al. 2006).

Reference conditions in this project were identified during the scoping process using input from NPS resource staff. In some cases, reference conditions represent a historical reference before human activity and disturbance was a major driver of ecological populations and processes, such as “pre-fire suppression." In other cases, peer-reviewed literature and ecological thresholds helped to define appropriate reference conditions.

Finalizing the Framework

An initial framework was adapted from the organizational framework outlined by the H. John Heinz III Center for Science's "State of Our Nation's Ecosystems 2008" (Heinz Center 2008). Key resources for the park were primarily adapted from the SEAN Vital Signs monitoring plan (Moynahan et al. 2008). This initial framework was presented to park resource staff to stimulate meaningful dialogue about key resources that should be assessed. Significant collaboration between SMUMN GSS analysts and NPS staff was needed to focus the scope of the SEAN project and finalize the framework of key resources to be assessed.

The NRCA framework was finalized following acceptance from NPS resource staff. It contains a total of 24 components (Figure 6) and was used to drive analysis in this NRCA. This framework outlines the components (resources), most appropriate measures, known or perceived stressors and threats to the resources, and the reference conditions for each component for comparison to current conditions.

GLBA Framework 1-5-2016

Natural Resource Condition Assessment Framework

Component	Measures (Significance Level)/Analysis Project	Stressors	Reference Condition
Ecosystem Extent and Function			
Glaciers	Glacial extent (3), glacial surface elevation (3)	Climate change, anthropogenic climate change	11% decrease in glaciation b/w '52 and 2010.
Marine shoreline	Abundance/distribution of: 1) anthropogenic marine debris in inter/supratidal (incl. vessel wrecks) (3); 2) stranded contaminants (3); 3) camper/user biophysical impacts (1)	Intentional/unintentional release by seagoing vessels and shoreside campers/users; uncontrolled shoreside visitor use.	Pristine or near-pristine: free of all anthropogenic objects and contaminants; biophysical impacts from shoreside park users are within designated limits (TBD).
Biotic Composition			
Harvested Marine Invertebrates (*Note that weathervane scallops and Tanner crabs will each receive their own WCS)	**Tanner Crabs:** Fishing effort and harvest (2), estimated stock abundance (3), health and incidence of disease (3); **Weathervane Scallops:** Fishing effort and harvest (3), shell height distribution (2), health and incidence of disease (2), habitat status (3)	Harvest/associated gear impact (especially for Weathervane scallops where spatial extent, frequency and impact could constitute impairment), environmental change, disease (Hematodidium in Tanner crab, weak meat scallops?)	**Tanner Crabs:** *fishing effort and harvest* - 1968 fishing season in stat/reg area A, *estimated stock abundance* - Bishop et al. (2013), *health and incidence of disease* - zero; **Weathervane scallop:** *fishing effort and harvest* - SAFE data from 1994 scallop season, *shell height distribution* - range in shell height of retained scallops in 1997, *health and incidence of disease* - no reference condition, *habitat status* - no reference condition
Birds			
Murrelets (Kittlitz's and Marbled)	Abundance in surveyed areas (3), spatial distribution of Kittlitz's murrelet within surveyed areas (3)	Altered habitat resulting from climate change (marine and terrestrial), varied availability of forage, disturbance from sailing vessels (all marine vessels)	Average abundance estimates from SEAN monitoring (2009-2015).
Sea Ducks	Species Richness (2), abundance of molting focal species (3), abundance of wintering focal species (3), winter age ratio of focal species (2), foraging effort of molting and wintering focal sea duck species (2)	Changes in abundance of forage species, climate change, contaminants, sea otters, vessel traffic, oil spills	Reference condition does not exist at this time
Breeding Landbirds	Species richness (3), species abundance (3), trends in apecies of conservation concern (2)	climate change, externally-derived environmental contaminants, diseases, sources of direct mortality, habitat alteration and destruction during the non-breeding season (for migratory breeding landbirds)	Reference condition does not exist at this time
Glaucous-winged gulls	Distribution (2), population trends in select areas (3), number of eggs harvested by location (3)	Plant succession, possibly egg harvest	Arimitsu et al. (2007), Zador and Piatt (1999), Zador (2001)

Figure 6. Glacier Bay National Park natural resource condition assessment framework.

GLBA Framework 1-5-2016

Natural Resource Condition Assessment Framework

Component	Measures (Significance Level)/Analysis Project	Stressors	Reference Condition
Ecosystem Extent and Function			
Mammals			
Moose	Distribution (2), abundance (2), population trends in select areas (3), age/sex ratios (2)	Habitat change, plant succession (sped up by climate change), harvest (in a couple of areas), predation, disease	Reference condition does not exist at this time
Bears	Black bear distribution (2), brown bear distribution (2), black bear numbers (3), brown bear numbers (3), black bears harvested (3), brown bears harvested (3)	Human conflict/caused mortality, harvest/DLP, interspecific competition, habitat change, plant succession	No reference condition for distribution or number of bears. Harvest = no more than 10% harvest of the bear population.
Mountain goat	Distribution (2), abundance (2), population trends in select areas (3), age/sex ratios (2)	Rising tree line, climate change, harvest, predation	Reference condition does not exist at this time.
Harbor seals	Population trends at ice sites (3), population trends at terrestrial sites (3), pup proportion (2)	Predation by killer whales, disease, pollutants, virus; legal harvest (inside or outside GLBA), illegal harvest (inside GLBA), incidental take/mortality as bycatch in fishing gear (outside GLBA), vessel/human disturbance, climate change (ice habitat and terrestrial habitat changes), competition	Stable or increasing 10-year average population trend (for both terrestrial and ice sites). Increasing or stable pup proportion (most years between 30-38%)
Humpback whales	Distribution (2), population estimate (2), crude birth rate (3)	Disturbance from underwater vessel noise, vessel strikes (lethal and non-lethal), entanglement in fishing gear, unknown changes in prey resources, competition,	Distribution - 1985 distribution; population estimate - the rate of population increase from 1985-2012 (4.4%); crude birth rate - median (8.7) and interquartile range (6.7-12.1) from 1984-2014;
Sea Otters	Distribution, abundance, and trends (3), foraging effort (3), kelp abundance and invertebrate prey abundance and size (2), strandings (1)	Predation, (disease, pollutants, virus); legal harvest (outside GLBA), illegal harvest (inside GLBA), vessel/human disturbance (and collisions), intra-specific competition (prey limitations)	Population characteristics at time prior to extirpation - few data exist, but non-food limited populations' average growth rate is 18% (Bodkin and Monson 2003).
Harbor porpoise	Distribution on NOAA tracklines (1), distribution on MARMAM database (2), density and abundance (NOAA) (2), sightings per year (2), strandings per year (1)	Predation by killer whales, disease, pollutants, virus; incidental take/mortality as bycatch in fishing gear (outside GLBA), vessel/human disturbance, climate change as it effects habitat or prey availability	Distribution - Dahlheim and Waite (2006), Dahlheim et al. (2012). Abundance and density - Dahlheim et al. (2012). Sightings - Gabriele et al. (2011). Strandings - NOAA (2013).

Figure 6 (continued). Glacier Bay National Park natural resource condition assessment framework.

GLBA Framework 1-5-2016
Natural Resource Condition Assessment Framework

	Component	Measures (Significance Level)/Analysis Project	Stressors	Reference Condition
Ecosystem Extent and Function				
Mammals				
	Steller sea lions	Annual estimates of population trends (3), spatial and seasonal distribution at haul-outs and rookeries (2), age-specific survival rates (3)	Health threats (disease, pollutants, viral); legal harvest (outside GLBA), illegal harvest (inside GLBA), incidental take/mortality as bycatch in fishing gear (entanglements, outside GLBA), vessel/human disturbance, climate change, predation, competition	Population trends - trend for SE AK from 1979-2005 as calculated by Pitcher et al. (2007) as an annual increase of 3.2%. Spatial and seasonal distribution at haulourts/rookeries - distribution when survey efforts began in park (mid 1970s). Survival rates - mean age-specific survival rates from 4 SE AK rookeries (Hastings et al. 2011).
Fishes				
	Anadromous Fish	Distribution (3), Removals (3)	Harvest, PDO, ocean acidification, spawning/rearing habitat change (uplift, hydrological change, vegetation community change), climate change & invasive species. Limited commercial fishery may be inordinately dependent on small but increasing proportional contribution of Doame River fish (1,000-5,000 fish/year escapement). Concern for presumed incidental harvest of other smaller, weaker stocks (Dog Salmon & Hazel Creeks).	Reference condition currently unknown
	Mid-trophic level marine forage community	Species specific spatial distribution of forage species (3), density (3), biomass (3), depth (3), change in species composition over time (3)	Changes in ocean climate or regime shifts, PDO, ocean acidification, increased predation, competition, disease (has regulated some herring populations in AK), harvest outside of GLBA (herring), acoustic/underwater noise. Availability of spawning habitat.	Reference condition currently unknown due to limited data

Figure 6 (continued). Glacier Bay National Park natural resource condition assessment framework.

GLBA Framework 1-5-2016
Natural Resource Condition Assessment Framework

	Component	Measures (Significance Level)/Analysis Project	Stressors	Reference Condition
Ecosystem Extent and Function				
Fishes				
	Pacific Halibut	Spawning stock biomass (3), removals (2), size at age (3)	Harvest, PDO, climate change, ocean acidification, competition (arrow tooth flounder), incidence of physiologically stressed condition (chalky & mushy flesh texture), pollutants/contaminants (Hg), parasites (Ichthyophonus).	Spawning stock biomass - no reference condition; Removals - total constant explotation yield for a particular calendar year (varies yearly); Size-at-age - no reference condition
Plant Communities				
	Plants	Species richness (3), number of invasive species (3), distribution of invasive species (3), spruce mortality distribution (2), yellow-cedar distribution (1), yellow-cedar density (1), yellow-cedar age structure (1), density in shore pine stands (1)	Climate change (cedar, driver or fungal and disease in park), vectors (invasive plants),	Species richness - NPS Certified Species List; Non-native species measures - no invasive species present; Spruce mortality - no mortality above 70% (highest observed in 1980s outbreak); Yellow-cedar measures - determined by NPS 2013 and Oakes et al. 2014; Density in shore pine stands - no reference condition
Herptiles				
	Western Toad	Abundance (for all life stages) (3), distribution (for all life stages) (3),	Chitrid fungus, climate change, contaminants, increased presence of vectors (disease), isostatic uplift,	No reference condition - anecdotal accounts of distribution and abundance can be consulted
Environmental Quality				
	Air Quality	Lichen sampling (2), Atmospheric deposition of sulfur and nitrogen (2), mercury deposition (2), particulate matter (2), visibility (2)	Industrial activity, local point sources of contaminants (boats, Gustavus?)	NPS ARD standards in accordance with NAAQS, ecosystem thresholds, and visibility improvement goals. Lichen reference condition - Dillman et al. (2007)
	Water Quality	**Freshwater contaminants:** Hg (3), MeHg (3), POPs (3)	Anthropogenic point source pollution. Natural processes. Bioaccumulation	AK DEC water quality standards (AK DEC 2008)
		Marine Contaminants: Metals (3), POP (3), PAH (3). Ocean Acidification (3)	Point source pollution, Bioaccumulation, Climate change (acidification)	AK DEC water quality standards (AK DEC 2008); Mussel sampling will use Kimbrough et al. (2008); Fish sampling - Eagles-Smith et al. (2014) and Beckvar et al. (2005)

Figure 6 (continued). Glacier Bay National Park natural resource condition assessment framework.

GLBA Framework 1-5-2016
Natural Resource Condition Assessment Framework

	Component	Measures (Significance Level)/Analysis Project	Stressors	Reference Condition
Ecosystem Extent and Function				
Environmental Quality				
	Underwater Soundscape	Underwater natural ambient sound level (dB) at frequencies of interest (3); Proportion of hourly samples without vessel noise (3); Duration of noise-free intervals (3); Masking Index of communication space for vocal species like humpbacks and harbor seals (2).	Vessel noise inside the park; vessel noise coming in from outside the park.	Historic condition of the area prior to park establishment and vessel traffic.
Physical Characteristics				
Geologic & Hydrologic				
	Wilderness*	**Solitude or primitive and unconfined recreation** (#, location, and condition of backcountry campsites [3]; # and distribution of backcountry users [3]; location and extent of facilities that decrease self reliant recreation [1]; extent and magnitude of intrusions on the natural soundscape [3])	Changes in visitor use; changes in drop-off locations; vessel noise inside the park; flightseeing; trail building; recreation facility development in wilderness; current/ongoing visitor use/vessel activities.	Wilderness provides outstanding opportunities for solitude or primitive and unconfined recreation
		Undeveloped (index [#] and location of authorized physical developments in wilderness [GIS layer] [3]; amount and location of administrative [or admin istrative approved] helicopter landings [3]; amount and location of administrative [or administrative approved] vessel use in Glacier Bay Proper [2]; cruise ship entries [2]; type and amount and location of authorized motor vehicle, motor equipment or transport vehicle use [2])	Facility development, helicopter and administrative vessel use; research and communication installations;	Wilderness retains its primeval character and influence, and is essentially without permanent improvement or modern human occupation
		Untrammeled (# and location of authorized and non authorized actions to manage plants, animals, pathogens, soil, water, or fire)	Spread of invasive weeds; collaring or marking animals; fire suppression activities; adaptive management strategies for climate change.	Wilderness is essentially unhindered and free from actions of modern human control or manipulation

Figure 6 (continued). Glacier Bay National Park natural resource condition assessment framework. *Indicates that the wilderness component was modified both in metrics and in methodology following the finalization of the framework. The full extent of these modifications are described in Chapter 4.24.

3.2.2. General Approach and Methods

This study involved gathering and reviewing existing literature and data relevant to each of the key resource components included in the framework. No new data were collected for this study; however, where appropriate, existing data were further analyzed to provide summaries of resource condition or to create new spatial representations. After all data and literature relevant to the measures of each component were reviewed and considered, a qualitative statement of overall current condition was created and compared to the reference condition when possible.

Data Mining

The data mining process (acquiring as much relevant data about key resources as possible) began at the initial scoping meeting, at which time GLBA staff provided data and literature in multiple forms, including: NPS reports and monitoring plans, reports from various state and federal agencies, published and unpublished research documents, databases, tabular data, and charts. GIS data were also provided by NPS staff. Additional data and literature were acquired through online bibliographic literature searches and inquiries on various state and federal government websites. Data and literature acquired throughout the data mining process were inventoried and analyzed for thoroughness, relevancy, and quality regarding the resource components identified at the scoping meeting.

Data Development and Analysis

Data development and analysis was highly specific to each component in the framework and depended largely on the amount of information and data available for the component, as well as recommendations from NPS reviewers and sources of expertise including NPS staff from GLBA and the SEAN. Specific approaches to data development and analysis can be found within the respective component assessment sections located in Chapter 4 of this report.

Scoring Methods and Assigning Condition

Significance Level

A set of measures are useful in describing the condition of a particular component, but all measures may not be equally important. A "*Significance Level*" represents a numeric categorization (integer scale from 1-3) of the importance of each measure in assessing the component's condition; each *Significance Level* is defined in Table 4. This categorization allows measures that are more important for determining condition of a component (higher *Significance Level*) to be more heavily weighted in calculating an overall condition. *Significance Level*s were determined for each component measure in this assessment through discussions with park staff and/or outside resource experts.

Table 4. Scale for a measure's *Significance Level* in determining a components overall condition.

Significance Level (SL)	Description
1	Measure is of low importance in defining the condition of this component.
2	Measure is of moderate importance in defining the condition of this component.
3	Measure is of high importance in defining the condition of this component.

Condition Level

After each component assessment is completed (including any possible data analysis), SMUMN GSS analysts assign a *Condition Level* for each measure on a 0-3 integer scale (Table 5). This is based on all the available literature and data reviewed for the component, as well as communications with park and outside experts.

Table 5. Scale for *Condition Level* of individual measures.

Condition Level (CL)	Description
0	Of NO concern. No net loss, degradation, negative change, or alteration.
1	Of LOW concern. Signs of limited and isolated degradation of the component.
2	Of MODERATE concern. Pronounced signs of widespread and uncontrolled degradation.
3	Of HIGH concern. Nearing catastrophic, complete, and irreparable degradation of the component.

Weighted Condition Score

After the *Significance Levels* (SL) and *Condition Levels* (CL) are assigned, a Weighted Condition Score (WCS) is calculated via the following equation:

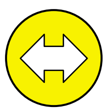
$$WCS = \frac{\sum_{i=1}^{\#\,of\,measures} SL_i * CL_i}{3 * \sum_{i=1}^{\#\,of\,measures} SL_i}$$

The resulting WCS value is placed into one of three possible categories: good condition (WCS = 0.0 – 0.33); condition of moderate concern (WCS = 0.34-0.66); and condition of significant concern (WCS = 0.67 to 1.0). Figure 7 displays all of the potential graphics used to represent a component's condition in this assessment. The colored circles represent the categorized WCS; red circles signify a significant concern, yellow circles a moderate concern and green circles that a resource is in good condition. White circles are used to represent situations in which SMUMN GSS analysts and park staff felt there were currently insufficient data to make a statement about the condition of a component. For example, condition is not assessed when no recent data or information are available, as the purpose of an NRCA is to provide a "snapshot-in-time" of current resource conditions. The arrows inside the circles indicate the trend of the condition of a resource component, based on data and literature from the past 5-10 years, as well as expert opinion. An upward pointing arrow indicates the condition of the component has been improving in recent times. A horizontal arrow indicates an unchanging condition or trend, and an arrow pointing down indicates deterioration in the condition of a component in recent times. These are only used when it is appropriate to comment on the trend of condition of a component. In situations where the trend of the component's condition is currently unknown, no arrow is given.

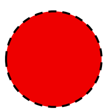
Condition Status		Trend in Condition		Confidence in Assessment	
	Resource is in Good Condition		Condition is Improving		High
	Resource warrants Moderate Concern		Condition is Unchanging		Medium
	Resource warrants Significant Concern		Condition is Deteriorating		Low

Examples of how the symbols should be interpreted

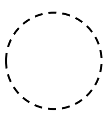
Resource is in good condition; its condition is improving; high confidence in the assessment.



Condition of resource warrants moderate concern; condition is unchanging; medium confidence in the assessment.



Condition of resource warrants significant concern; trend in condition is unknown or not applicable; low confidence in the assessment.



Current condition is unknown or indeterminate due to inadequate data, lack of reference value(s) for comparative purposes, and/or insufficient expert knowledge to reach a more specific condition determination; trend in condition is unknown or not applicable; low confidence in the assessment.

Preparation and Review of Component Draft Assessments

The preparation of draft assessments for each component was a highly cooperative process among SMUMN GSS analysts and GLBA and SEAN staff. Though SMUMN GSS analysts rely heavily on peer-reviewed literature and existing data in conducting the assessment, the expertise of NPS resource staff also plays a significant and invaluable role in providing insights into the appropriate direction for analysis and assessment of each component. This step is especially important when data or literature are limited for a resource component.

The process of developing draft documents for each component began with a detailed phone or e-mail conversation with an individual or multiple individuals considered local experts on the resource components under examination. These conversations were a way for analysts to verify the most relevant data and literature sources that should be used and also to formulate ideas about current condition with respect to the NPS staff opinions. Upon completion, draft assessments were forwarded to component experts for initial review and comments.

Development and Review of Final Component Assessments
Following review of the component draft assessments, analysts used the review feedback from resource experts to compile the final component assessments. As a result of this process, and based on the recommendations and insights provided by GLBA resource staff and other experts, the final component assessments represent the most relevant and current data available for each component and the sentiments of park resource staff and outside resource experts.

Format of Component Assessment Documents
All resource component assessments are presented in a standard format. The format and structure of these assessments is described below.

Description
This section describes the relevance of the resource component to the park and the context within which it occurs in the park setting. For example, a component may represent a unique feature of the park, it may be a key process or resource in park ecology, or it may be a resource that is of high management priority. Also emphasized are interrelationships that occur among the featured component and other resource components included in the NRCA.

Measures
Resource component measures were defined in the scoping process and refined through dialogue with resource experts. Those measures deemed most appropriate for assessing the current condition of a component are listed in this section, typically as bulleted items.

Reference Conditions/Values
This section explains the reference condition determined for each resource component as it is defined in the framework. Explanation is provided as to why specific reference conditions are appropriate or logical to use. Also included in this section is a discussion of any available data and literature that explain and elaborate on the designated reference conditions. If these conditions or values originated with the NPS experts or SMUMN GSS analysts, an explanation of how they were developed is provided.

Data and Methods
This section includes a discussion of the data sets used to evaluate the component and if or how these data sets were adjusted or processed as a lead-up to analysis. If adjustment or processing of data involved an extensive or highly technical process, these descriptions are included in an appendix for the reader or a GIS metadata file. Also discussed is how the data were evaluated and analyzed to determine current condition (and trend when appropriate).

Current Condition and Trend
This section presents and discusses in-depth key findings regarding the current condition of the resource component and trends (when available). The information is presented primarily with text but is often accompanied by detailed maps or plates that display different analyses, as well as graphs, charts, and/or tables that summarize relevant data or show interesting relationships. All relevant data and information for a component is presented and interpreted in this section.

Threats and Stressor Factors

This section provides a summary of the threats and stressors that may impact the resource and influence to varying degrees the current condition of a resource component. Relevant stressors were described in the scoping process and are outlined in the NRCA framework. However, these are elaborated on in this section to create a summary of threats and stressors based on a combination of available data and literature, and discussions with resource experts and NPS natural resources staff.

Data Needs/Gaps

This section outlines critical data needs or gaps for the resource component. Specifically, what is discussed is how these data needs/gaps, if addressed, would provide further insight in determining the current condition or trend of a given component in future assessments. In some cases, the data needs/gaps are significant enough to make it inappropriate or impossible to determine condition of the resource component. In these cases, stating the data needs/gaps is useful to natural resources staff seeking to prioritize monitoring or data gathering efforts.

Overall Condition

This section provides a qualitative summary statement of the current condition that was determined for the resource component using the WCS method. Condition is determined after thoughtful review of available literature, data, and any insights from NPS staff and experts, which are presented in the Current Condition and Trend section. The Overall Condition section summarizes the key findings and highlights the key elements used in determining and justifying the level of concern, if any, that analysts attribute to the condition of the resource component. Also included in this section are the graphics used to represent the component condition.

Sources of Expertise

This is a listing of the individuals (including their title and affiliation with offices or programs) who had a primary role in providing expertise, insight, and interpretation to determine current condition (and trend when appropriate) for each resource component.

Literature Cited

This is a list of formal citations for literature or datasets used in the analysis and assessment of condition for the resource component. Note, citations used in appendices referenced in each section (component) of Chapter 4 are listed in that component's "Literature Cited" section.

3.3. Literature Cited

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Chapter 4. Natural Resource Conditions

This chapter presents the background, analysis, and condition summaries for the 20 key resource components in the project framework. The following sections discuss the key resources and their measures, stressors, and reference conditions. The summary for each component is arranged around the following sections:

1. Description
2. Measures
3. Reference Condition
4. Data and Methods
5. Current Condition and Trend (including threats and stressor factors, data needs/gaps, and overall condition)
6. Sources of Expertise
7. Literature Cited

The order of components follows the project framework (Figure 6):

4.1 Glaciers
4.2 Marine Shoreline
4.3 Marine Invertebrate Community
4.4 Murrelets (Kittlitz's and Marbled)
4.5 Sea Ducks
4.6 Breeding Landbirds
4.7 Glaucous-winged Gulls
4.8 Moose
4.9 Bears
4.10 Mountain Goat
4.11 Harbor Seals
4.12 Humpback Whales
4.13 Sea Otters
4.14 Harbor Porpoise
4.15 Steller Sea Lions
4.16 Anadromous Fish
4.17 Mid-trophic Level Marine Forage Community
4.18 Pacific Halibut
4.19 Western Toad
4.20 Air Quality
4.21 Water Quality
4.22 Underwater Soundscape
4.23 Wilderness

4.1. Glaciers

4.1.1. Description

Glaciers are large persistent bodies of ice that flow under the influence of gravity (Marshak 2005). The formation of a glacier requires three conditions: abundant snowfall, cool summer temperatures, and the gravitational flow of ice (NPS 2014). In Alaska, nine out of 15 NPS units have glaciers; GLBA is considered to have significant glacial coverage. According to Loso et al. (2014), glacial ice covered 5,323 km^2 (2,055 mi^2) of GLBA in 2010, forming an estimated 40.1% of the total park area. There are two types of glaciers in GLBA: tidewater and terrestrial glaciers. Tidewater glaciers are those that terminate in water, whereas terrestrial glaciers terminate on land. Photo 4 displays John Hopkins Glacier, a tidewater glacier in northern GLBA (Lawson 2004).

Photo 4. Johns Hopkins Glacier (NPS photo).

Glaciation begins with the accumulation of fresh, loosely packed snow containing 90% air, due to the space created by its hexagonal crystals (Marshak 2005). As new layers of snow accumulate on top of the old snow, pressure increases from the weight, squeezing out air pockets and, over time, transforming the snow into a packed granular material called firn, which contains only 25% air (Marshak 2005). As melting occurs, water recrystallizes in the spaces between grains until the firn is transformed into a solid mass of glacial ice containing only 20% air (Marshak 2005).

Glacier mass balance studies determine the annual difference between accumulation and ablation (all processes that remove mass, i.e., sublimation, melting, and calving) (Cogley et al. 2011, NPS 2014). Mass balance is commonly reported using the water year, following the assumption that accumulation begins, and ablation ends, on 1 October. In reality, the end of the mass balance year is elevation-dependent; it may occur earlier at higher elevations and later at lower elevations (Mayo et al. 1972).

If the rate of accumulation is higher than that of ablation, the glacier will thicken or advance. However, if the rate of ablation is higher than that of accumulation, the glacier will thin, retreat, or both (Marshak 2005). The accumulation zone is the area on a glacier where more mass is gained than lost, whereas the area where more mass is lost than gained is known as the ablation zone (Cogley et al. 2011). The accumulation area ratio (AAR) represents the ratio of the accumulation zone to the area of the glacier at the end of a mass balance year (Cogley et al. 2011). Mass balance studies can provide information on the stability of glaciers, runoff predictions, and a measurement of climatic variation and trends (Muirhead 1978).

Glacier snow lines define the boundary between the melting ablation zone and the snow-covered accumulation zone. Late summer is the end of the ablation season, and during this time, the late summer snow line reaches its highest elevation, called the annual snow line. The annual snow line is closely related to the equilibrium line, which separates the accumulation zone from the ablation zone (Muirhead 1978). The equilibrium line altitude (ELA) is the spatially averaged altitude of the equilibrium line at the end of a mass balance year (Cogley et al. 2011). The position of the snow line varies depending on the season. During winter, snow covers the entire glacier. As spring thaw occurs, the snow line moves up the glacier. The amount of accumulation and the ablation rate together determine how far the snow line will move up the glacier before the cycle repeats (Muirhead 1978).

4.1.2. Measures

- Glacial extent
- Glacial surface elevation

4.1.3. Reference Conditions/Values

The reference conditions for glaciers within GLBA are the dimensions of individual glaciers, the number of total glaciers, and the overall glacial extent within GLBA in 1951 (map date).

4.1.4. Data and Methods

Loso et al. (2014) provides glacier numbers, glacial extent, and glacial surface elevation data for GLBA glaciers based on the interpolation of topographic maps between 1948 and 1987 with a median of 1951 (map date 1951), satellite imagery from 2010 (satellite date 2010), and laser altimetry data taken from May 1994 to August 2012. Map date and satellite date were defined by the median years of the topographic maps and satellite imagery sources utilized.

The USGS (2012) has also used repeat photography to document the change in Alaska's glaciers, including several within GLBA.

4.1.5. Current Condition and Trend

<u>Glacial Extent</u>

Loso et al. (2014) determined that glacial cover in all nine Alaska NPS parks analyzed in the study showed diminished glacial cover over the last half century. Glacial cover in GLBA determined in 1951 (map date) was 15% greater than the current extent values in 2010 (satellite date). Glacial extent in GLBA decreased by over 950 km^2, from 6,284 km^2 in 1951 to 5,323 km^2 in 2010 (Figure 8, Table 6, Loso et al. 2014). During this time period, the total number of glaciers wholly or partly within the park increased by 50%; however, the mean glacier size decreased by 43% (Table 6). The increase in glacier numbers is attributed to larger glaciers retreating and thinning to the point where they break into one or more separate glacial areas (Loso et al. 2014).

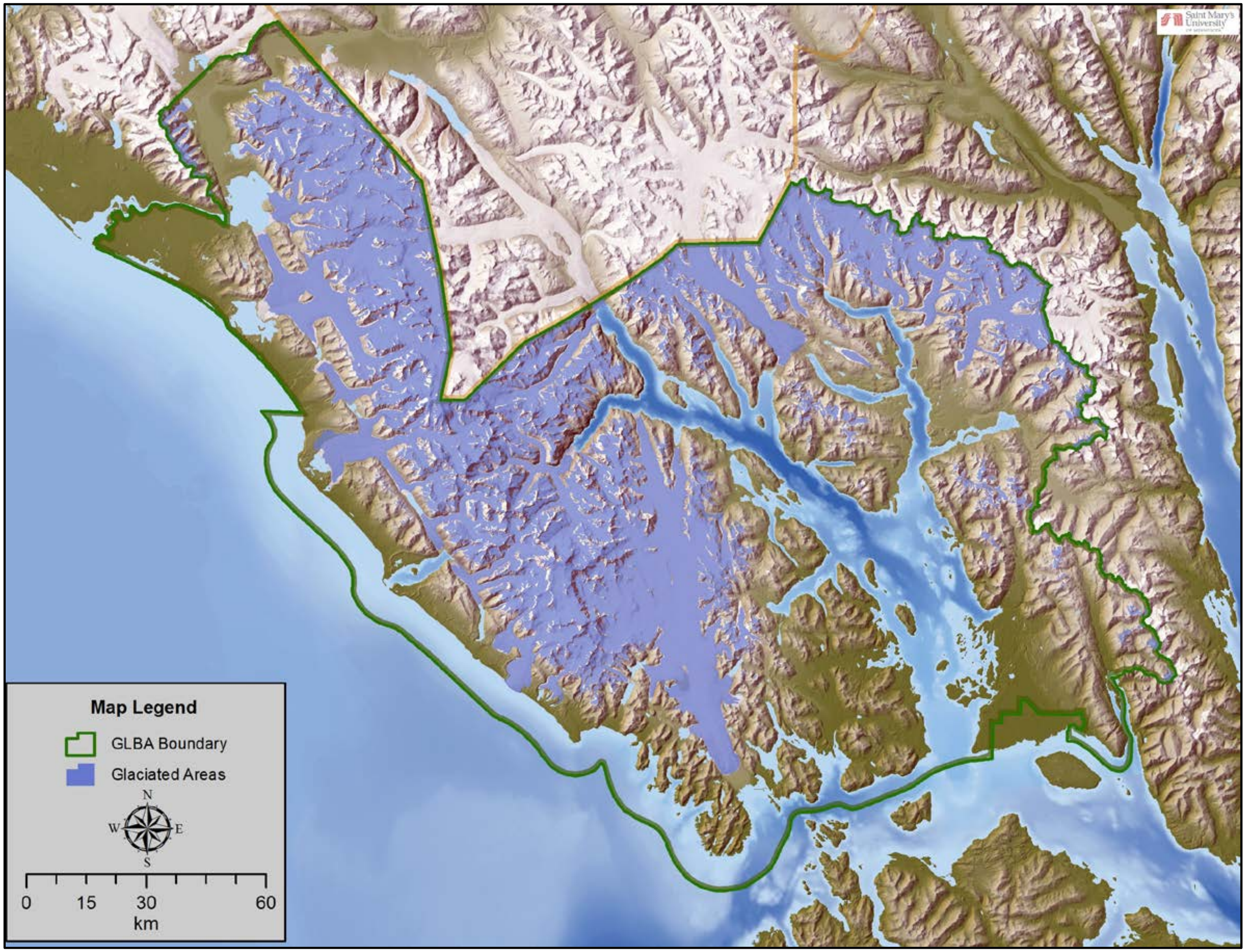

Figure 8. Satellite date (2010) glaciated extent within GLBA (Arendt and Rich 2011).

Table 6. Changes in glacier numbers, glacial size, and total glaciated area from map date (1951) to satellite date (2010) (Loso et al. 2014).

Time Period	Number of glaciers	Total glacier area (km^2)	Mean glacier size (km^2)	Max glacier size (km^2)
Map date (1951)	698	6,284.5	9	641
Satellite date (2010)	1,045	5,323.2	5.1	539
Absolute change	347	-961.3	-3.9	-102.1
Percent change	50	-15.0	-43	-16

Several glaciers have retreated at noticeable rates, including Grand Plateau, Desolation, Geikie, Casement, McBride, Burroughs, Plateau, and Muir Glaciers. However, some glaciers also expanded, including the Grand Pacific, Johns Hopkins, Lamplugh, Rendu, and North Crillon Glaciers (Loso et al. 2014). Figure 9 represents glacial ice coverage by elevation in GLBA; this analysis is considered the most reflective of the actual change in glaciers within the park across the two time periods represented. Glacial changes above 1,600 m (5,250 ft) are considered to be minimal, while changes below this threshold have been much more apparent when comparing map and satellite glacial extents. Actual glacial area change is most prominent between the elevations of 400 and 800 m (1,312 and 2,624 ft) (Figure 10, Loso et al. 2014).

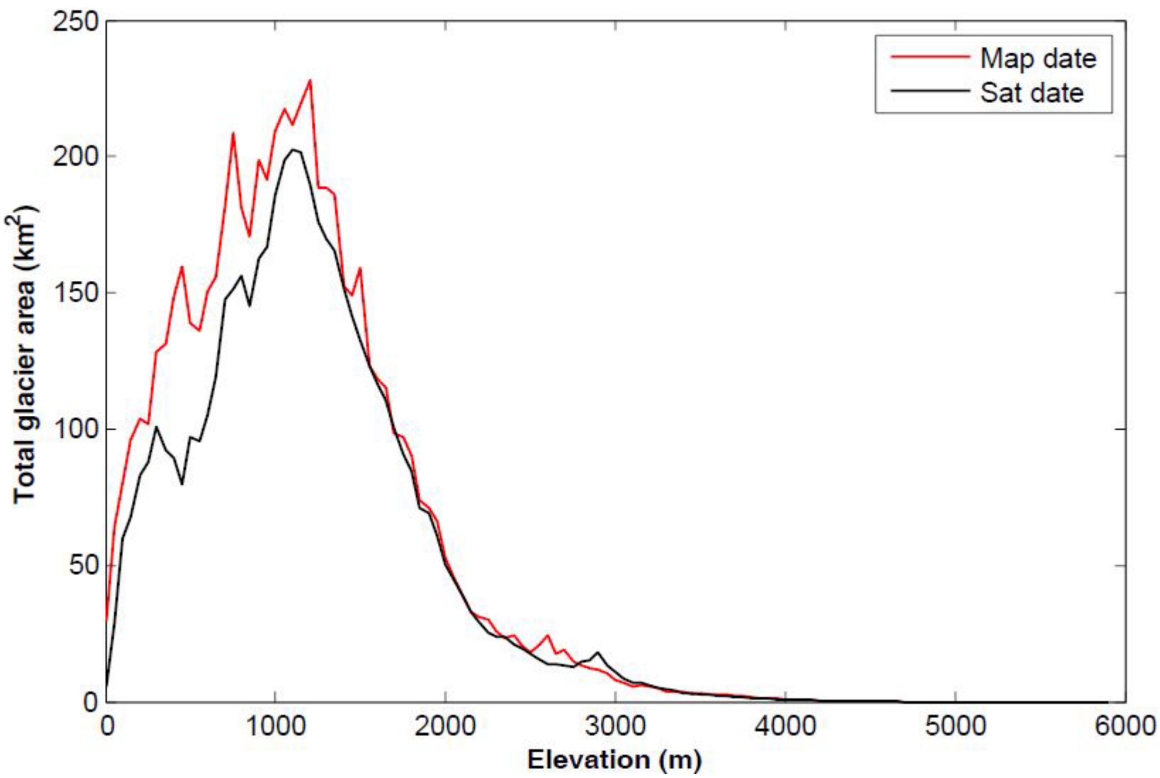

Figure 9. Total glacier-ice coverage by elevation in GLBA from map date (1951 - red line) to satellite date (2010 - black line) (Loso et al. 2014).

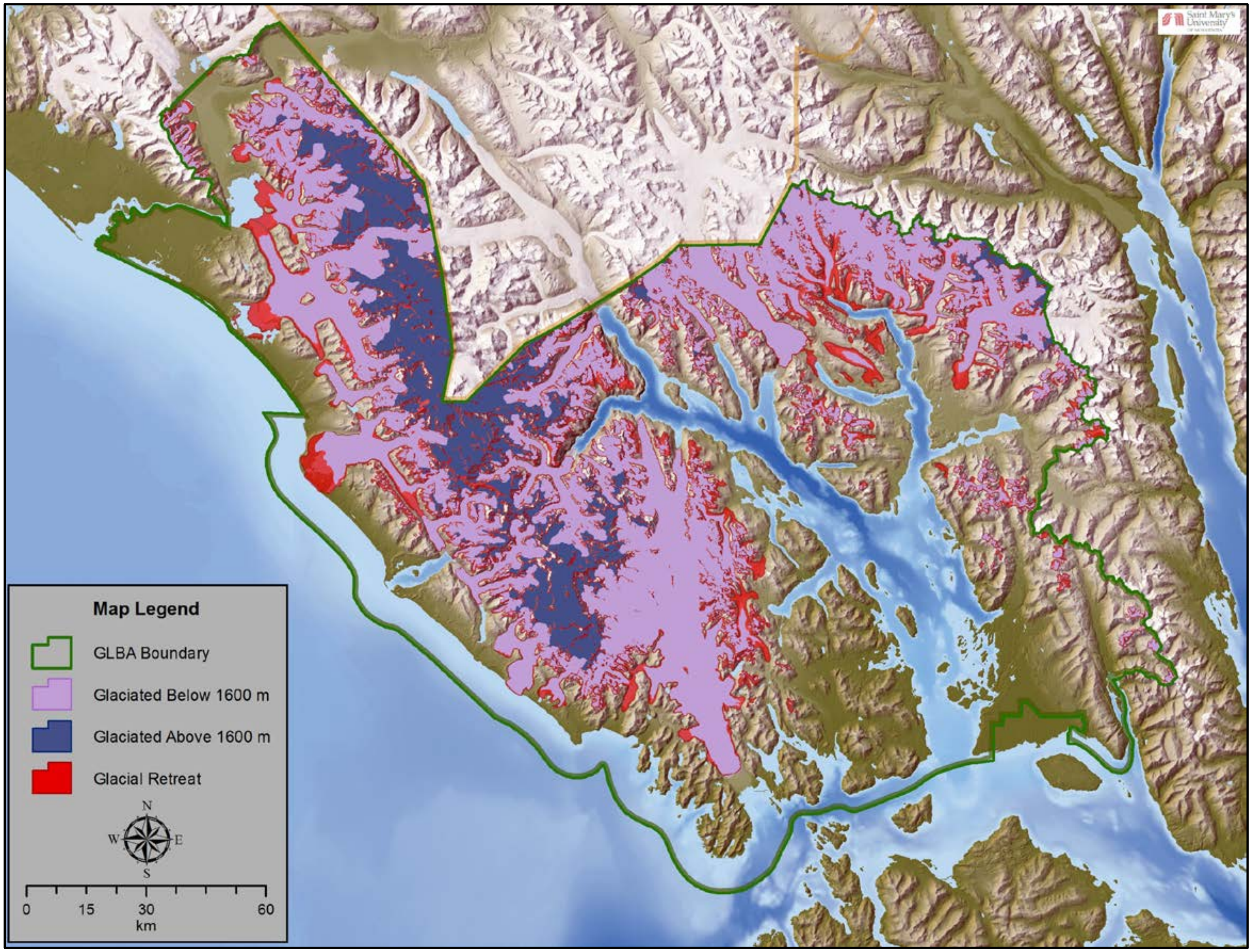

Figure 10. Extent of glacial ice in GLBA above and below 1,600 m (5,250 ft) and glacial recession areas (red) between map date (1951) and satellite date (2010). Glacial thinning and retreat was most prominent in glacial ice below 1,600 m; remaining glaciers below this elevation are at the greatest risk of extirpation (Loso et al. 2014).

Changes in glacial extent can also be documented through repeat photography. With this technique, historical and modern photos from the exact same location are compared to determine changes over time. The USGS has used repeat photography to explore glacial changes in several Alaska national parks, including GLBA (USGS 2012). The first images below are taken from the west shore of Muir Inlet looking northeast (Photo 5). Muir Glacier is visible in the 1892 photo but has retreated out of view by 2005.

Photo 5. View from the west shore of Muir Inlet looking northeast on 2 September 1892 (left, H. F. Reid photograph, courtesy of National Snow and Ice Data Center) and on 11 August 2005 (right, USGS photo by Bruce Molnia).

The next pair of images was taken from Muir Point looking north (Photo 6). Muir Glacier dominates the 1899 photo but is absent from the 2003 photo (USGS 2012).

Photo 6. View from Muir Point looking north in June 1899 (left, Grove Karl Gilbert photo from the USGS Photo Library) and in September 2003 (right, USGS photo by Bruce Molnia).

The final images are from several hundred meters up a side valley on the east side of Queen Inlet looking northwest (Photo 7). The 1906 photo shows Carroll Glacier terminating in Queen Inlet. By

2004, the glacier had receded and the inlet filled with sediment (USGS 2012). See USGS (2012) for more repeat photography of these and other GLBA glaciers.

Photo 7. View from the east side of Queen Inlet looking northwest in August 1906 (left, Charles Will Wright photo from the USGS Photo Library) and on 21 June 2004 (right, USGS photo by Bruce Molnia).

<u>Glacial Surface Elevation</u>

Loso et al. (2014) analyzed the surface elevation for glaciers in GLBA from 1995 to 2011. Glacial elevation readings are used to estimate volume and mass balance changes over time, measured in meters of water equivalent per year (m/yr w.e.) units. Glacial elevations showed declines at several glaciers in the park, particularly at lower elevations. Casement, Carroll, Grand Pacific, Grand Plateau, Konamoxt, and Tkope Glaciers have experienced declines near 6 m/yr since 2009 at elevations below 400 m in elevation (Loso et al. 2014). Surface elevations at Brady, Riggs, Davidson, Muir, and Melbern Glaciers have declined at a slower rate of 4 m/yr at elevations below 400 m (1,312 ft) during the same period. Reid, Lamplugh, and Margerie Glaciers have shown the smallest declines, with rates at or below 2 m/yr (Loso et al. 2014). Surface elevations actually increased in some areas of Margerie Glacier (Photo 8) from 2005-2011. Graphs of surface elevation change by glacier and time period can be found in Loso et al. (2014). Elevation changes for several glaciers are also depicted spatially in Appendix A.

Photo 8. Margerie Glacier, terminating just above Tarr Inlet (NPS photo).

Estimates of mass balance from surface elevation data indicate a thinning trend for 15 of the 16 glaciers studied within GLBA (Table 7). Generally, the thinning of glaciers in GLBA was between 0.1 and 1.5 m/yr w.e. (Loso et al. 2014). Only Margerie Glacier experienced a period of positive mass balance, gaining mass between 2009 and 2011 (Table 7).

Table 7. Glacial mass balance change estimates for GLBA (Loso et al. 2014).

Glacier	Type[A]	Start Date	End Date	Mass Balance (m/year)	MB+[B]	MB-[B]
Brady	L	6/4/1995	5/25/2000	-1.04	0.19	0.19
Brady	L	5/25/2000	6/1/2005	-1.84	0.26	0.24
Brady	L	6/1/2005	6/2/2009	-0.73	0.37	0.31
Brady	L	6/2/2009	5/30/2011	-1.35	0.23	0.24
Carroll	L/S	6/2/2009	5/30/2011	-0.68	0.37	0.28
Casement	L	6/1/2005	6/2/2009	-1.18	0.41	0.44
Casement	L	6/2/2009	5/30/2011	-1.46	0.42	0.51
Davidson	LK	6/1/2005	6/2/2009	-0.68	0.35	0.35
Davidson	LK	6/2/2009	5/30/2011	-1.18	0.28	0.26
Fairweather	L	6/2/2009	5/30/2011	-1.35	0.70	0.93
Grand Pacific	T	6/7/1996	6/6/2001	-0.50	0.70	0.70
Grand Pacific	T	6/6/2001	6/2/2009	-1.15	0.62	0.62
Grand Pacific	T	6/2/2009	5/29/2011	-1.58	0.44	0.64
Grand Plateau	LK	6/2/2005	6/2/2009	-1.02	0.46	0.51

[A] The “Type” field indicates the termination area of the glacier; if two types are listed this indicates a classification change during the study period, L= land terminating, LK = lake terminating, T = tidewater, and S = surge.

[B] “MB+” and “MB-“ columns give positive and negative 95% confidence intervals for mass balance values.

Table 7 (continued). Glacial mass balance change estimates for GLBA (Loso et al. 2014).

Glacier	Type[A]	Start Date	End Date	Mass Balance (m/year)	MB+[B]	MB-[B]
Grand Plateau	LK	6/2/2009	5/30/2011	-2.85	0.77	0.83
Konamoxt	LK	6/7/1996	5/30/2011	-1.28	0.62	0.65
Lamplugh	T	6/4/1995	5/25/2000	-0.32	0.31	0.30
Lamplugh	T	5/25/2000	6/1/2005	-0.54	0.44	0.42
Lamplugh	T	6/1/2005	6/2/2009	-0.10	0.50	0.50
Lamplugh	T	6/2/2009	5/30/2011	-0.10	0.40	0.40
Little Jarvis	L	5/31/1995	5/28/2000	-0.43	0.34	0.26
Margerie	T/S	6/2/2005	6/2/2009	0.00	0.60	0.20
Margerie	T/S	6/2/2009	5/30/2011	0.40	0.80	0.80
Melbern	LK	6/2/2009	5/29/2011	-0.80	1.00	1.00
Muir	L	5/28/2000	6/1/2005	-0.50	0.50	0.60
Muir	L	6/1/2005	6/2/2009	0.0	0.40	0.40
Muir	L	6/2/2009	5/30/2011	-0.10	0.30	0.40
Reid	T	6/4/1995	5/25/2000	-0.31	0.27	0.30
Reid	T	5/25/2000	6/1/2005	-0.93	0.23	0.22
Reid	T	6/1/2005	6/2/2009	-0.10	0.30	0.30
Reid	T	6/2/2009	5/30/2011	-0.20	0.40	0.50
Riggs	L	6/1/2005	6/2/2009	-0.41	0.29	0.29
Riggs	L	6/2/2009	5/30/2011	-0.93	0.32	0.31
Tkope	L	6/2/2009	5/30/2011	-0.10	0.4	0.40

[A] The "Type" field indicates the termination area of the glacier; if two types are listed this indicates a classification change during the study period, L= land terminating, LK = lake terminating, T = tidewater, and S = surge.

[B] "MB+" and "MB-" columns give positive and negative 95% confidence intervals for mass balance values.

Threats and Stressor Factors

Climate Change

Climate is one of the most important factors influencing ecosystems. In Alaska, climate is constantly fluctuating on multiple temporal scales, including several natural cycles. One climate fluctuation of particular importance in Alaska is the PDO (Lindsay 2011). Mantua et al. (1997) formally identified this pattern of climate variability in a study relating climate oscillation to salmon production. The PDO, which is related to sea surface temperatures in the northern Pacific Ocean, affects atmospheric circulation patterns and alternates between positive and negative phases (Wendler and Shulski 2009). A positive phase is associated with a relatively strong low pressure center over the Aleutian Islands, which moves warmer air into the state, particularly during the winter (Wendler and Shulski 2009). Some of the variation in Alaska's climate over time can be explained by major shifts in the PDO which occurred in 1925 (negative to positive), 1947 (positive to negative), and 1977 (negative to positive) (Mantua et al. 1997). Hartmann and Wendler (2005) found that much of the warming that occurred in Alaska during the last half of the twentieth century was likely due to the PDO shift in 1976-77.

There is a scientific consensus that human activities, particularly those that produce greenhouse gasses, have contributed to a general warming trend in global climate (IPCC 2007). Current warming has accelerated natural processes that release greenhouse gases into the atmosphere, such as permafrost thawing and ebullition (methane bubbling) from northern lakes, further contributing to global warming (Anisimov 2007, Walter et al. 2007).

Weather and climate data are extremely sparse within GLBA; only four weather and climate stations are currently in place in GLBA, with two of these just installed in 2015. The full effects of climate change remain somewhat speculative and a point of active research in the SEAN (Davey et al. 2007), with eight more stations planned for installation in the next few years.

The Scenarios Network for Alaska Planning (SNAP et al. 2009) provides some projected estimates regarding changes in temperature through 2080. Currently, the average annual temperature within GLBA is 1.9°C (35.4°F); this average is expected to rise to 5.3°C (41.5°F) by 2080, an increase of about 0.6°C (1°F) per decade. Mean winter temperatures could rise more dramatically, by as much as 3.9°C (7°F) by 2080. These predicted temperature increases would exacerbate glacial thinning and retreat within GLBA and possibly lead to the extirpation of several glaciers within the park (SNAP et al. 2009).

Data Needs/Gaps

The continuation of laser altimetry analysis to determine the mean elevation of the 16 target glaciers within GLBA will be very import in understanding the rate of glacial thinning or advancement. The most recent mean elevation and mass balance data from 2011 show a thinning trend in GLBA glaciers. Monitoring of the resource in this manner should be continued as glaciers are considered to be of high concern, particularly given the threat of climate change.

The lack of consistent monitoring from weather stations in GLBA prevents the opportunity to correlate glacial changes from the 1950s to 2010 with precipitation fluctuations, temperature changes, etc. (Davey et al. 2007).

Overall Condition

Glacial Extent

The project team assigned this measure a *Significance Level* of 3. Glacial extent within GLBA was determined to have decreased by 15% between the early 1950s and 2010 (Loso et al. 2014). Although an increased number of glaciers were observed over this time period, the increase was from large pre-existing glaciers splitting into distinct, smaller glaciers as a result of glacial retreat and thinning over the past 60 years (Loso et al. 2014). Therefore, this measure received a *Condition Level* of 3, indicating high concern.

Glacial Surface Elevation

This measure was assigned a *Significance Level* of 2. All of the 16 glaciers within GLBA that underwent surface elevation analysis from 1995 to 2011 showed declines (i.e., thinning) over that time period (Loso et al. 2014). Mass balance estimates based on surface elevation changes showed that thinning generally ranged from 0.1 to 1.5 m/yr w.e. Only Margerie Glacier experienced increases in surface elevations in some areas which offset declines in other areas, contributing to a positive mass balance between 2009 and 2011 (Loso et al. 2014). Surface elevation is currently considered of moderate concern (*Condition Level* = 2).

<u>Weighted Condition Score</u>

The *Weighted Condition Score* for glaciers in GLBA is 0.87. This WCS represents an overall condition of high concern. Given recent studies (Loso et al. 2014), the trend of this resource's condition appears to be deteriorating.

Glaciers			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.87**
Glacial Extent	3	3	
Glacial Surface Elevation	2	2	

4.1.6. Sources of Expertise

- Guy Adema, NPS Alaska Region Natural Resources Team Manager

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4.2. Marine Shoreline

4.2.1. Description

Marine shorelines are a natural border between land and marine ecosystems. Shorelines can be divided into two zones: the intertidal zone, which is exposed at low tide and submerged under water at high tide, and the supratidal zone, which is exposed during both high and low tides but is submerged occasionally during storms. GLBA contains nearly 1,900 km (1,181 mi) of marine shoreline within its boundaries (Sharman et al. 2007; Photo 9). These areas are among the most productive and diverse habitats of the park, and are also a focus for much of the human activity (e.g., kayaking, camping) in GLBA (Sharman et al. 2007). Marine shorelines provide important foraging habitat, denning locations, and travel corridors for terrestrial wildlife in the park (Lewis and Drumheller 2004). Shorelines also serve as haul-out sites for harbor seals and Steller sea lions (*Eumetopias jubatus*), particularly during the pupping and molting seasons. Additionally, they provide nesting areas for a variety of birds, and resting areas for molting sea ducks (Lewis and Drumheller 2004).

Photo 9. The GLBA marine shoreline at Bartlett Cove (NPS photo).

Marine shorelines are vulnerable to the accumulation of debris and contaminants carried by adjacent bodies of water. Marine debris is defined by the National Oceanic and Atmospheric Administration (NOAA) as “any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the marine environment” (NOAA 2015). Marine debris can include plastics, metals, styrofoam, rubber, lost or abandoned fishing gear, and wrecked vessels (EPA 2011). This debris degrades ocean and shoreline habitats, endangers marine and coastal wildlife, threatens human health and safety, and can interfere with navigation (EPA 2011). Even the largely pristine beaches of Alaska have been impacted by marine debris for decades (AK DEC 2013). Entanglement in or ingestion of marine debris is a particular threat to marine mammals, seabirds, and shorebirds (Laist 1997, Raum-Suryan et al. 2009). Marine debris, especially plastics and microplastics, can carry contaminants or produce POPs as it breaks down in the environment (Frias et al. 2010, Rios et al. 2010, EPA 2011). Other sources of contaminants include fuel and chemical spills from vessel and terrestrial sources, as well as wastewater and ballast-water discharge from vessels (NPS 1998). Contaminants can cause mortality or injury to wildlife and can reduce reproductive success. Impacts may be due to direct exposure to contaminants or indirectly through the consumption of contaminated prey (NPS 1998).

4.2.2. Measures

- Abundance and distribution of anthropogenic marine debris in inter- and supratidal zones (including vessel wrecks)
- Abundance and distribution of stranded contaminants (e.g., spills, plastic microparticles)
- Camper and other user biophysical impacts

4.2.3. Reference Conditions/Values

The reference condition for GLBA's marine shoreline is pristine or near-pristine conditions (e.g., free of anthropogenic debris and contaminants). Camper/user biophysical impacts should not exceed park-designated limits; however, these limits have not yet been determined.

4.2.4. Data and Methods

From 1997-2003, the GLBA Coastal Resources Inventory and Mapping Program surveyed over 1,500 km (932 mi) of coastline in GLBA (Sharman et al. 2007). These efforts resulted in the collection of over 21,000 ground photos (Photo 10), approximately 300 high-resolution georeferenced aerial photos, and coarse-scale descriptions of a variety of coastal biophysical resources characterizing more than 6,000 discrete shoreline segment polygons. During field mapping, teams of two walked coastal segments during low-tide windows and gathered information on physical and biological resources including intertidal community composition, adjacent upland vegetation type, and the presence of special features such as streams, tide pools, interstadial wood, intertidal reefs, flotsam collection areas, and seabird colonies (Sharman et al. 2007). These data were organized into an electronic database containing photos, distribution maps for marine intertidal organisms, and a variety of other ecological and ethnological information. This coastal resources database provides a baseline that will allow for the detection and documentation of future environmental change and could aid managers in responding to human-caused disturbances (e.g., oil spills or other contaminant exposures) (Sharman et al. 2007).

Photo 10. A ground photo taken during the GLBA coastline mapping efforts (NPS photo).

Marine Debris

The Alaska Department of Environmental Conservation (AK DEC) conducted an aerial survey of the southern Alaska coast in 2012 to collect imagery and data needed to assess marine debris density and composition (AK DEC 2013). The survey area extended from Dixon Entrance at Alaska's southern border to Hallo Bay in the western Gulf of Alaska, within Katmai National Park (Figure 11). The survey yielded over 8,000 high-resolution georeferenced aerial photos; each image was assigned a qualitative debris rating from 0 (no debris) to 5 (heaviest debris density) (AK DEC 2013). The survey was triggered by an apparent increase in marine debris on Alaskan beaches following the March 2011 earthquake and tsunami in Tohoku, Japan (AK DEC 2013). Tsunami debris, such as buoys/floats from oyster farms and styrofoam pieces, began arriving on Alaska's coasts by December 2011 and was expected to continue washing up for several years (AK DEC 2013).

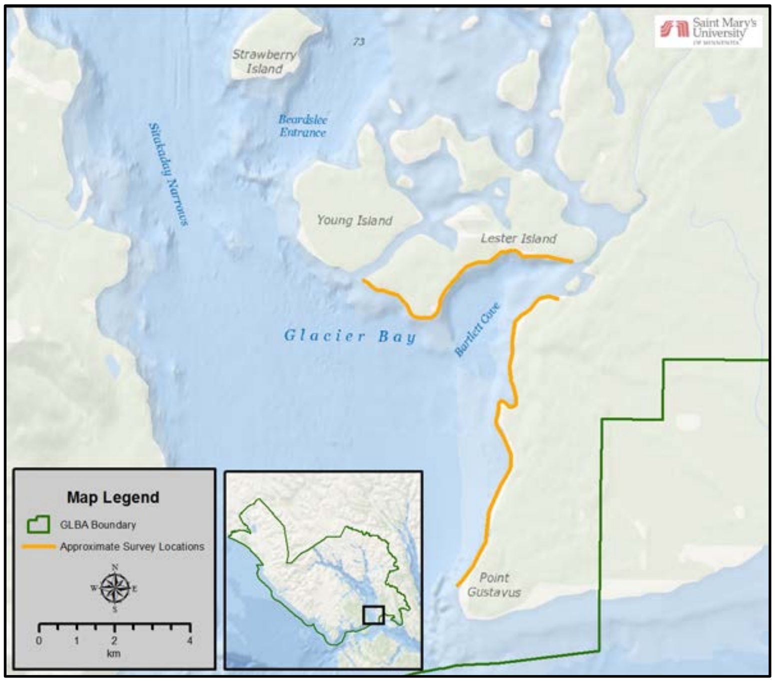

Figure 11. Approximate locations of early marine debris survey locations in the Bartlett Cove area. The earliest studies of marine debris on GLBA's shorelines were conducted by Rettew (1991) and Polasky (1992). These efforts surveyed two stretches of shoreline near Bartlett Cove: a 9.7 km (6 mi) stretch on the southern coast of Lester Island, and an 11.3 km (7 mi) stretch from Point Gustavus to the outer dock in Bartlett Cove (Rettew 1991; Figure 11). The two shorelines were surveyed, and debris was removed twice during the summers of 1991 and 1992. The surveys were intended to assess the types and amounts of marine debris on these beaches, as well as relative accumulation rates (Polasky 1992).

Contaminants

The SEAN has established a monitoring protocol for marine contaminants utilizing tissue samples from the bay mussel (*Mytilus trossulus*) and marine sediment (Tallmon 2012). Mussels were chosen for sampling because they are quick and cheap to obtain, allowing for large sample sizes. As filter feeders, any contaminants present in the surrounding waters are incorporated into the mussels' tissues. Mussels are also long-lived and generally stay in one place, meaning they can provide insight for both long-term and recent contamination specific to the area where they are found (Tallmon 2012). An additional benefit of using mussels is that results can be compared to data from the long-standing Mussel Watch Program, which has monitored over 100 contaminants at over 300 sites since 1986 (Tallmon 2012, NOS 2014). The initial study in July and August of 2007 collected mussel samples from random locations throughout GLBA (Figure 12), as well as non-random, potentially contaminated sites (called "hot" controls) including the Bartlett Cove fueling dock and boat ramp (Tallmon 2012). Sediment samples were also collected and analyzed from five sites in the park. Mussels were resampled at four of the initial GLBA study sites (Bartlett Cove, Ripple Cove, W. Hazelton Camp, and E. Russell Rocks) during August and September of 2009 and 2011 (Tallmon 2012). Contaminants tested for included metals (e.g., arsenic, mercury), polycyclic aromatic hydrocarbons (PAHs), and POPs (e.g., pesticides, industrial chemicals).

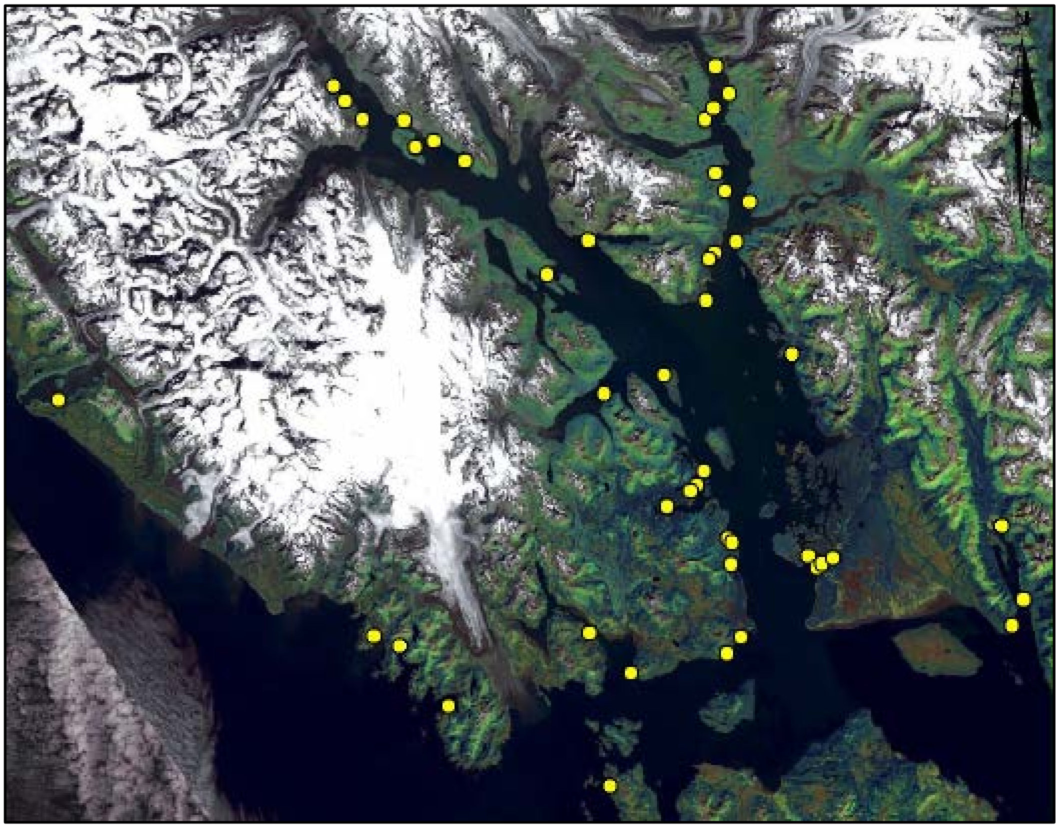

Figure 12. Map of SEAN marine contaminant sampling locations within GLBA (Tallmon 2012).

Visitor Impacts

Kralovec et al. (2007) provided information on the amount and distribution of visitor use in GBP from 1996-2003. Data were gathered through voluntary surveys completed by private boaters (motorized), sea kayakers, and backcountry campers and from the park's backcountry permit program. This information allowed researchers to identify peak travel periods, popular destinations, frequently used campsites, anchorages, travel routes, party size, and duration of visits (Kralovec et al. 2007).

Lewis and Drumheller (2004) reported the results of 2002-2003 surveys of campsite impacts on the GLBA shoreline. A total of 134 survey areas were visited, and 257 shoreline campsites were identified and assessed for social impacts (Figure 13, Photo 11). At each site, surveyors recorded campsite locations, signs of humans (e.g., footprints, trails), and signs of animals (e.g., tracks, scat, nests) (Lewis and Drumheller 2004). Animal signs and dominant plant species were noted in order to identify potential ecological impacts from campers at these sites. Species of particular management concern included invasive plants, shore-nesting birds, black and brown bears, harbor seals, and spawning salmon (Lewis and Drumheller 2004). Social impact indicators documented included vegetation damage, trailing, inter- and supratidal fire pits, trash, firewood, and footprints, as well as the size of impacted site. These indicators were used to calculate a final social impact rating, which fell into one of four categories: none, low, medium, and high (Lewis and Drumheller 2004).

Photo 11. A backcountry shoreline campsite in Reid Inlet at GLBA (Photo by Kelly Goonan, Utah State University).

Figure 13. Locations of survey areas (pink) along Glacier Bay visited by Lewis and Drumheller (2004) and Goonan et al. (2015) (Tania Lewis, November 2015).

Goonan et al. (2015) designed and tested a recreation impact assessment protocol for coastal backcountry campsites at GLBA. Surveyors visited the same study areas assessed by Lewis and Drumheller (2004) in 2002-2003 (Figure 13). However, Goonan et al. (2015) noted that statistical comparisons between the two studies are difficult due to differences in assessment parameters and the variety of methods used. Goonan et al. (2015) assessed 265 backcountry campsites during the summer of 2012. The assessment protocol defines a campsite as

> *…a location containing clear evidence of recent (within last 2-3 years) camping activity. Evidence of camping includes: vegetation loss or flattened vegetation clearly caused by human use (i.e. in the pattern of a tent, framed by tent rocks, etc.), compressed gravel clearly caused by human use (i.e. in the pattern of a tent, framed by tent rocks, etc.), recently placed tent rocks, camp trash, recent tree or shrub damage, campfire in site (Goonan et al. 2015, p. 16).*

Variables and attributes documented in Goonan et al. (2015) were litter/trash, campfire signs, human-made structures, human waste, tent rocks, damage to live trees/shrubs, vegetation types and cover on-site and in control areas, landing (beach) substrate type, and campsite substrate type. Unlike previous assessments (e.g., Lewis and Drumheller 2004), Goonan et al. (2015) focused exclusively on quantitative counts/estimates of human impact variables and did not assign qualitative impact categories (e.g., low vs. high).

4.2.5. Current Condition and Trend

<u>Abundance and Distribution of Anthropogenic Marine Debris in Inter- and Supratidal Zones</u>
The first documented surveys for marine debris at GLBA occurred in the Bartlett Cove area in the early 1990s (Rettew 1991, Polasky 1992). Debris found on beaches during these surveys included fishing floats, styrofoam, metal objects, ropes, and plastic sheets, bottles, and bags (Polasky 1992). During 1992, non-styrofoam plastics were the most common debris, comprising 56% of items, and styrofoam items were second most common at 23% (Polasky 1992). The Lester and Young Island beaches accumulated more debris than the Point Gustavus-to-Bartlett-Cove-dock stretch. During the 31 days between surveys of the island beaches in 1992, 55 debris items accumulated, while only seven items accumulated in 33 days between surveys along the other stretch (Polasky 1992). In 1991, Rettew (1991) noted that more debris was noted on both beaches during the second survey of the summer than during the first survey, even though debris had been accumulating for a longer period of time (i.e., all winter) prior to the first visit. Rettew (1991) suggested that summer boat traffic in the Bartlett Cove area was the primary source of marine debris in the survey areas. It is important to note that these data are now nearly 25 years old and are geographically limited to Bartlett Cove; while these surveys provide some insight into debris conditions at that time, they cannot be used to evaluate current conditions in GLBA.

Using aerial photos taken in 2012, the AK DEC assigned qualitative debris ratings to 1.6-km (1-mi) beach segments along the outer coast of southeastern Alaska (AK DEC 2013). Segments along the outer coast of GLBA were rated from zero (no debris) to three (moderate debris) (Figure 14; AK

DEC 2015). The longest stretches with moderate ratings were just south of Lituya Bay and south of Dry Bay (Figure 14).

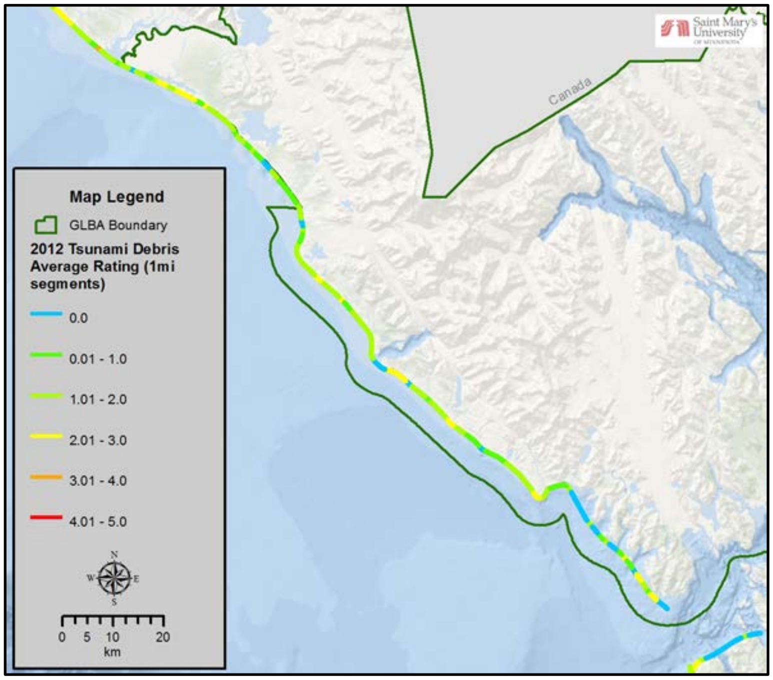

Figure 14. Debris ratings for 1.6-km (1-mi) beach segments along the outer coast of GLBA, based on 2012 aerial photos (AK DEC 2015).

Abundance and Distribution of Stranded Contaminants

Incidents resulting in known contaminant releases within GLBA boundaries are relatively rare (Eley 2000). However, accidents involving boats that could result in contaminant spills could occur almost anywhere in the park, because private and commercial excursion boats frequently enter even out-of-the way inlets and bays in search of wildlife and natural beauty (Eley 2000). The SEAN marine contaminants monitoring program indicates that contaminant levels are generally very low in GLBA's intertidal areas and that marine water quality is relatively pristine compared to most of the U.S. (Tallmon 2012). Metals tested for and detected in mussel samples were arsenic, cadmium, and mercury. All of the mussel samples collected from GLBA in 2007, 2009, and 2011 contained arsenic and cadmium, although very few samples were at or above 1.0 µg/g (Tallmon 2012). However, all four mussel samples collected in 2011 showed arsenic and cadmium concentrations above 1.0 µg/g (Table 8). All the GLBA mussel samples, except for one from Ripple Cove in 2011, also contained detectable levels of mercury. Only four samples in 2007 and three samples in 2011 (out of four) showed concentrations at or above 0.01 µg/g (Tallmon 2012). These levels are still quite low, particularly compared to much of the U.S. (Kimbrough et al. 2008). Slight increases were seen between 2007 and 2011, but are not thought to indicate a strong temporal pattern (Tallmon 2012). Metal contaminant levels for the four GLBA sites sampled in multiple years and for the "hot" control sites sampled in 2007 are shown in Table 8.

Table 8. Metal contaminant levels (µg/g wet tissue) in mussel samples collected in or near GLBA (Tallmon 2012; see Figure 15 for site locations). Some samples were also tested for tributyltin, but no detectable levels were found. For comparison, Kimbrough et al. (2008) considers 5-11 µg/g of arsenic, 0-3 µg/g of cadmium, and 0-0.17 µg/g of mercury to be low concentrations in mussel samples from the lower 48 states.

Year	Arsenic	Cadmium	Mercury
Bartlett Cove			
2007	0.60	0.41	0.0088
2009	1.10	0.75	0.0084
2011	1.10	1.30	0.0100
Ripple Cove			
2007	0.70	0.51	0.0086
2009	0.53	0.26	0.0023
2011	1.40	1.30	<LOQ[A]
E. Russell Rocks			
2007	0.71	0.60	0.0051
2009	0.26	0.15	0.0025
2011	1.80	1.00	0.0170
W. Hazelton Camp			
2007	1.80	0.76	0.0069
2009	0.52	0.18	0.0022
2011	1.30	1.20	0.0120
Hot control sites (2007 only)			
Outer Elfin Cove[B]	0.75	0.47	0.0100
Excursion Fish Plant[C]	0.47	0.60	0.0086
Bartlett Fuel Dock	0.51	0.41	0.0094
Bartlett Boat Ramp	0.88	0.49	0.0082

[A] <LOQ = below quantitation limits (i.e., measurable levels)

[B] This site is outside GLBA boundaries, on the south side of Cross Sound.

[C] This site is just outside GLBA boundaries, on the east side of Excursion Inlet.

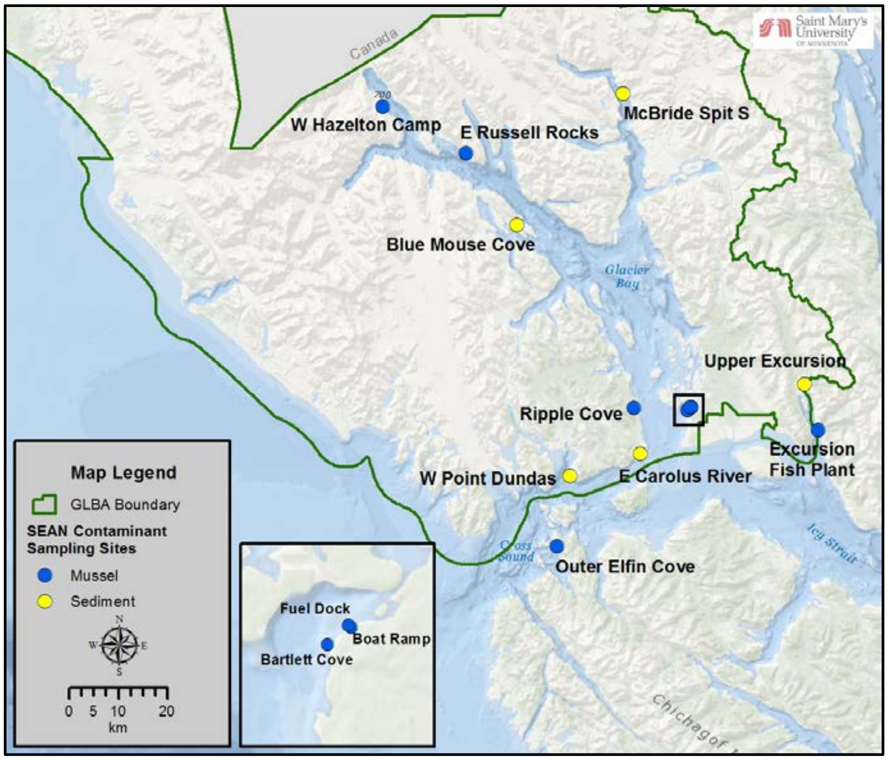

Figure 15. Contaminant sampling locations for which data are included in Table 8-Table 10 (Tallmon 2012).

Organic pollutants (PAHs and POPs) were frequently below quantifiable levels and were generally low when detected. However, the highest total PAH level detected in SEAN parks was from a Bartlett Cove fuel dock mussel sample (a "hot" control site), with a concentration of 1,488 ng/g (Tallmon 2012; Table 9). According to Tallmon (2012), an examination of the relative PAH profiles suggests the source of contamination is likely creosote, which is consistent with materials used in the old dock construction. A 2007 mussel sample from Berg Bay also showed an unexpectedly high total PAH concentration at 138 ng/g (Tallmon 2012). The PAH profile of this sample suggested a recent petrogenic source, possibly related to boat use of the bay (Tallmon 2012). Of five GLBA sediment samples tested, only two contained detectable levels of total polycyclic aromatic hydrocarbons (TPAHs) (Tallmon 2012; Table 10). POP levels were also low when detected. Polychlorinated biphenyls (PCBs) were the most commonly detected POPs, occurring in over 75% of mussel samples, but only samples from two of the "hot" control sites showed concentrations above 2 ng/g (Tallmon 2012; Table 9). Hexachlorocyclohexanes (HCHs) and polybrominated biphenyl ethers (PBDEs) were detected in GLBA samples for the first time in 2011; however, these were just above detection limits and are not a serious cause for concern (Tallmon 2012).

Table 9. Total polycyclic aromatic hydrocarbons (TPAH) and persistent organic pollutants (POP) contamination levels (ng/g) in mussel samples collected in or near GLBA (Tallmon 2012).[A] For comparison, Kimbrough et al. (2008) considers 63-1,187 ng/g of PAHs, 0-8 ng/g of CHLD, 0-112 ng/g of DDTs, and 3-153 ng/g of PCBs to be low concentrations in mussel samples from the lower 48 states.

Year	TPAH	ΣCHLD	ΣDDT	ΣHCH	ΣPCB	ΣPBDE
Bartlett Cove						
2007	<LOQ	<LOQ	<LOQ	<LOQ	0.62	<LOQ
2009	0.78	<LOQ	<LOQ	<LOQ	0.36	<LOQ
2011	0.67	<LOQ	<LOQ	0.29	<LOQ	0.50
Ripple Cove						
2007	<LOQ	<LOQ	<LOQ	<LOQ	0.77	<LOQ
2009	0.48	<LOQ	<LOQ	<LOQ	0.35	<LOQ
2011	0.84	<LOQ	<LOQ	0.41	<LOQ	<LOQ
E. Russell Rocks						
2007	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ
2009	0.83	<LOQ	<LOQ	<LOQ	0.36	<LOQ
2011	0.34	<LOQ	<LOQ	0.32	<LOQ	0.16
W. Hazelton Camp						
2007	<LOQ	<LOQ	<LOQ	<LOQ	0.66	<LOQ
2009	1.09	<LOQ	<LOQ	<LOQ	0.33	<LOQ
2011	0.30	<LOQ	<LOQ	0.33	0.32	<LOQ
Hot control sites (2007 only)						
Outer Elfin Cove[B]	69.74	<LOQ	0.48	<LOQ	3.7	6.3
Excursion Fish Plant[C]	13.55	0.45	0.25	<LOQ	1.8	<LOQ
Bartlett Fuel Dock	1,488.27	<LOQ	<LOQ	<LOQ	2.2	<LOQ
Bartlett Boat Ramp	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ

[A] POP abbreviations are: CHLD = chlordanes, DDT = dichloro diphenyl trichloroethanes, HCH = hexachlorocyclohexanes, PCB = polychlorinated biphenyls, PBDE = polybrominated biphenyl ethers. <LOQ = below quantitation limits.

[B] This site is outside GLBA boundaries, on the south side of Cross Sound.

[C] This site is just outside GLBA boundaries, on the east side of Excursion Inlet.

Table 10. Total polycyclic aromatic hydrocarbons (TPAH) contamination levels (ng/g) in sediment samples collected in GLBA (Tallmon 2012; see Figure 15 for site locations).

	McBride Spit S.	Blue Mouse Cove	Upper Excursion	E. Carolus Riv.	W. Point Dundas
TPAH	<LOQ*	3.59	6.94	<LOQ	<LOQ

* <LOQ = below quantitation limits.

Threats and Stressor Factors

Threats to the marine shoreline related to marine debris, contaminants, and human impacts include release of debris/contaminants by vessels, shoreside users (e.g., campers or boaters), and nearby developed areas, and uncontrolled shoreside visitor use. Vessels that travel in or near GLBA waters range from small private boats to cargo ships (limited to the outer coast and Cross Sound/Icy Strait). These vessels are a potential source of marine debris and contaminant spills (Eley 2000, Wuebben et al. 2000, EPA 2011).

Plastic marine debris poses a serious threat to the marine environment, as it is persistent (e.g., does not break down for many years) and is widely transported by ocean currents (Rios et al. 2010, EPA 2011). Plastic can also smother benthic organisms living in marine sediments, as it blocks out light and reduces oxygen exchange (Uneputty and Evans 1997, EPA 2011). Many plastics contain POPs such as phthalates, organotins, and phenols, which can be released into the environment as plastics slowly degrade (Rios et al. 2010, EPA 2011). Other pollutants such as PAHs and PCBs can attach to plastics and spread through the environment with the drifting debris, potentially entering the food chain if plastic particles are ingested by wildlife (Frias et al. 2010). In addition to degrading habitat and threatening wildlife, plastic marine debris can also interfere with navigation, impact recreational fishing, and reduce tourism (Carswell et al. 2011, EPA 2011).

Vessels are also potential sources of contaminants through chemical spills (e.g., fuel, lubricating oils, or hydraulic fluid), wastewater discharge, or ballast water release (NPS 1998, Wuebben et al. 2000). Some fuels and other oils are persistent chemicals, meaning that very little of the product would evaporate following a spill, and the contamination could be transported long distances by wind and/or currents (Eley 2000). In the late 1990s, two small passenger vessels grounded within GLBA, resulting in fuel spills (Eley 2000). The first grounding occurred near Geikie Rock on the west side of Glacier Bay in 1995, causing a minor fuel oil spill (Figure 16). The second grounding was in Dundas Bay in 1999 and resulted in a several hundred gallon diesel spill; fortunately, a swift response minimized the spill volume, limiting the environmental damage (Eley 2000). The NPS maintains a vessel fueling and underground petroleum storage facility at Bartlett Cove near the public dock, and small fuel spills do occur here (Wuebben et al. 2000). Fuel is delivered to Bartlett Cove by barge two to three times per year. Recently, the average amount of fuel per delivery is approximately 8,500 gallons of gasoline and 62,000 gallons of diesel fuel (NPS unpublished data). Other high-risk transit areas in or near the park where fuel spills could occur include Lituya Bay on the outer coast, North Inian Pass, Point Gustavus, and Sitakaday Narrows (Eley 2000; Figure 16). These were identified by boat pilots as “areas with the smallest margin for error in navigation or where a loss of power or steering would have the most serious consequence” (Eley 2000, p. 14).

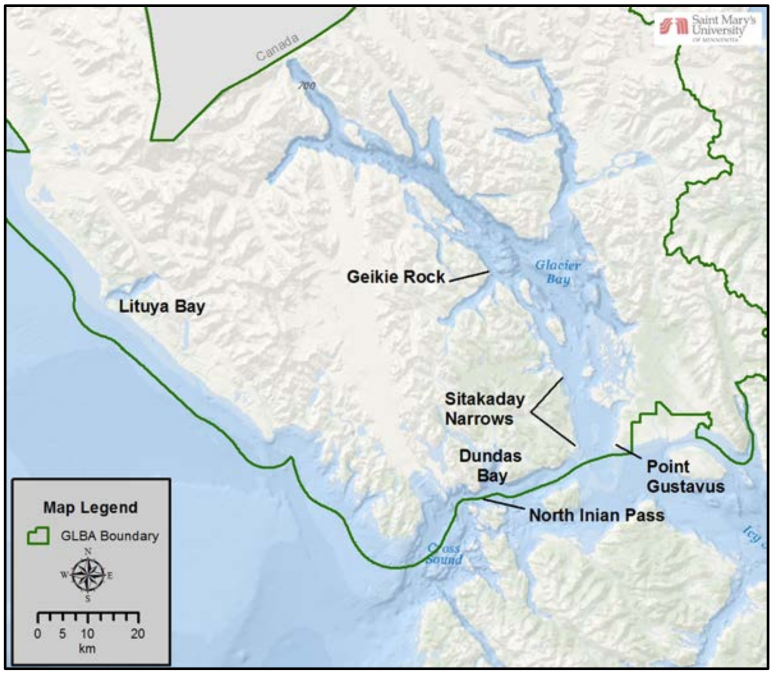

Figure 16. Locations where vessel fuel spills have occurred or the potential for vessel spills is high (Eley 2000).

GLBA currently experiences around 3,500 wilderness visitor use nights (i.e., backcountry camping) annually (Goonan et al. 2015). Although this number may sound high for a remote Alaskan park, it is about two-thirds of the level seen during a wilderness visitation peak in the late 1990s (Goonan et al. 2015). There are no designated campsites or travel routes in the GLBA backcountry, so campers are allowed to select their own sites. Wilderness visitors planning to camp in GBP are required to attend an orientation that teaches them about "leave no trace" camping, and they are asked to build fires only in the intertidal zone (Lewis and Drumheller 2004). Many visitor use impacts in wilderness areas result from trampling, which can damage vegetation or surface soil organic layers and compact mineral soils (McEwen and Cole 1997). This can reduce plant biomass and vegetative cover.

Based on voluntary camper surveys, Kralovec et al. (2007) concluded that much of the shoreline along GBP had been used for camping at some point during their study (1996-2003; Figure 17). The most commonly utilized sites where human impact was becoming apparent were often near designated concession drop-off/pick-up locations, tidewater glaciers, or freshwater streams. These areas include McBride and Lamplugh Glaciers, Adams and Johns Hopkins Inlets, Ptarmigan Creek, and the Beardslee Islands (Kralovec et al. 2007). The Beardslee Islands area was especially popular with campers and sea kayakers, as it is a motorless water area and can be accessed from Bartlett Cove without the use of a drop-off boat (Kralovec et al. 2007). For visitors that want to venture further into the backcountry, the Glacier Bay Lodge concession provides drop-off and pick-up service at several points within the Bay (Figure 18).

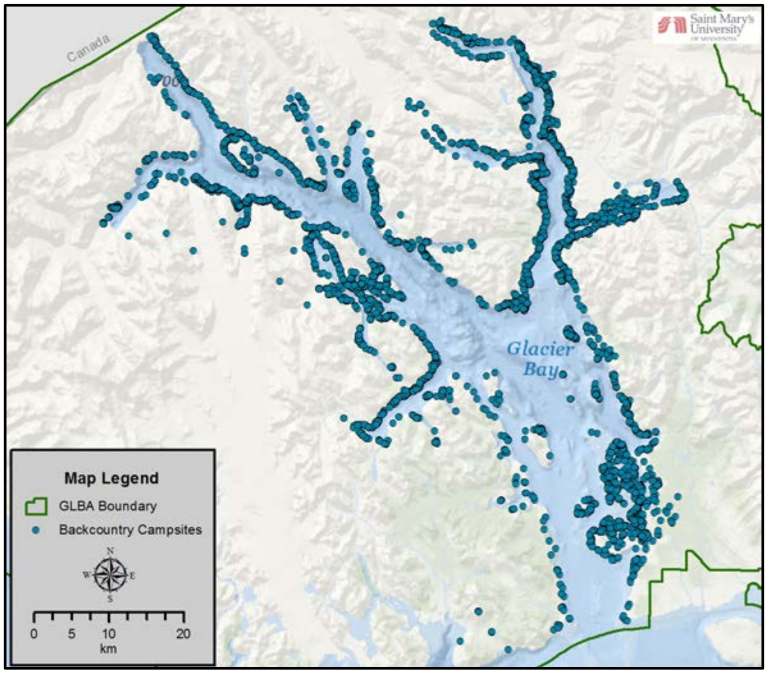

Figure 17. Backcountry campsite locations reported in voluntary camper surveys, 1996-2003 (Map created using GIS data provided by GLBA).

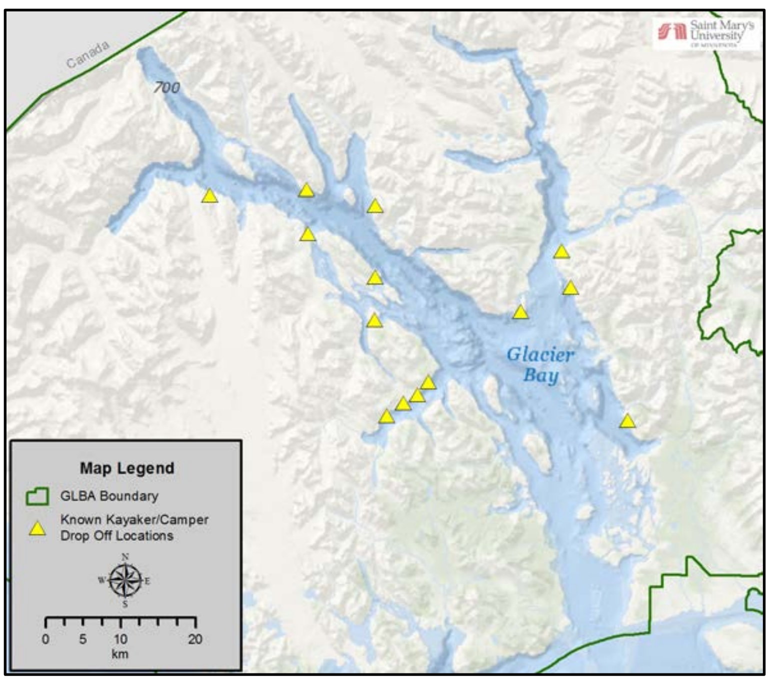

Figure 18. Known kayaker/camper drop-off locations. Between two and four locations distributed throughout the bay are used each year and are often rotated between seasons (Lewis and Drumheller 2004).

Data Needs/Gaps

Recent data on the abundance and distribution of marine debris in GBP are not available. Given the vast extent of the coastline and the lack of any evidence that marine debris is a serious concern, park-wide debris survey efforts may not be worth the time commitment and expense at this time. However, it may be useful to conduct surveys in the Bartlett Cove area, similar to Rettew (1991) and Polasky (1992), to determine whether marine debris accumulation in this higher-use area has changed over time. Tallmon (2012) recommends continuing mussel sampling every 2-5 years at a small number of sites and repeating larger-scale sampling once a decade. This will allow managers to monitor the most likely contamination sources in and around the park and also identify any long-term trends in contaminants (Tallmon 2012). Sampling was conducted in 2013 and 2015 (Lewis Sharman, GLBA Ecologist, written communication, 16 November 2015), but results were not available in time for inclusion in this assessment. Researchers have also recommended studying current structure and velocity and developing a water circulation model for the bay, to predict the spread of any potential contaminant spills in the park and to aid in a more effective spill response (Eley 2000, Wuebben et al. 2000). The impacts of camping and other visitor uses should continue to be monitored as well.

Overall Condition

Abundance and Distribution of Anthropogenic Marine Debris in Inter- and Supratidal Zones

The project team assigned a *Significance Level* of 3 for this measure. Surveys of the Bartlett Cove area in the 1990s showed that anthropogenic debris (e.g., plastics, discarded fishing gear, styrofoam) accumulates on park beaches and the most likely source was local boat traffic (Rettew 1991, Polasky 1992). The 2011 tsunami in Japan has increased the occurrence of marine debris on Alaska's shorelines, but most of GLBA's outer coast showed little to no debris during a 2012 aerial survey (AK DEC 2013, 2015). At this time, there is no evidence that marine debris is of concern park-wide. As a result, this measure is assigned a *Condition Level* of 1, indicating low concern.

Abundance and Distribution of Stranded Contaminants

This measure was also assigned a *Significance Level* of 3. The SEAN marine contaminant monitoring program has shown that contaminant levels are generally very low in GLBA's intertidal areas, with the exception of some "hot" control sites in areas of human activity (e.g., Bartlett Cove) (Tallmon 2012). Overall, marine water quality within GLBA is considered very good. Therefore, the *Condition Level* is currently a 1, or of low concern.

Camper and User Biophysical Impacts

This measure was assigned a *Significance Level* of 1. Measures with a *Significance Level* of 1 are not discussed in depth in the current condition section of this assessment, but available information is summarized here in the overall condition section. Camper impacts tend to be localized and their intensity is dependent on frequency of use, user behavior, and environmental conditions (McEwen and Cole 1997). In 2002-2003, Lewis and Drumheller (2004) surveyed shoreline campsites along GBP for signs of human impact. The majority (81%) of the 257 campsites evaluated were classified as small (<250 m^2). In terms of evidence of human presence, 74% of sites visited contained rock rings, 28% contained footprints, 22% contained trash, 16% had noticeable trails, and 9% had supratidal fire pits (Photo 12; Lewis and Drumheller 2004). Fewer than 5% of sites contained structures, human waste, or firewood. Nearly half of the campsites (48%) showed no evidence of vegetative damage; 25% showed low levels of damage, 18% had moderate damage, and only 9% were classified as having substantial vegetative damage (Lewis and Drumheller 2004). Evidence of species of management concern was noted in all survey areas, indicating the potential for human impact to these species. The majority of campsites (59%) received overall impact ratings of "low." Only 4% of sites were rated as "high," while 23% received ratings of "moderate" and 14% showed no sign of human impact (Lewis and Drumheller 2004). Figure 19 shows the locations of campsites surveyed along with their final impact ratings.

Photo 12. Evidence of trailing (left) and a supratidal fire pit documented at GLBA shoreline campsites by Lewis and Drumheller (2004) (NPS photos).

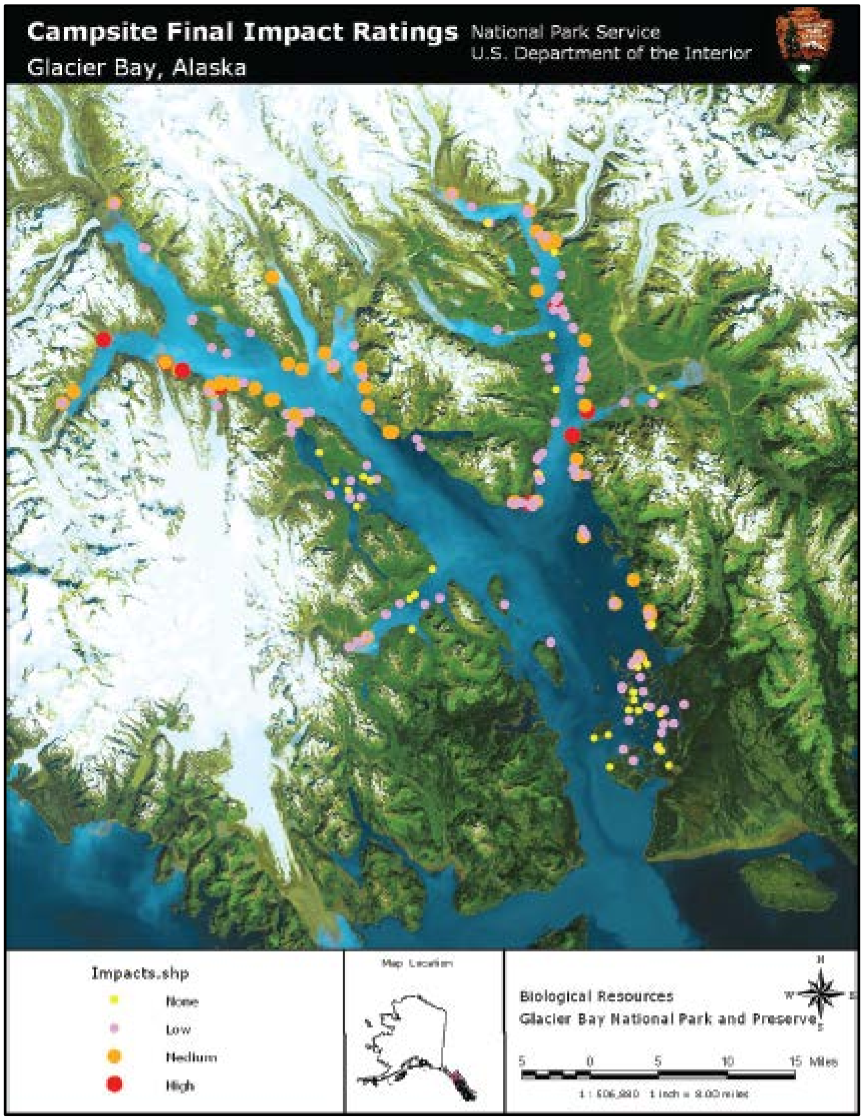

Figure 19. Final social impact ratings for Glacier Bay shoreline campsites, 2002-2003 (Lewis et al. 2007).

Goonan et al. (2015) revisited these campsite survey areas in 2012. Most sites were still considered small (room for one tent); only 10 of 265 campsites visited were of medium size or larger (two or three tents) (Goonan et al. 2015). Tent rocks were the most common evidence of human presence, occurring at all sites surveyed. The mean number of tent rocks per site was 10.5, and over 85% of sites had more than three tent rocks (Goonan et al. 2015). Reductions in vegetative cover were considered moderate, with cover reduced by more than 50% (compared to nearby control sites) at around 40% of campsites. However, very few trees or shrubs showed visible damage. Litter/trash was observed at 9-10% of sites and signs of campfires occurred at only 5% of sites (Goonan et al. 2015). Goonan et al. (2015) concluded that, overall, shoreline campsites appeared lightly impacted by users. Compared to data from 2002-2003 (Lewis and Drumheller 2004), the percentage of sites where campfire signs and litter/trash were found has actually decreased (Table 11). Based on these recent findings, camper and user biophysical impacts are assigned a *Condition Level* of 1, indicating low concern.

Table 11. Percentage of shoreline campsites where tent rocks, campfire signs, or trash were observed during 2002-2003 (Lewis and Drumheller 2004) and 2012 surveys (Goonan et al. 2015).

Year	% sites with rocks	% sites with fire signs	% sites with trash
2002-2003	92	11	24
2012	91	5	9

<u>Weighted Condition Score</u>

The *Weighted Condition Score* for GLBA's marine shoreline is 0.33, which falls at the high end of the good condition range. Marine shorelines are currently in good condition and appear to be unchanging, but any considerable contaminant release or increase in user impacts could shift the condition to moderate concern.

Marine Shoreline			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.33**
Abundance and distribution of debris	3	1	
Abundance and distribution of stranded contaminants	3	1	
Camper and user biophysical impacts	1	1	

4.2.6. Sources of Expertise

- Lewis Sharman, GLBA Ecologist

4.2.7. Literature Cited

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4.3. Harvested Marine Invertebrates

4.3.1. Description

The Alaskan coastal waters are home to many species of marine invertebrates; marine invertebrates can be found in the open-ocean, seafloor, and intertidal zones. Communities may include clams, snails, sea stars, barnacles, crabs, and marine worms (Field and Field 1999). Marine invertebrates are often sensitive to environmental changes and are considered indicator species in many ecosystems. It is also common for marine invertebrates to be foundation species. Foundation species provide the seafloor ecosystems with topographic relief, habitat for epifaunal organisms, and can change hydrodynamic processes (Dayton 1972). A wide variety of marine invertebrates occur in GLBA; however, this assessment focuses on Tanner crabs and weathervane scallops, two of several species of fish and invertebrates that are commercially fished within the park (Photo 13).

Photo 13. Weathervane scallops (top) and Tanner crabs (bottom) (NPS photos).

Tanner Crab Life History and Fishery Background

The Tanner crab is considered a true crab and is a popular fishery from Southeast Alaska to the Bering Sea. Tanner crabs feed on a variety of marine invertebrates including mussels, snails, clams, and marine worms (ADFG 2014), and are also prey for many bottomfish and pelagic fish species. Tanner crab embryos hatch and are adrift in the water column for about two months during late winter through early summer (Stratman et al. 2011). Tanner crab larvae undergo three stages of molts after which the young crabs, now called a megalops, spend another month free-floating before settling to ocean floor (Stratman et al. 2011). From hatch to maturity is about 5 years for females and 6 years for males and includes several instars (12 for females and 18 for males) (Stratman et al. 2011).

The Southeast Alaska (Area A) Tanner crab pot and ring fishery occurs within GBP and Excursion Inlet between 10-17 February to 1 May, typically for a 7-9 day opening during February. An estimated Area A male crab stock threshold of 2.3 million lbs or greater is required for the commercial fishery to commence (5 AAC 35.113 (a)). Fishery duration is dependent on stock abundance and the number of pre-fishery registered pots (5 AAC 35.113 (b) (1)). The majority of fishing effort in GBP (an ADFG designated productive or "core" area) occurs mid bay from approximately Willoughby Island to Sundew Cove (Figure 20).

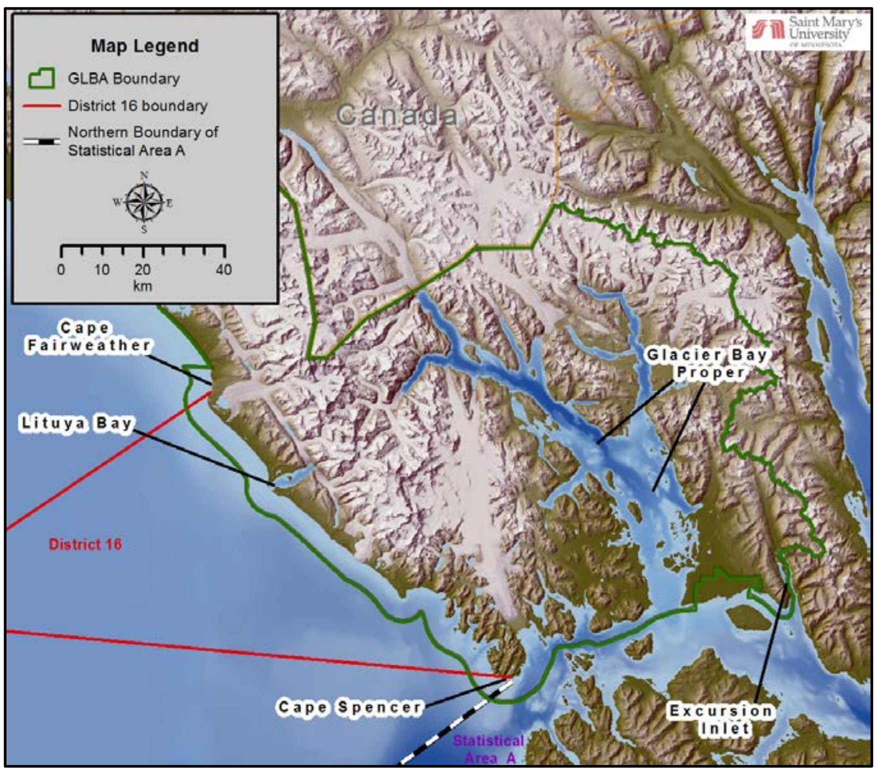

Figure 20. The Tanner crab statistical reporting areas include Statistical Area A (Southeast Alaska), Excursion Inlet, the two label whiskers indicating GBP are spanning the section where Tanner crab is harvested; the weathervane scallop commercial harvest reporting area is District 16.

The pot fishery is a limited entry fishery with an 80 pot permit limit (5 AAC 35. 125(b) (1)). Pot fishers may hold one of several possible permits, ranging from Tanner crab only to various Tanner plus king crab (i.e., brown, golden or blue; Lithodidae spp.) combination permits. The ring fishery is open access and permit holders are limited to 20 rings. Passive pot gear is significantly more effective than active ring gear and ADFG regulates ring harvest to ≤4% of long-term total harvest (5 AAC 35.116). Ring gear requires relatively calm sea conditions found in sheltered bays and inlets in order to be fished effectively. Some limited but undocumented recreational and personal use Tanner fishing occurs within the park in both GBP and Excursion Inlet.

While the Excursion Inlet fishery is authorized to continue indefinitely (36 CFR 13.1130), participants in the GBP Tanner crab fishery must hold a Lifetime Access Permit (LAP; 36 CFR 13.1132). LAP holders must have qualified to continue fishing in GBP based on past fishery participation (i.e., 3 years during 1980-1998; 36 CFR 13.1134(b) (2)). Permits are non-transferable, renewed at 5 year intervals (since 2000), and will sunset over time when LAP holders are no longer able to fish. Twenty-eight fishers held Tanner LAPs during the 2010-2015 period. However, five or fewer permit holders have reported landings in GBP since 2010. An age-based model of fishery participants incorporating variable annual attrition and advanced longevity indicates few LAP holders will continue fishing beyond 2050 resulting in a complete commercial Tanner fishery cessation over the next 3-4 decades (Chad Soiseth, GLBA Fisheries Biologist, written communication, 16 September 2015).

Pot and ring gear specifications are described in Alaska Administrative Code (5 AAC 35.125) but only aspects of pot gear and fishing pots are described here. Eighty eight inch diameter conical pots are typically fished at depths <200 fathoms. Pots must be rigged to allow escapement of undersized and female crab. At least four circular escape rings of 12.06 cm (4.75 in) minimum inside diameter are required for each pot. Only male Tanner crab with a carapace width of 13.97 cm (5.5 in) can be retained allowing at least one to two years of breeding before entering the fishery. All undersized males and female crabs must be released. Pots may be “double hauled” (i.e., two sets and pulls per day) during the first day or two of the opening when catches are high. However, one haul per day is more typical after initial catch rates decline. Short soak times have the potential for increasing handling injury (i.e., leg loss) or crab mortality in released crab. Cold temperatures below freezing during the February Tanner opening can result in leg loss rates of around 30% (Rumble et al. 2008).

<u>Weathervane Scallop Life History and Fishery Background</u>
Weathervane scallop, or the giant Pacific scallop, is also commercially harvested and an important fishery in Alaska. In Alaska, these scallops often bed in sand, gravel, and rock substrates and generally occur at depths of 40-130 m (1331-427 ft), but can be found in depths up to 300 m (984 ft) (NPFMC 2015). The Weathervane scallop beds are usually elongated, parallel to the direction of the current (NPFMC 2015). Spawning and gamete production is poorly understood, although temperature is considered a probable trigger that may initiate the release of sperm and eggs (NPFMC 2015). Eggs and sperm are discharged by the millions to be randomly fertilized in the water column, where fertilized eggs drift for several weeks (NPFMC 2015). At the larval stage Weathervane scallops actively swim within the drifting water column using a ciliated velum, which is a dual purpose organ that enables them to swim and feed on unicellular phytoplankton as they develop (NPFMC 2015). After two to three weeks, larvae settle to the bottom and metamorphose into a mature scallop, or spat, which often will attach itself to the substrate, swim, or use its foot to move around (NPFMC 2015). Mature scallops feed by filtering water for plankton and organic materials; they tend to reach reproductive age after 2-3 years (NPFMC 2015).

A weathervane scallop dredge fishery, authorized by the NPS, occurs off GLBA's outer coast from Cape Fairweather to Icy Point within District 16 from July 1 through February 15 (5 AAC 38.167) unless closed by emergency order. Fishery duration is dependent on exceeding the District 16 guideline harvest range of 35,000 lbs. of shucked meat. Scallop meat typically comprises only 8-12% of the total live weight of the catch including the shell. No stock assessment exists for this fishery. The majority of fishing activity is thought to occur within national park waters from north of Cape Fairweather to Lituya Bay. But a much smaller scallop bed also exists just southeast of the Lituya Bay entrance. Although precise locations for scallop beds within the NPS boundary are not well known, the smaller of the two beds is thought to occur entirely within the park and the larger one off Cape Fairweather straddles the park boundary three miles offshore.

Since January of 2014, as a consequence of a State of Alaska sunset stipulation, the weathervane scallop fishery is no longer limited entry but rather an open-access fishery. However, because of the larger vessel size (avg. vessel size 34.3 m [112 ft] in 1991) needed to accommodate and work the gear, there are very few permit holders engaged in and reporting harvest. No more than two scallop dredges (≤4.6 m [15 ft] wide) may be operated from a vessel at one time (5 AAC 38.076 (f) (4)). Typically, two or fewer vessels have been operating in park waters since 2001/2002. However, the spatial distribution, frequency and duration (i.e., total dredge hours) of fishing effort are currently not well known.

Vessels typically tow two "New Bedford" style dredges along the bottom at depths of 27-60 fathoms at a 4.7 knot average speed. The weight of a single dredge can exceed 1 metric ton (2,200 lbs; Pol and Carr 2002). Each dredge is attached to the towing vessel by a single steel wire cable controlled by a deck winch. A bag of 10.16 cm (4 in) diameter steel rings retain scallops and all organisms that exceed this dimension and fail to escape. Each haul occurs over about an hour and vessels can reportedly make 15-20 hauls per day. The ADFG observer program data indicates that 74-92% of District 16 catch weight was comprised of scallops over the 1999/00 through 2005/06 seasons with the remainder of catch comprised of bycatch and debris. Bycatch (by weight) over these seven seasons averaged 6% invertebrates, 4% skates and egg cases, 2% benthic fishes, 3% debris (i.e., kelp, rocks, wood, fishing gear and trash), and 2.5% unoccupied scallop shells.

Dredging impacts on benthic habitat and species include both direct and indirect effects. Direct effects include substrate scraping and plowing, sediment re-suspension, benthos mortality and disposal of processing waste. Indirect effects include displacement, delayed mortality, and long term changes to benthic community composition and abundance. Dredging can reduce habitat complexity by destroying or removing sea floor biological and/or physical structure. The spatial coverage, frequency, and duration of dredging effects determine the level of impact but these details currently remain unknown. Scallop dredging activity may be causing unacceptable impact or even impairment to benthic habitats and communities as outlined in NPS Management Policies (2006). However, NPS staff is currently unable to evaluate fishery impacts because Alaska Statute (16.05.815) does not currently specify the NPS as an excepted recipient and public access to fishery information identifying individual fishermen is prohibited.

Annual weathervane scallop stock assessments are reported in Alaska by the Scallop Plan Team (SPT). These reports document the annual catch, catch per unit effort (CPUE; in terms of pounds of meat per unit of effort (i.e., dredge hours), and provide an overview of current conditions and any new management concerns.

4.3.2. Measures

<u>Tanner Crabs</u>

- Fishing effort (# of vessels), harvest (lbs. crab) or harvest rate per unit effort (HPUE) within ADFG statistical area A (Southeast Alaska [Glacier Bay proper and Excursion Inlet]) (Figure 20)
- Estimated Tanner stock abundance (mature and legal males, females and recruitment by Glacier Bay Proper and Excursion Inlet survey area since 1999)
- Crab health and incidence of disease (e.g., Bitter crab syndrome [BCS])

<u>Weathervane Scallops</u>

- District 16 fishing effort (dredge hours, # vessels) and harvest (lbs meat, CPUE)
- Harvest amount in lieu of stock size
- Scallop health and incidence of weak meated scallop (WMS)
- Habitat status

4.3.3. Reference Conditions/Values

<u>Tanner Crabs</u>

Fishing Effort and Harvest

The reference condition for fishing effort (HPUE) will be the 1968 fishing season for Statistical/Registration Area A Tanner crab fishing effort; the number of permits issued (one permit = one vessel) was 29 for Statistical Area A. These were entirely pot fishing permits and the total harvest was 177,825 lbs for that season (Table 12; Stratman et al. 2011). This will serve as reference because it is the earliest available data that pertains to this measure within the area. To assess HPUE of Tanner crabs, the CPUE (crabs per pot lift) will be used. The earliest recorded CPUE is from 1993 when the CPUE was 16 crabs per pot lift (Table 12; Stratman et al. 2011).

Table 12. Reference condition values for Tanner crab fishing in Reg. Area A (Stratman et al. 2011).

Year	CPUE	Pot Fishery Permits	Ring Net Permits	Combined Gear Permits	lbs of crab
1968	n/a	29	0	29	177,825
1993	16	83	51	125	2,001,526

Although the earliest fishing efforts on record for Tanner crab in Southeast Alaska begin in 1968, which was prior to the commercialization of this fishery, the Tanner crab has been a sought after species in Southeast Alaska since the early 1960s (Stratman et al. 2011). Pot fishing permits are the primary and bulk of the total Tanner crab harvest in Alaska and ring net fishing lacks a CPUE. There are data starting in 1972 specifically for GBP and Excursion Inlet which is discussed in the current condition and trends section of this report.

Estimated Stock Abundance

Bishop et al. (2013) will be used to establish a reference condition for estimated Tanner crab stock abundance in Glacier Bay Proper and Excursion Inlet (Figure 20). The reader should note that the entire area within Excursion Inlet is not within GLBA, so estimates do not necessarily reflect the actual local stock abundance within the park portion of the inlet. Bishop et al. (2013) conducted catch-survey data modeling in order to obtain biomass estimates (stock abundance) of mature and legal Tanner crab stocks from several Southeast Alaska fishing areas; the reference condition time period is from 1993-2002. The estimated stock abundance (mean mature biomass [pounds]) for GBP was 1,021,928 lbs and Excursion Inlet was 318,820 lbs derived from the 1993-2002 catch-survey data (See Table 3 in Bishop et al. 2013).

Crab Health and Incidence of Disease

The major threat to Tanner crab stock health is incidence of BCS. Bitter crab syndrome is caused by a parasitic dinoflagellate (*Hematodinium* spp.) which causes a wasting condition in crab resulting in a bitter taste that makes the product commercially unsaleable. It was reportedly first observed in Southeast Alaska and has been found in at least a third of the commercial fishing areas managed by ADFG (Meyers et al. 1990). This disease is a threat to stock health because BCS has a 100% mortality rate in Tanner crabs that contract the infection (Meyers et al. 1996). The reference condition for crab health and incidence of disease will be zero occurrence for both GBP and Excursion Inlet as of 1987 (Meyers et al. 1990).

Weathervane Scallops

Fishing Effort and Harvest

The District 16 (Figure 20) reference condition for the fishing effort measure is based on stock assessment and fishery evaluations (SAFE) data from the 1994 scallop season. The fishing effort in 1994 involved seven vessels that dredged a total of 408 hours; the reference values for the fishing effort are 54 lbs of meat per dredge hour and 22,226 lbs of meat (retained catch) for the harvest (Table 13). There was one vessel in 1993, and the guideline harvest level (GHL) was 35,000 lbs of meat; however, the CPUE and total dredge hours were confidential and no values are available for that first recorded season (NPFMC 2006). GHLs are established to avoid overfishing and are adjusted annually as needed based on compliance with federal requirements for fisheries management to include an annual catch limit (ACL) (ADFG 2015).

Table 13. The reference condition for the fishing effort measure regarding the weathervane scallop stock in District 16 (NPFMC 2006).

Year	GHL (lbs meat)	Retained Catch (lbs meat)	CPUE (lbs meat per dredge hour)	# of Vessels	Dredge Hours
1994/1995	35,000	22,226	54	7	408

The first District 16 harvest data available are from 1994. Since this is the earliest record available, the 1994 harvest data will serve as the reference condition of weathervane scallop harvest. District 16 harvest in 1994 was 22,226 lbs (Table 13) with seven unique vessels reporting harvest. District 16 GHL at that time (35,000 lbs) was based on previous catch amounts from that area, which are not available (NPFMC 2006).

Harvested Scallop Shell Size Distribution

The reference condition for harvest shell size distribution, referring to the range in shell size (height), of retained scallops for District 16 (most relevant to the park) is from 1997 (Figure 20 and Figure 21; NPFMC 2006). The stock assessment report from 2007 contains a set of histograms that are records from 1997 for all Alaskan scallop stocks, for this assessment, only District 16 is discussed since it includes NPS waters from mean high tide to three miles offshore.

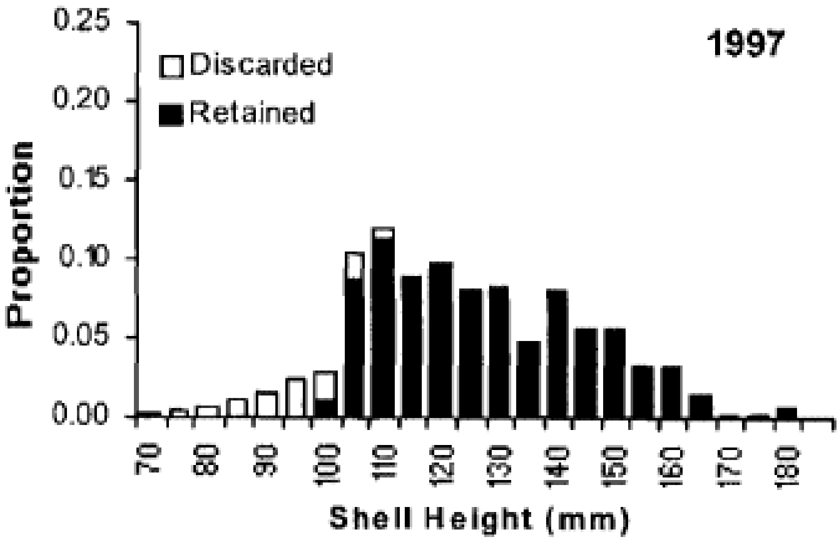

Figure 21. Scallop shell size distribution from District 16 scallop stock for 1997 (NPFMC 2007).

Scallop Health and Incidence of Disease

Weathervane scallops from some areas of the eastern Gulf of Alaska including areas near Yakutat (District 16) exhibit poor-quality adductor muscles characterized by off-white to grayish color, tissue that tears easily during hand shucking, and possessing a characteristic stringy texture and soft, spongy consistency (NPFMC 2008). The Alaska scallop industry refers to scallops exhibiting these characteristics as “weak meat” scallops and reports difficulty in marketing them (Brenner et al. 2012). Brenner et al. (2012) reported that WMS may be indicative of nutritional stress, although, further studies to determine causation are warranted. Weak meat scallops are reportedly typical in larger and older individuals (Brenner et al. 2012).

Weak meat scallops reportedly occur in this region, but there are no data available to assess a rate of occurrence or where it’s distributed. Due to this data gap, there is not a reference condition for this measure. WMS has varying degrees of prevalence in Alaska and is currently under study to gain understanding of what may cause scallops to develop weak meats. The occurrence of WMS is not discussed in the SAFE reports (NPFMC 2007-2008, 2010-2015) for District 16 which may indicate a lack of significant occurrence in those stocks.

Habitat Status

The trends in scallop abundances and distribution may be habitat status related, but there are not sufficient data to assess whether this is significantly impacted by harvest efforts. In Alaska, weathervane scallops are typically harvested by towing two New Bedford-style dredges that are 4.57 m (15 ft) wide (NPS 2009, Glass 2014). These dredges are considered efficient; however, there are bycatch of various bottom-dwelling (benthic) animals (Glass 2014). The species composition of bycatch can vary distinctly from place to place (i.e., scallop bed to scallop bed), even on a relatively small spatial scale (<50 km [<31 mi]) (Glass 2014). The benthic community structure differences are attributed to various physical parameters such as water depth, location and substrate (Glass 2014).

4.3.4. Data and Methods

Tanner Crab

Meyers et al. (1987) utilized necropsy and histopathology in a study intended to provide new information on the causal agent of BCS. The report included a gross and microscopic description of BCS and the features of pathogenesis in Tanner crabs in Alaska, ultrastructural morphology of *Hematodinium*, transmissibility in both Tanner and king crabs, and the prevalence and distribution in Alaskan Tanner crabs using the most current catch information at that time (Meyers et al. 1987). The information on prevalence and distribution will be the focus for the purposes of this NRCA. Meyers et al. (1987) used landing information from two Alaska processors to assess disease distribution and estimate the resulting commercial losses. Crabs with the appearance typical of BCS were confirmed by visual inspection of cooked and uncooked carcasses and finalized by a taste test; fisherman provided distributional information on legal size ≥140 mm (≥5.5 in) male crabs (Meyers et al. 1987). Additionally, the ADFG Commercial Fisheries Division conducted a systematic pot survey to determine infection prevalence between crab sexes and varied size classes (Meyers et al. 1987). These actions all occurred during 1986. The pot surveys used twenty 2x2 m (6.5x6.5 ft) commercial king crab pots inside a 1/3 nautical mile grid pattern in areas that were reported by fisherman to have high prevalence of diseased crabs (Meyers et al. 1987). These crab pots were baited with herring and set to depths of 15-60 fathoms (90-360 feet), each soaking for about 15-22 hours (Meyers et al. 1987). Random Tanner crabs caught in the survey were inspected in the manner described above along with collection of hemolymph smears (Meyers et al. 1987). The following year (1987) the grids were resampled in a similar manner using a 12 m (40 ft) wide beam trawl (Meyers et al. 1987).

Meyers et al. (1990) used data that was collected using systematic population surveys in several harvest areas in Alaska. The purpose was to determine the distribution of BCS within crab fisheries in Southeast Alaska to assess the prevalence of the disease in that area (Meyers et al. 1990). Catch census data was also used to obtain an estimated economic loss to the industry due to BCS in the area (Meyers et al. 1990). The methods of pot surveys and data acquisition for this report are described above in Meyers et al. (1987).

Zheng et al. (2006) developed a catch-survey (CS) analysis model for Tanner crab stocks in Southeast Alaska. The models estimated absolute abundance, mature and legal males, and relative abundance for mature females. Model inputs were derived from pot survey catch rates and commercial catch data time series ranging from a 6 year period (1999-2005) to a 27 year period (1978-2005) (Zheng et al. 2006). Tanner crab survey areas included GBP, Thomas Bay, Port Camden, and Port Frederick. Upon determination of survey catchabilities being similar for square pots 2.06x2.06 m (7x7 ft) and conical pots with 2.06 m (7 ft) base diameters, only the conical pots were employed in surveys after 2000 (Zheng et al. 2006). Initiation of Tanner crab surveys occurred in 1997, but GBP was not added until 1999 (Zheng et al. 2006). Sampling areas were stratified by depth and then pot locations were selected on a grid systematically until 2002 when the locations were selected randomly (Zheng et al. 2006). Soak times ranged from 16-24 hours; individuals captured were each assessed by sex, carapace width (to the nearest mm), shell condition (based on several factors), length condition, presence of parasites, and reproductive status (Zheng et al. 2006). The annual commercial catch was summarized as number of crabs for each stock (by location) from the catch database of ADFG (Zheng et al. 2006).

Nielsen et al. (2007), in conjunction with the USGS Glacier Bay Field Station, conducted surveys for two crab species in GBP during the summer of 2002. From July through August, a 1.5 km (0.9 mi) grid was delineated for sampling. A total of 415 sites were chosen within the grid and set with two pots each; a commercial crab pot was set to sample adults with a commercial shrimp pot tethered out 20 m (66 ft) to sample for juveniles (Neilsen et al. 2007). Captured crabs bodies were measured with a caliper to the nearest mm and the condition of each shell was assessed visually. The caliper measurements collected followed the Biological Field Techniques for *Chionoecetes* crabs outlined in Jadamec et al. (1999) (Nielsen et al. 2007). Mapped distributions of juvenile and adult females were created; females were used since males are indistinguishable as to adult or juvenile stages. The juveniles’ distributions were then characterized based on depth using statistical analysis. This analysis was done with the Kolmogorov-Smirnov two-sample test, comparing cumulative frequency distribution of CPUE for each age class with the cumulative frequency of depths sampled (Neilsen et al. 2007).

Bednarski et al. (2010) used Tanner crab stock data that was collected annually by the ADFG from 2001 to the time of this publication to compare trends in BCS among Tanner and red king crab stocks in Southeast Alaska. The information discussed regarding GBP and Excursion Inlet will be the primary focus of discussion although the analysis includes 14 geographically separate areas. Variations in prevalence of BCS, were examined within these localized stocks (Bednarski et al. 2010). Bednarski et al. (2010) examined: yearly trends of prevalence of BCS among areas; correlation between mature male biomass and BCS prevalence to determine the effect of host density; the difference between water depth and prevalence of BCS among areas; and differences in BCS prevalence between males and females and size classes. In short, this report was used primarily to discuss and compare the incidences of BCS in GLBA and Excursion Inlet since the established reference condition.

Stratman et al. (2011) provided an extensive report directed to the Board of Fisheries on the commercial Tanner crab fisheries in Region I in the Gulf of Alaska. Region I includes Registration Areas A and D, which are Southeast Alaska and Yakutat, respectively (Stratman et al. 2011). The report includes an introduction to the fishery followed by fishery details for both registration areas as they were available. Stratman et al. (2011) describes details such as life history, distribution, regional commercial fishery development and history, development of regulations, management concerns, stock assessment, and summaries of recent seasons.

Bishop et al. (2013) reported the methods and findings of two Tanner crab surveys that, in tandem, provide annual stock assessment information that is used to manage the Tanner crab fishery. Relevant parameters that were gathered by the two surveys included CPUE, female reproductive status, male length/weight relationship, male chela height/carapace width relationship, and visual detection of disease and limb loss (Bishop et al. 2013). From these parameters, assessments of stock health, exploitation rates, and the CS modeling were assessed. The CS models for each survey area were used to estimate total Tanner crab biomass; the survey and the commercial catch data were used to calculate the estimates (Bishop et al. 2013). Reproductive condition was studied by collecting 40 Tanner crab females from two survey areas, GBP and Thomas Bay, to assist clarification of various factors that may produce variability in reproductive condition of females (Bishop et al. 2013). Bitter Crab Syndrome instances were detected visually in Tanner crabs. To ensure accuracy in visually identified BCS detections, hemolymph samples were drawn from 100 randomly selected individuals at each survey location to test for the disease (Bishop et al. 2013). Stock health was assigned a range of decreasing or increasing trend scores (-6 to +6) that are categorized into five ranges: Healthy = >3.24, Above Average = 1.25 to 3.24, Moderate = -1.25 to 1.24, Below Average = -3.25 to -1.26, and Poor = <-3.26 and are assigned corresponding exploitation rates of 20%, 15%, 10%, 5%, and 0% of mature male crab or a maximum of 40% of legal male crab (Bishop et al. 2013).

<u>Weathervane Scallop</u>
The North Pacific Fishery Management Council (NPFMC) reviews SAFE reports on Alaska's weathervane scallop fisheries which are reported by the SPT under the management umbrella of the NPFMC. The NPFMC SPT (2006, 2007, 2008, 2010, 2011, 2012, 2013, 2014, 2015) reported on the weathervane scallop fishery stocks by summarizing current biological and economic status of the fishery; this involves reviewing the most recent analytical data that are collected and used in fisheries management (e. g. guideline harvest levels and harvest strategies). The SPT assembled these reports with input from several other agencies involved in the management of Alaska's natural resources. These include the ADFG, and the National Marine Fisheries Service (NMFS). The SPT reports these data annually in order to review the status of weathervane scallop stocks, discuss issues of importance to stock management, and compile the SAFE report.

4.3.5. Current Condition and Trend

<u>Tanner Crab</u>

Fishing Effort and Harvest
Stratmen et al. (2011) reported all fishing effort and harvest in Southeast Alaska from 1968 through 2010. The reference condition starts with 1968 fishing effort in Statistical Area A where there were a total of 29 permits (all pot permits) and a total harvest of 177,825 lbs. Figure 22 shows the total annual harvest in pounds for each subsequent year alongside the number of permits that were issued. This includes both pot and ring net permits after 1982, ratios are shown in the CPUE column of

Table 14, although ring net fishing only comprises a small portion (≤4% annually on average) of total harvest by statute. Harvest escalated quickly once commercialization occurred, and the annual estimated harvest was around 1.5 million lbs by 1972; this was an 88% increase in 4 years with very little increase in fishing effort (permits) (Stratmen et al. 2011). The 1981 season was the highest haul on record with 3.3 million lbs harvested between 1 December 1981 and 16 April 1982.

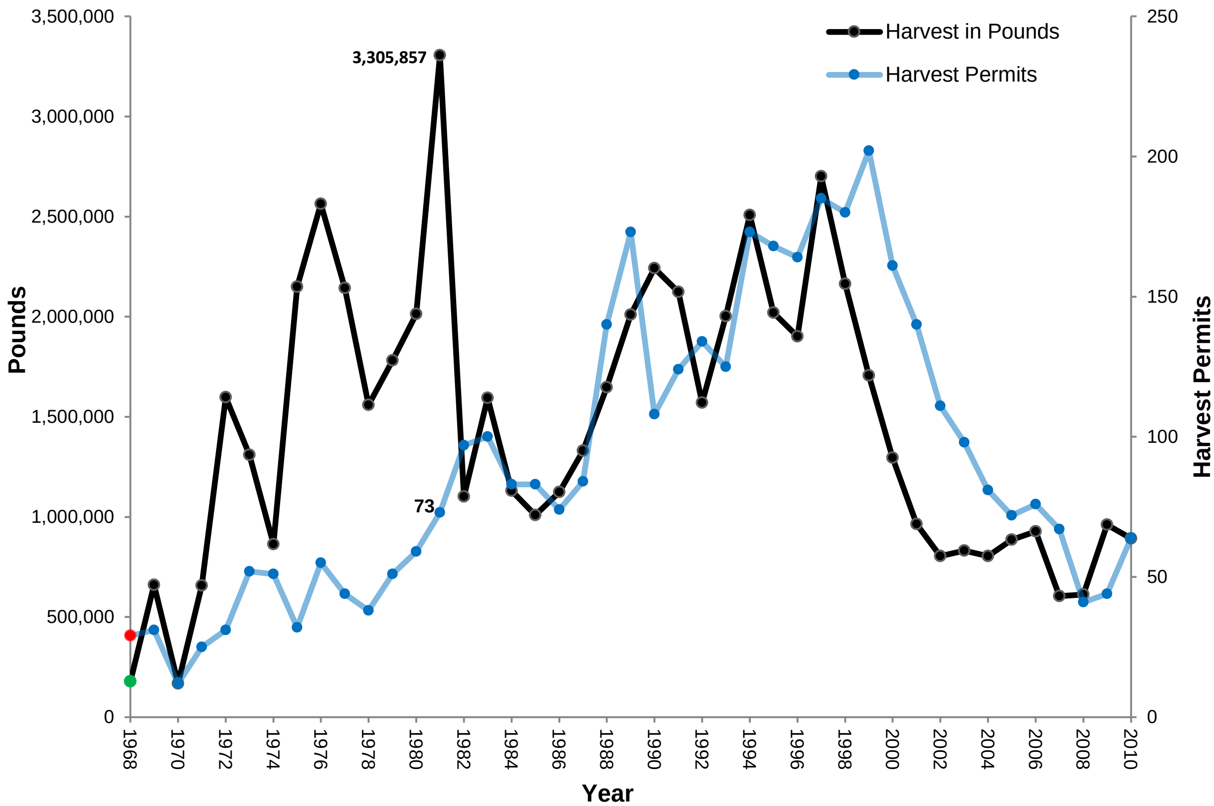

Figure 22. Tanner crab harvest and fishing effort from 1968-2010 within Southeast Alaska (Statistical Area A). The 1968 reference condition as represented by harvest (lbs; green marker) and effort (no. of permits; red marker) (Stratman et al. 2011).

Fishing effort has also fluctuated since commercialization. The bumper harvest of 1981 was obtained with a total of 73 (pot only) permits (Figure 22, Table 14; Stratman et al. 2011). The largest number of permit holders, including both pot and ring net permits, to fish Southeast Alaska for Tanner crab was in 1999, when 202 permits were issued (Figure 22; Stratman et al. 2011). A total of 64 permits (combined gear, both pots and ring nets) reported harvest in Southeast Alaska during 2010; the last year that exceeded 100 permits was in 2002 when 111 permit holders reported (Stratman et al. 2011). Overall, the total number of permits issued has been in steady decline since the late 1990s possibly as a consequence of limited entry, which was imposed in 1997 (Stratman et al. 2011; Soiseth, written communication, 21 August 2015). The CPUE has fluctuated somewhat over the years, with highest recorded in 1997 when CPUE was 24 crabs per pot lift, but subsequent CPUEs have remained below 20 (Table 14; Stratmen et al. 2011).

Table 14. Total CPUE, permits (both types), and lbs of Tanner crab; the bolded line indicates when the CPUE began to be recorded in 1993 (Stratman et al. 2011).

Year/ Season	CPUE (crabs per pot lift)	Combined Gear Permits (pot:ring)	lbs of crab
1968	–	29 (29:0)	177,825
1969	–	31 (31:0)	660,337
1970	–	12 (12:0)	167,378
1971	–	25 (25:0)	656,661
1972	–	31 (31:0)	1,597,838
1973	–	52 (52:0)	1,309,673
1974	–	51 (51:0)	863,751
1975	–	32 (32:0)	2,149,397
1976	–	55 (55:0)	2,563,710
1977	–	44 (44:0)	2,142,409
1978	–	38 (38:0)	1,559,769
1979	–	51 (51:0)	1,781,175
1980	–	59 (59:0)	2,013,276
1981	–	73 (73:0)	3,305,857
1982	–	97 (95:2)	1,101,630
1983	–	100 (100:0)	1,593,468
1984	–	83 (78:5)	1,130,924
1985	–	83 (72:11)	1,009,005
1986	–	74 (67:7)	1,123,974
1987	–	84 (71:13)	1,330,485
1988	–	140 (77:63)	1,646,332
1989	–	173 (81:92)	2,009,669
1990	–	108 (72:36)	2,241,593
1991	–	124 (83:41)	2,122,921
1992	–	134 (83:51)	1,569,687
1993	16	125 (81:44)	2,001,526
1994	17	173 (91:82)	2,507,147
1995	16	168 (94:74)	2,020,036
1996	16	164 (94:70)	1,900,819
1997	24	185 (92:93)	2,701,322
1998	21	180 (93:87)	2,164,131
1999	17	202 (92:110)	1,706,156
2000	14	161 (81:80)	1,295,680
2001	12	140 (83:57)	964,836
2002	13	111 (67:44)	804,234
2003	14	98 (68:30)	832,158

Table 14 (continued). Total CPUE, permits (both types), and lbs of Tanner crab; the bolded line indicates when the CPUE began to be recorded in 1993 (Stratman et al. 2011).

Year/ Season	CPUE (crabs per pot lift)	Combined Gear Permits (pot:ring)	lbs of crab
2004	17	81 (60:21)	804,035
2005	18	72 (53:19)	886,521
2006	16	76 (57:19)	927,900
2007	14	67 (49:18)	605,062
2008	14	41 (31:10)	612,550
2009	19	44 (33:11)	961,681
2010	17	64 (48:16)	891,344

The fishing efforts and harvests that have occurred in GBP and Excursion Inlet alone are a small portion of the total harvest from Statistical Area A in Southeast Alaska. The earliest data available for GBP starts in 1972/73, with some years missing due to confidentiality of certain data (i.e., fewer than three permits issued). Only about 3.8% of the entire harvest of Tanner crab in Statistical Area A in 1972/73 was harvested in GBP. Some years have had larger portions (e.g., 1980/81 GBP comprised 12.8% of total harvest), but in general it has been a very small portion of total Statistical Area A Tanner crab harvest (lbs). In the most recent harvest season (2014/2015), GBP harvest was 71,492 lbs, which was about 5% of the Statistical Area A harvest (Figure 23; Stratman, ADFG data).

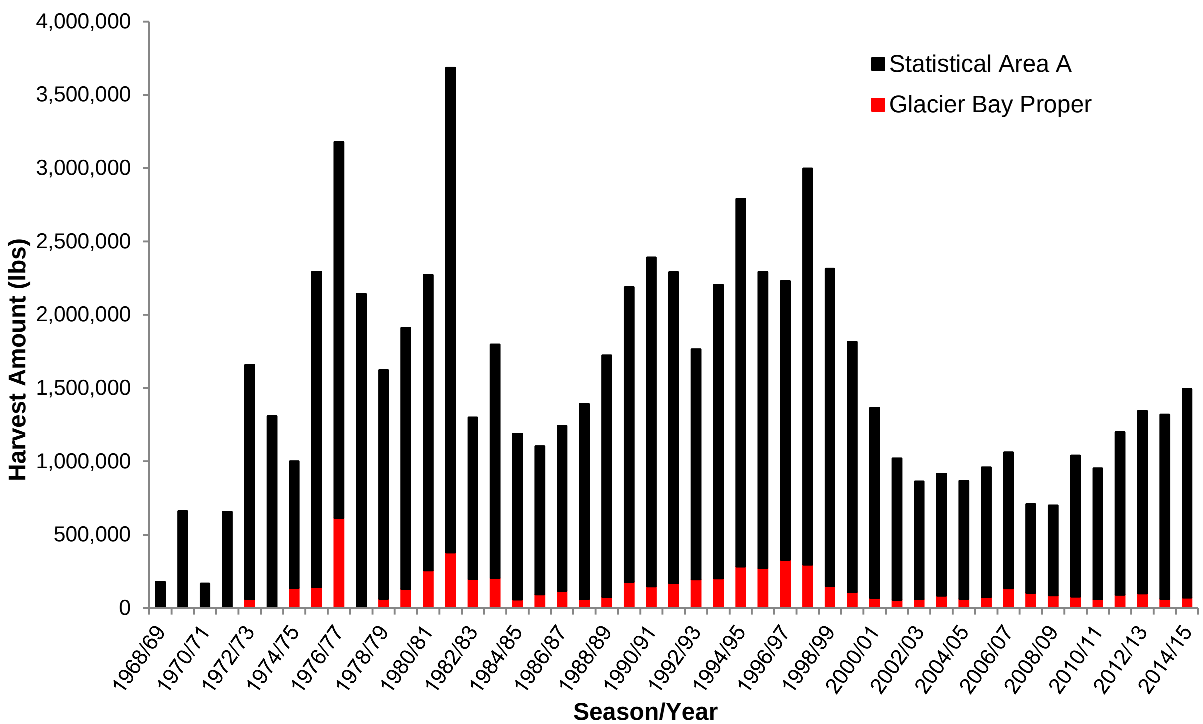

Figure 23. Total Tanner crab harvests in Statistical Area A (lbs) from 1968/69 to 2014/15 and the portion of it that was harvested in GBP; data available for GBP start in the 1972/73 season (Stratman, ADFG data).

The oldest available data for the portion of Tanner crab harvest (lbs) that was taken from Excursion Inlet is from 1977/78 when this was about 1.4% of the total harvest from Statistical Area A (Messmer, ADFG data). During the largest harvest year in 1981/82, when over three million pounds of Tanner crab were harvested from Statistical Area A, only 2.2% (73,576 lbs) was harvested in Excursion Inlet and it is not known how much of the Excursion Inlet harvest was actually harvested within the GLBA boundary (Messmer, ADFG data). During the most recent season on record (2014/15) about 2% of the total Tanner crab harvest was taken form Excursion Inlet (Figure 24; Messmer, ADFG data).

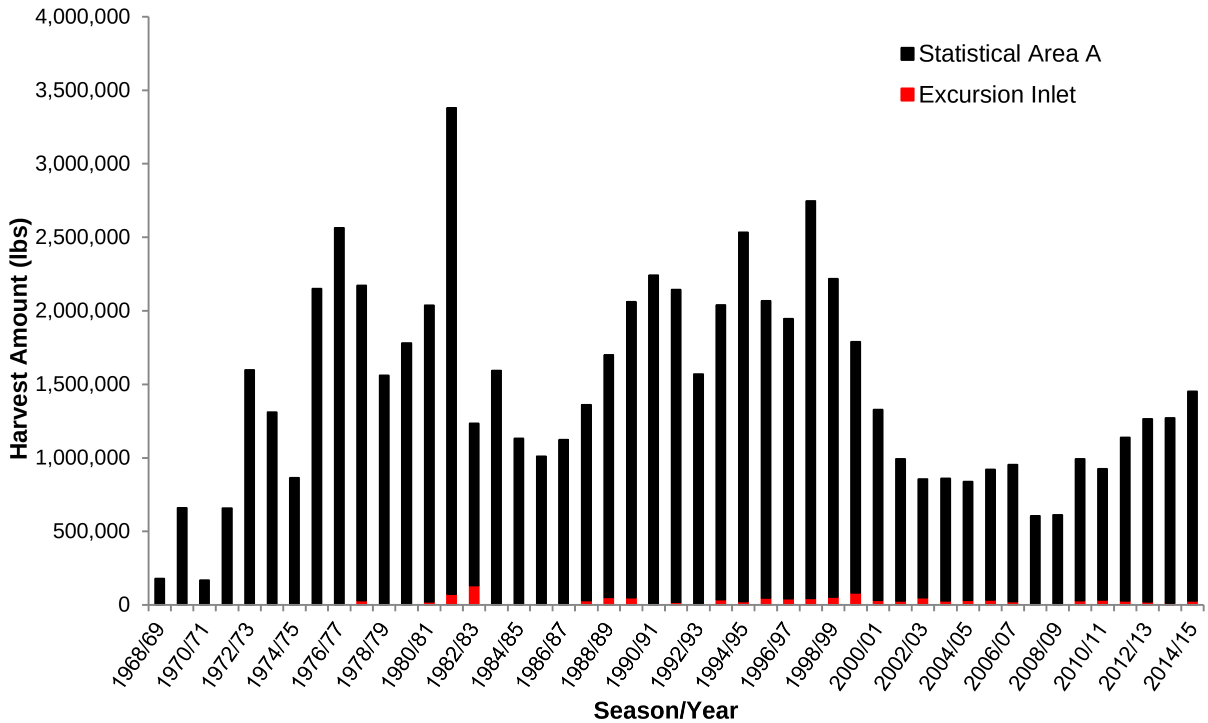

Figure 24. The total Tanner crab harvests in Statistical Area A (lbs) from 1968/69 to 2014/15 and the portion of it that was harvested in Excursion Inlet, for which the first available data begin in 1977/78 (Messmer, ADFG data).

Estimated Stock Abundance

The reference condition of estimated Tanner crab stock abundance in GBP and Excursion Inlet compared to subsequent estimates shows a downward trend in GBP (Bishop et al. 2013), with Excursion Inlet also showing a decrease but not nearly as pointed (Bishop et al. 2013). Between the 1993-2002 estimate and the 2011 estimate there is a 75% decrease (decreased by 769,440 lbs) in mean mature biomass (estimated stock abundance) in GBP, and a 24.5% decrease (decreased by 78,017 lbs) in Excursion Inlet (Table 15; Bishop et al. 2013).

Table 15. The estimated Tanner crab stock abundances (mean mature biomass) in GLBA and Excursion Inlet (Bishop et al. 2013).

Year	Survey Area	Mean Mature Biomass (lbs)
1993-2002	Glacier Bay	1,021,928
	Excursion Inlet	318,820
2010	Glacier Bay	346,303
	Excursion Inlet	258,409
2011	Glacier Bay	252,488
	Excursion Inlet	240,803

Determining trends in estimated stock abundance in GBP and Excursion Inlet is accomplished by estimating rates of male and female recruitment and the long term CPUE averages. The overall stock health can be assessed using data on the average CPUE of mature females, clutch fullness of gravid females, and prerecruit, recruit, and postrecruit males. Tanner crab surveys were conducted in Glacier Bay from 17-26 October 2011 (Bishop et al. 2013). In Glacier Bay, the stock health score had improved slightly from the year before; in 2010 the score was -2.00 ("below average") and in 2011 was -1.25 ("below average"), an increase of 0.75 (Bishop et al. 2013). The increase was attributed to CPUE of recruit and post-recruit crab no longer falling below the long-term average, although mature females and pre-recruit crab are still below the long-term average (Bishop et al. 2013). However, there were significant declines in pre-recruits in the short-term trends (Bishop et al. 2013). Previous catch survey analysis (CSA) estimates have shown that GBP had high recruitment in comparison to ten other stocks in Southeast Alaska during the early to mid-2000s (see Figure 9 in Zheng et al. 2006). In comparison to the reference condition, both recruitment (CPUE of mature females, and male prerecruits, and recruits [Figure 23 and Figure 24]) and estimated stock abundance (mean mature biomass in lbs) have recently declined in GLBA (Table 15; Bishop et al. 2013).

Recruitment within the Southeast Alaska Tanner crab stocks is what comprises the majority of harvests from 1979 onward, which is a concern as even a short term recruitment failure could result in stock collapse (Zheng et al. 2006). This heavy reliance on recruitment is likely the result of high harvest rates depleting older, large crabs, leaving the fishery reliant upon recruitment for harvest (Zheng et al. 2006). For a Tanner crab fishery to achieve recruitment into the fishery (only males are harvested) the carapace must reach the minimum size of 13.97 cm (5.5 in) in width; the condition of the shell relative to molt-phase is also a factor in recruitment categorization. Zheng et al. (2006) conducted studies on 11 Tanner crab stocks in Alaska, including GBP.

In Glacier Bay, male pot survey catchability was the lowest of all 11 stocks, while estimates of selectivity for sublegal males were among the highest (Table 16; Zheng et al. 2006). The values in Table 16 are probabilities; male pot survey catchability or simply "catchability" is the relationship between how many are caught per available (crabs) per effort unit (pot) and per unit of time (hour) and is scaling parameter to convert crabs per pot lift to absolute crab abundance (Zheng et al. 2006).

Table 16. Estimates of GLBA male Tanner crab survey catchability, selectivity, and molting probability, and new shell female selectivity that were derived from catch-survey analysis (CSA); values are reported as probabilities (Zheng et al. 2006).

Category	GLBA Value	Excursion Inlet
Male Pot Survey Catchability	0.00004158	0.00006461
Sublegal Male Selectivity	1.000	0.900
Mean Sublegal Male Molting Probability	0.320	0.854
New shell Female selectivity	0.628	0.232

The average molting probability for legal males was lowest in GBP, while new shell mature female selectivity was mid-range in GBP (Table 16; Zheng et al. 2006). Excursion Inlet had the lowest new shell mature female selectivity of all the stocks, but was one of the stocks with a much higher rate in male pre-recruit CPUE during the 1990s and 2000s, while others remained relatively constant (Table 16; Zheng et al. 2006).

Bishop et al. (2013) also found that GBP exhibited a marked difference between the sub-legal (pre-recruit or smaller) CPUE (crab per pot lift) and the legal (recruit) CPUE after 2000. While Glacier Bay Tanner crab stock exhibited much higher pre-recruit than recruit catch rates (CPUE), the other stocks exhibited the opposite trend (Bishop et al. 2013). Since Glacier Bay has exhibited a relatively high recruitment within its Tanner crab stock, it may serve as a highly valuable stock reserve. The Excursion Inlet stock had notable differences between CPUEs of mature males and females between 1984 and 2005. In the early 1980s, the CPUE of males and females was high with only a slight increase in male CPUE from 1984-2005, while female CPUE sharply increased during the same time frame (Zheng et al. 2006).

Overall, the Tanner crab legal male harvest rates in Southeast Alaska have been relatively high; however in Glacier Bay they were relatively low (i.e., <0.5) (see Figure 8 in Zheng et al. 2006). The estimated average legal harvest rate[1] of Tanner crabs was 0.60 for all Southeast Alaska stocks combined, and ranged from 0.38 to 0.84 during 1978 through 2005; the highest legal harvest rates[1] (0.79-0.84) were from 1979 through 1981 in Southeast Alaska (Zheng et al. 2006).

Estimated CPUEs of mature males and females separately are depicted in Figure 25 (Bishop et al. 2013). The Glacier Bay mature male CPUE has declined with prerecruits in recent years and is a concern for that stock since all size classes have a discernable reduction in the CPUE, and harvests are heavily reliant on recruitment (Figure 25; Zheng et al. 2006, Bishop et al. 2013). There is also a slight increase in CPUE of mature females in GBP since 2009, where the CPUE has been below the long term average for that stock (Figure 25; Bishop et al. 2013). Excursion Inlet is also shown in Figure 25, but doesn't appear to be in the same condition or downward CPUE trend as the GBP stock

[1] The legal harvest rate is defined as the ratio of retained catch to estimated legal abundance adjusted by natural mortality to the midpoint of each fishing season (Zheng et al. 2006).

(Figure 25; Bishop et al. 2013). There is actually an increase over the last four years (since 2008) in postrecruit mature male CPUE in Excursion Inlet, although the reader should note that this stock doesn't necessarily lie entirely within the park. The park boundary bisects Excursion Inlet and commercial fishing undoubtedly occurs both within and outside the park although the distribution of the fishing grounds is currently not known (Figure 20, Figure 25; Bishop et al. 2013).

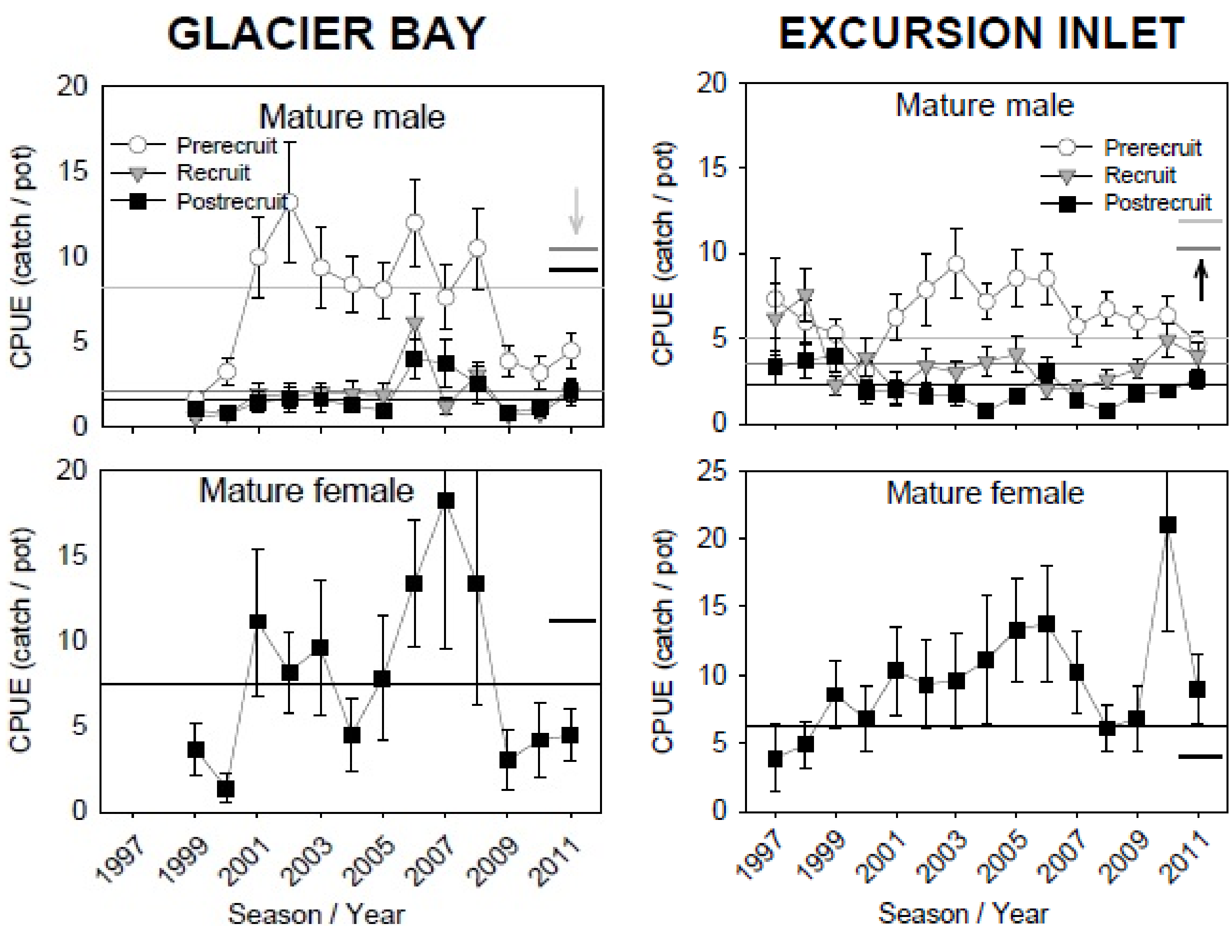

Figure 25. The CPUE of mature male and female Tanner crab in Glacier Bay (left) and Excursion Inlet (right), beginning in the late 1990s; the reference line through the midway of the figure represents the long-term average CPUE (Bishop et al. 2013).

According to Zheng et al. (2006), male crabs are categorized as prerecruit, recruit, and postrecruit; the meaning of these terms is to categorize males in terms of whether they will or will not meet the guidelines for legal harvest size, which is based on size range-categories of carapace width and molt interval. A prerecruit is defined as a male crab between 109-137 mm (4.3-5.4in) and will reach legal harvest size of >14 cm (>5.5 in) carapace width following its next molt by the next harvest season. A recruit is simply a crab that has achieved legal retention size for the first time and ranges between 138-169 mm (5.4-6.7 in). A postrecruit is a crab with a new shell carapace ≥170 mm (≥6.7 in) or an old shell carapace width ≥138 mm. The size at male sexual maturity is defined as >109 mm. There are also juveniles, which are crabs that are ≤109 mm (≤4.3 in) in width.

Habitat quality has become a topic of study in light of drastic Tanner crab fishery stock fluctuations that have been documented over recent years in Alaska (Neilsen et al. 2007). The relative abundance and distribution of Tanner crabs was studied in GLBA in the summer of 2002 (Neilsen et al. 2007). The purpose of the study was to assess the possibility of the fjords inside of GBP serving as a nursery and refuge for the Tanner crab stock (Neilsen et al. 2007) after cessation of the LAP fishery. Understanding the spatial relationship of Tanner crab recruitment processes could have considerable implications in determining the best management practices to ensure healthy stocks that will effectively sustain future harvests (Neilsen et al. 2007). That being considered, the permit holders that are currently authorized for commercial Tanner crab harvests in GBP (currently estimated at <30 in 2015) have what is designated as a "Lifetime Access Permit" (Soiseth, written communication, 21 August 2015). Authorized fishers can continue fishing as long as they are able but can't sell or pass the permit along to a successor. GBP will eventually become protected from commercial harvest entirely, which is in alignment with Public Law. It is anticipated that elimination of commercial harvest of Tanner crab in GBP will occur sometime after 2050 (Soiseth, written communication, 21 August 2015).

Other management techniques have involved spatial considerations by defining marine reserves and defining essential fish habitat designations to protect source populations; this spatial approach to fishery management requires a general understanding of target animal distribution, movement and habitat associations that are involved with the fishery being managed. Surveys conducted by Neilsen et al. (2007) found that Tanner crabs in Glacier Bay were dispersed widely, but were segregated by size class (i.e., juveniles and adults congregated separately). The segregation is thought to be the result of competition between adults and juveniles for the same food sources, as well as cannibalism by adult crabs that will prey on juveniles. The areas outlined in red in Figure 8 had higher concentrations of juvenile Tanner crabs. Forty-four percent of all juveniles captured during Neilsen et al. (2007) surveys were caught in Wachusett Inlet and Scidmore-Charpentier Inlet (Figure 26). Although Neilsen et al. (2007) didn't directly address the status of the habitat itself, the study did indicate that there are habitats in Glacier Bay that are favored by the juvenile component of the local population. These patterns may represent an avoidance response to adults and their associated habitat, areas that juveniles actively avoid in order to avoid being eaten as well as avoiding competition for food (Neilsen et al. 2007). In general, Glacier Bay Tanner crabs were widespread, segregated by age group, with the juveniles found in thick clustered groups separate from the adults (Neilsen et al. 2007).

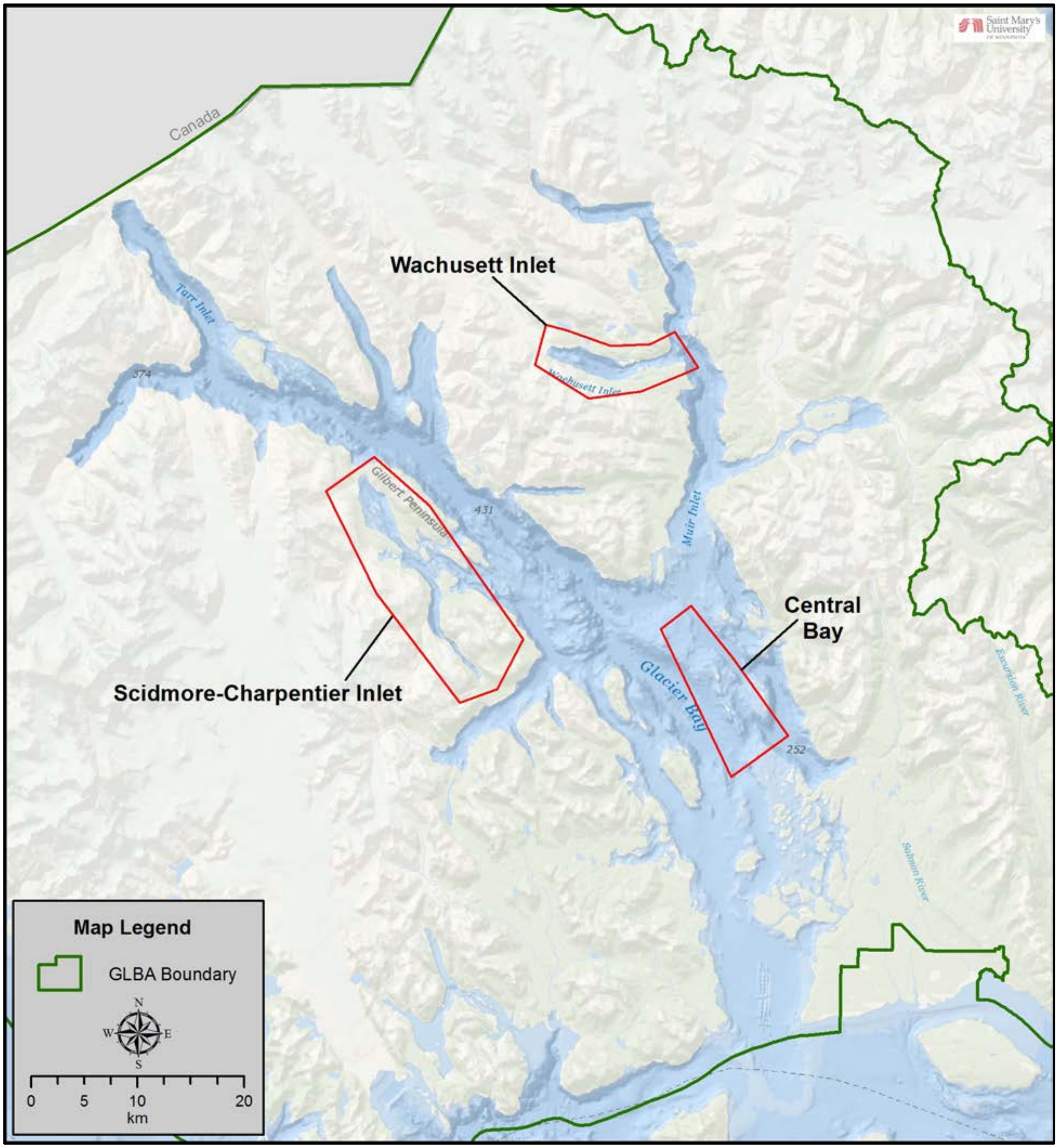

Figure 26. Areas where congregations of juvenile Tanner crabs were found during Neilsen et al. (2007).

Crab Health and Incidence of Disease

At the time of the 1987 reference condition for crab health and incidence of disease, BCS was not detected in crabs in GBP or Excursion Inlet. However, researcher and survey results indicated that it was widespread in the other Alaskan Tanner crab fisheries at the time (Meyers et al. 1990). In fact, a third of commercially fished areas in Southeast Alaska were detected as infected with BCS in 1988 and 1989 (Meyers et al. 1990). BCS was initially confirmed in Southeast Alaska in 1985, but was likely present prior to this (Meyers et al. 1987, Bednarski et al. 2010). Results from Meyers et al. (1990) showed infected crabs were detected (5.6% of sampled crabs) in Excursion Inlet in 1988 when the year prior there were zero incidences detected in the area. Infected crabs were discarded overboard in Southeast Alaska (sorting, where diseased crabs were separated from healthy ones), quite possibly in areas other than where they were caught, likely leading to exacerbation of the disease (Meyers et al. 1996).

Bednarski et al. (2010) surveyed for incidence of BCS in 14 locations including GBP and Excursion Inlet each year from 2001 through 2008 (Figure 20). This survey effort found BCS in all areas with varied levels of prevalence (Bednarski et al. 2010). Overall results indicated a strong relationship between depth and incidence of BCS; according to Bednarski et al. (2010) this suggests that shallow and deep water habitats may be more stressful and lead to higher incidence of BCS, although this was reportedly not a significant factor for GLBA or Excursion Inlet. There was also a strong relationship between BCS incidence and sex and shell condition. Males had much higher incidence rates than females, and both male and female crabs with new shell condition (recently molted) had significantly higher incidence of BCS than those with old shell condition (Bednarski et al. 2010). Size mattered with new shell males, although in some areas the larger crabs had higher incidences, while the opposite was observed in other areas (Bednarski et al. 2010). In Excursion Inlet, BCS incidence increased with the size of new shell male crabs, this was the case for eight other areas as well (Bednarski et al. 2010). In Glacier Bay, the incidence of BCS decreased in larger new shell males (Bednarski et al. 2010).

Weathervane Scallops

Weathervane Scallop Fishing Effort

District 16 fishing effort was reported in NPFMC (2006, 2007, 2008, 2010-2015) annual SAFE reports and the 1994 fishing efforts reported by the SPT serve as the reference condition for this measure (1994 refers to the 1994/1995 year in the table, hereafter the first part of the season/year will be used). The reference condition for fishing effort and harvest are 54 lbs of meat per dredge hour and 22,226 lbs of meat (retained catch) for the 1994/95 season since it is the earliest available data on fishing effort in the District 16 area. Subsequent years are shown in Table 17. There has been concern by park management about the potential but unknown impact of this activity on benthic habitat and community composition, changes to the GHL, a declining trend in catch rate (CPUE) and a recent (as of 2014) shift from a limited entry to an open access fishery.

Table 17. Compiled results of SAFE reports on District 16 weathervane scallop stock harvest rates and fishing effort (NPFMC 2006-2008, 2010-2015).

Year	GHL (lbs meat)	Retained Catch (lbs meat)	CPUE (lbs meat per dredge hour)	Number of Vessels	Dredge Hours
1993/1994	35,000	n/a	n/a	1	n/a
1994/1995[A]	35,000	22,226	54	7	408
1995/1996	35,000	33,302	30	6	1,095
1996/1997	35,000	34,060	37	2	917
1997/1998	35,000	22,890	41	4	561
1998/1999	35,000	34,153	49	2	702
1999/2000	35,000	34,624	51	2	674
2000/2001	35,000	30,904	65	3	476
2001/2002	35,000	20,398	49	2	417
2002/2003	35,000	3,685	37	2	100
2003/2004	35,000	1,072	60	2	18
2004/2005	35,000	24,430	58	2	419
2005/2006	35,000	13,650	34	2	407
2006/2007	21,000	13,445	44	2	309
2007/2008	21,000	180	30	1	6
2008/2009	21,000	20,986	50	2	423
2009/2010	25,000	11,791	27	2	439
2010/2011	25,000	2,655	32	1	83
2011/2012	25,000	1,777	31	1	57
2012/2013	25,000	25,255	37	1	684
2013/2014	25,000	25,510	40	2	634
2014/2015	25,000	9,141	21	2	439

[A] Is the reference condition of weathervane scallop fishing effort and harvest amount (also highlighted in grey).

The annual GHL was reduced from 35,000-21,000 lbs of meat during the 2006/07 season (Table 17). There were two vessels that year and a total of 309 dredge hours were reported, but CPUE had declined from 60 pounds of meat/dredge hour in 2003 to 44 over the three year interval (Table 17, Figure 27). The GHL subsequently increased during the 2009 season to 25,000 lbs and has remained at this level despite catch rates that have remained at 40 pounds or less of meats per dredge hour. The GHLs are based on data collected from previous harvest seasons; a decline in catch rate can result in a decreased GHL for that stock. Fishing effort in District 16 has fluctuated downward from seven vessels and stabilized since the 1994 reference period. Fishing effort in terms of number of vessels has remained at between one and two vessels since 2001, while the total hours of dredging effort has fluctuated annually from 6-684 dredge hours (Table 17, Figure 28). Overall scallop catch rates increased up to 65 meats per dredge hour in 2000 and exhibited a declining trend to about a third of the peak level to 21 meats per dredge hour in 2014; however, these fluctuations may be a factor of severe weather conditions hampering harvest efforts. Moreover, the decreasing harvest rate evident over recent years corresponds with an increasing trend in fishing effort measured as dredging hours.

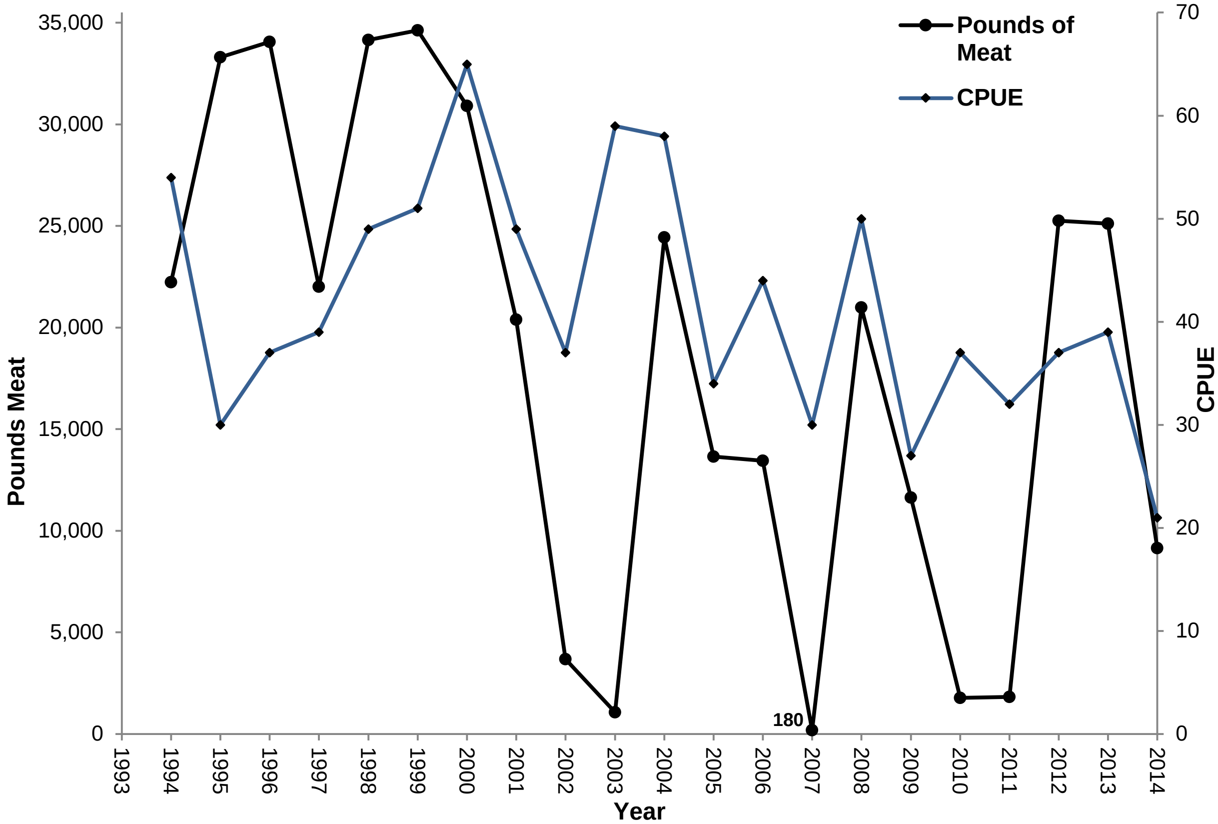

Figure 27. Weathervane scallop commercial harvest and catch rate for District 16. The graph shows pounds of weathervane scallop meat in black, and the CPUE (lbs of meat per dredge hour) in blue, from 1994 to 2014 (NPFMC 2006-2008, 2010-2015).

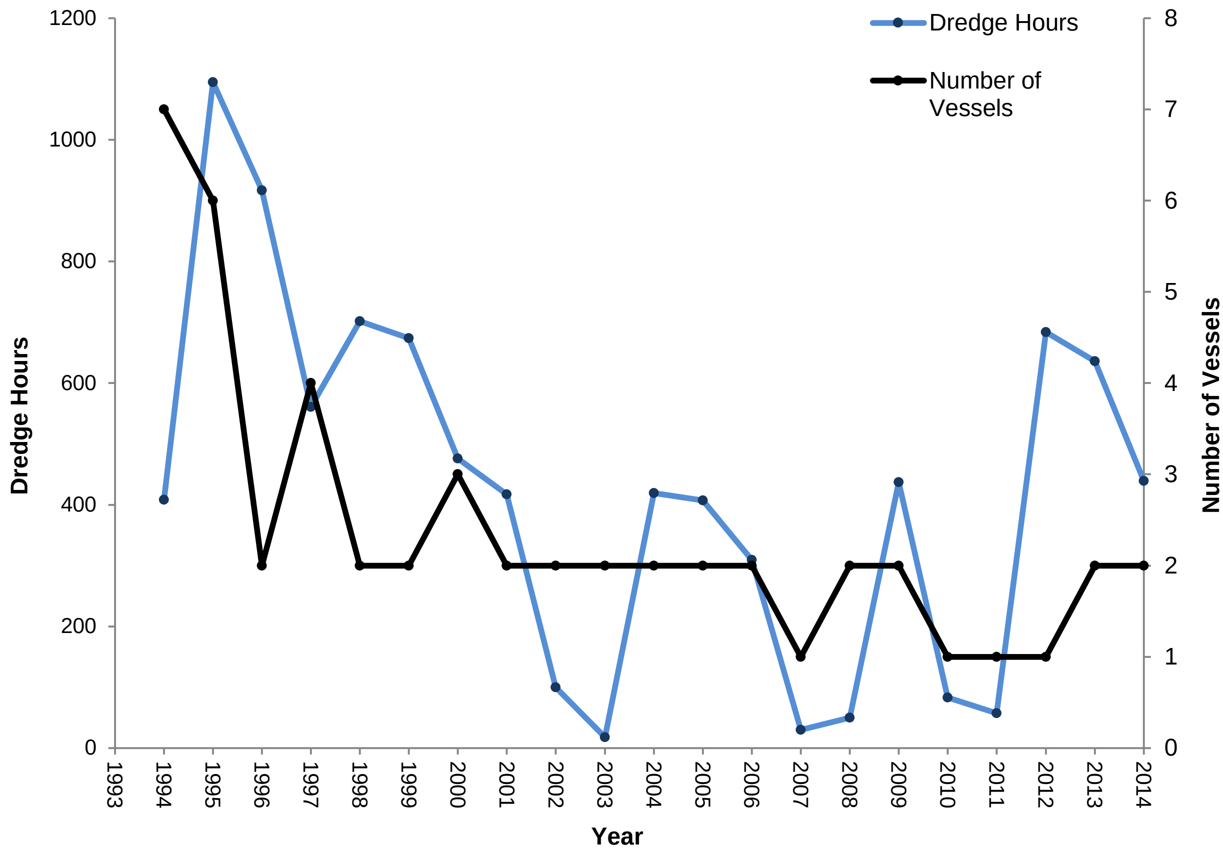

Figure 28. Weathervane scallop commercial fishing effort in District 16. Scallop fishing effort: dredge hours are depicted in blue and the number of vessels in black (NPFMC 2006, 2008, 2010-2015).

Figures 3-1 in NPFMC (2015) indicates that according to results from the 2013/14 fishery, District 16 exhibited the 5th largest (of nine areas) retained catch, 5th largest amount of effort (dredge hours) and exhibited one of the lowest catch rates (Semidi I and SW District were comparable or less) (NPFMC 2015). Some undersized scallops are likely crushed by the dredges and not captured. Similarly, not all scallops are brought to the surface as some individuals, at least during the early part of the tow, slip through the 4 inch diameter (ca. 101.6 mm) opening rings. Although some, perhaps most, survive, some are likely injured during the dredge up and pass through the dredge bag.

Fisheries harvest and effort over two decades in District 16 indicate that scallop meats represent only 8-12% of the total round weight (biomass) depending on area and season (NPFMC 2015). So a GHL of 25,000 lbs is roughly equivalent to 250,000 lbs of scallop which is, in actuality, a ten-fold greater amount of biomass than indicated solely by harvest.

Harvest Shell Size Distribution

The reference condition for harvest shell size distribution, referring to the range in shell size (height), for District 16 is from 1997 (Figure 20 and Figure 21; NPFMC 2007). This is different than the reference for harvest size due to the lack of axis values in the NPFMC (2006) SAFE report. Scallop shell height range in District 16 has decreased recently, and according to the NPFMC (2015) stock assessment, District 16 has seen an increase in smaller scallops with shell heights <110 mm (4.3 in) since the 2011/2012 and 2013/14 seasons (Figure 29; NPFMC 2015).

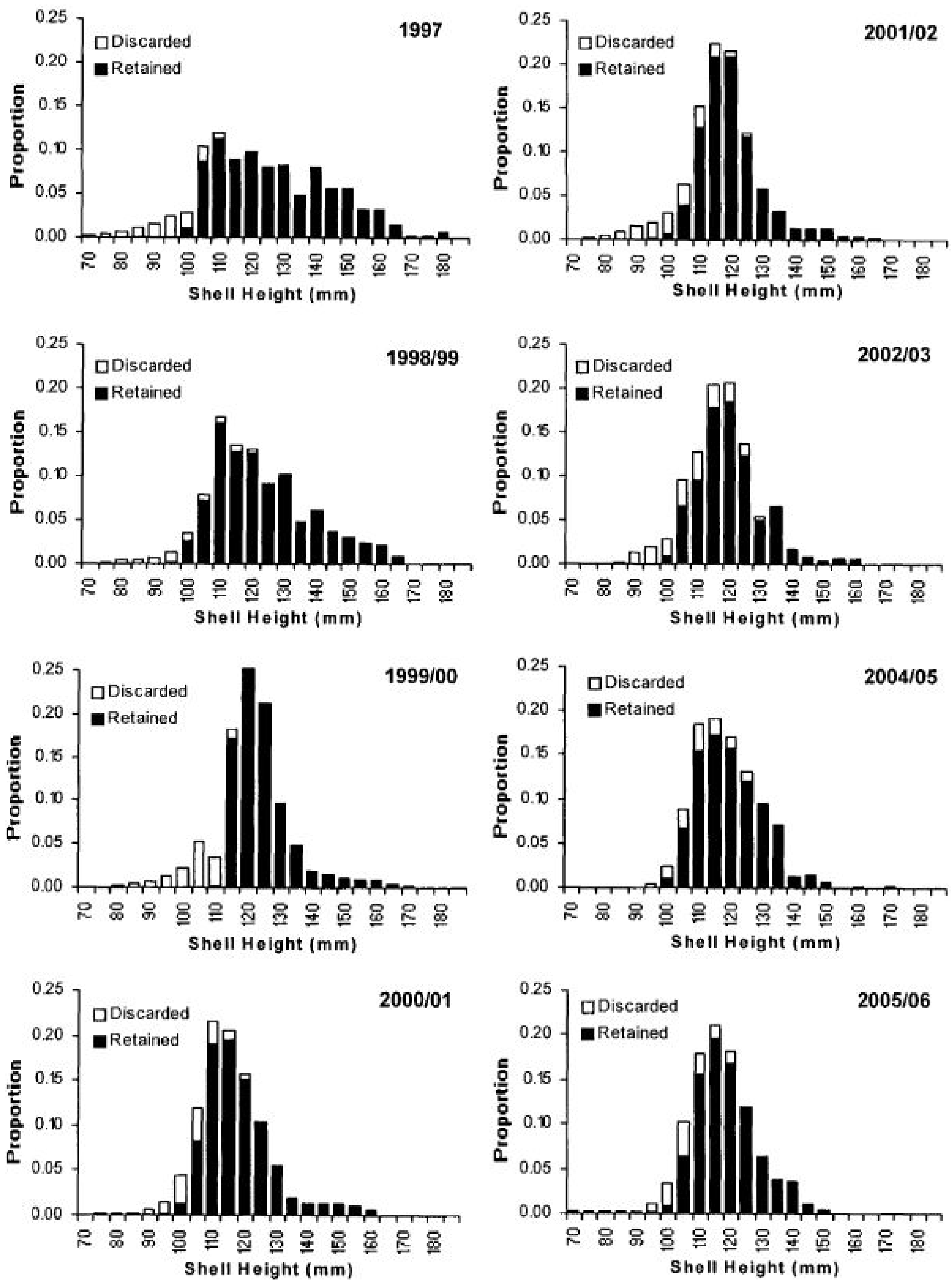

Figure 29. The records of shell height distributions from the reference period (earliest available data) to the most recent season on record showing shell height distributions for weathervane scallops harvested within District 16 (NPFMC 2015).

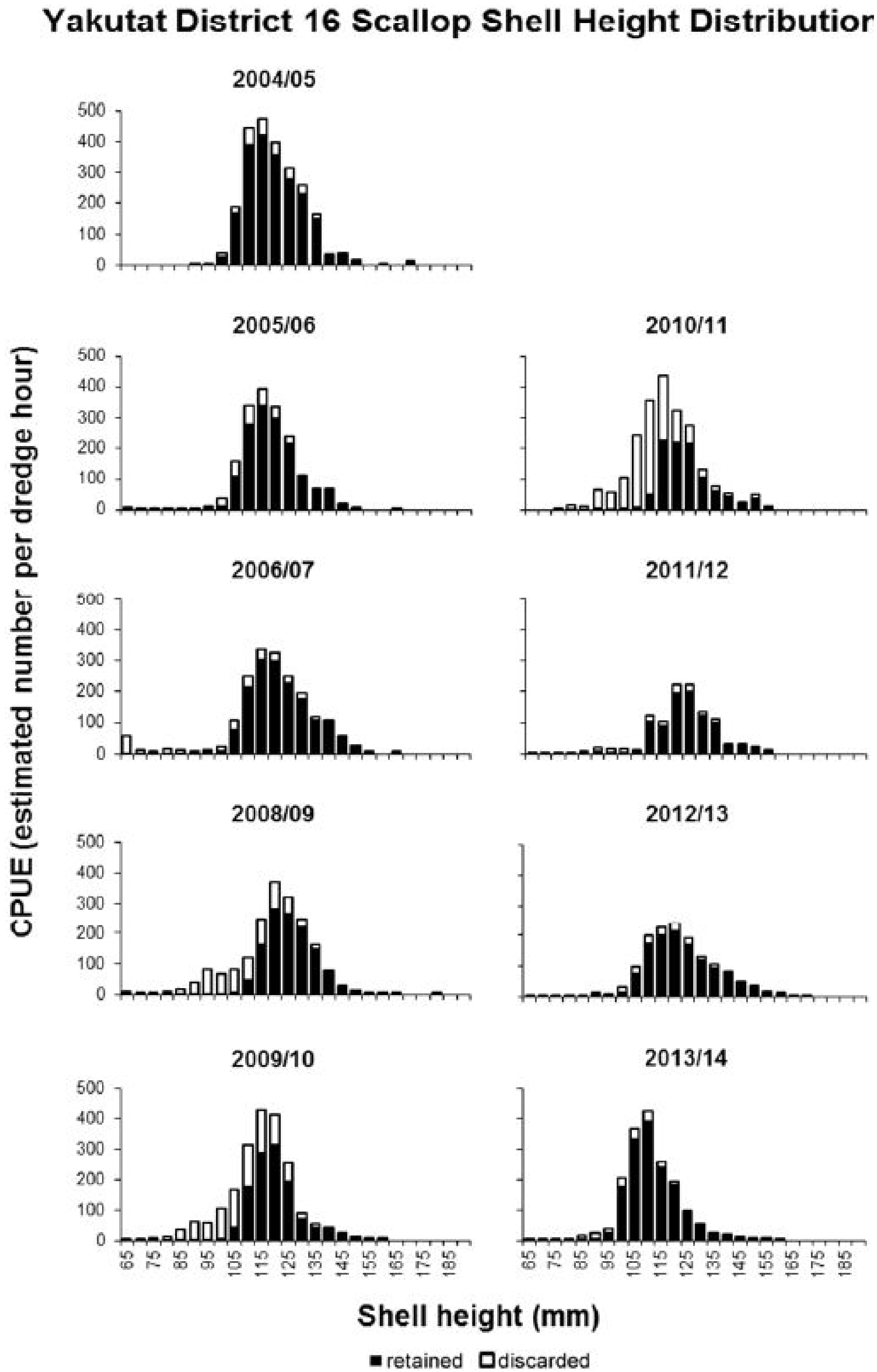

Figure 29. The records of shell height distributions from the reference period (earliest available data) to the most recent season on record showing shell height distributions for scallops harvested within District 16 (NPFMC 2015) (continued).

Scallop Health and Incidence of Weak Meated Scallop

Weak meated scallops do occur in Southeast Alaska, including the District 16 stock. The rate of occurrence of WMS in District 16 is considered a data gap for both the reference condition and the current condition. It is also unclear what proportion, if any, of the District 16 stock occurs within the GLBA boundary, but it is likely only a small portion of the whole area designated as District 16.

Scallop Habitat Status

Not much is known about the habitat status of weathervane scallops in district 16. This is particularly a concern considering the intensity of this fishery in terms of habitat disturbances, which is due to the direct contact of quite heavy fishing gear with the substrate. This certainly merits further investigation into the condition of scallop beds that could be protected under the park mandates. At this time the habitat status for weathervane scallop beds is a data gap.

Threats and Stressor Factors

The main concern for weathervane scallops in and around GLBA (location of beds is not clearly inside or outside the NPS boundary of GLBA) is impacts from harvest (e.g., damage to habitat or reduction in recruitment), the occurrence of WMS, and climate change.

The harvest impact on Tanner crab stocks is considered minimal and it is considered a stable fishery in Southeast Alaska. However, Bishop et al. (2013) has indicated that the Glacier Bay Tanner stock has suffered a significant decline. There have been fluctuations regarding the prevalence of BCS in some stocks and this merits continued monitoring and regulatory management to avoid exacerbation of this disease by way of discarding infected crabs in the area. The weathervane scallop stock in District 16 is showing signs of decline; observer data in District 16 identified a need for a 40% reduction in GHL in 2006 due to a sharp decline in CPUE over the previous three seasons (Table 17). The GHL was subsequently increased by 20% just prior to the 2009 season since the mean scallop size had increased as well as number of small scallops (NPFMC 2014).

Environmental factors may be impacting the Tanner crab and weathervane scallop stocks in Alaska. Climate change has brought on increasing variability in temperatures; scallop reproduction is suspected to be temperature triggered, although it is not clearly understood at this time (Morado 2007; NPFMC 2014). Temperature affects embryonic and larval development in scallops and changes in climatic regimes could impact recruitment in the District 16 scallop stocks (NPFMC 2014). Morado (2007) also notes the correspondence of BCS prevalence and distribution increasing with climate changes that are causing warmer average temperatures which has been linked to global coral-health declines and the spread of amphibian diseases.

There is some increasing concern with pH declines in the oceans as a consequence of climate change. This has implications for organisms that rely on calcium carbonate (i.e., exoskeleton of crabs and shells of bivalves) as a component of their biology (Soiseth, pers. communication, 2015). Much of the research pertaining to climate change in marine ecosystems has dealt with temperature variations (Harley et al. 2006); however, for invertebrate species that produce carbonate structures (e.g., clams, mussels, crabs), reduction of oceanic pH levels and the accompanied reduction in CO_2 levels could have pronounced impacts (Pörtner and Langenbuch 2005, Harley et al. 2006). There may be unforeseen impacts of climatic shifts that alter the velocity and direction of oceanic circulation in the Gulf of Alaska (Harley et al. 2006).The ocean current and chemistry is a physical, abiotic characteristic that may directly and/or indirectly influence the population dynamics of weathervane scallops in Alaska. How this will impact the population dynamics is difficult to predict when considering that scallop reproduction is poorly understood (NPFMC 2014).

Data Needs/Gaps

The habitat status for both Tanner crabs and weathervane scallops is considered a data gap. This is a particularly large data gap for the impact of the weathervane scallop fishery gear effect on benthic habitat and communities, which is considerably more impactful (dredging directly upon and through the substrate for long distances) than is Tanner crab fishing by way of stationary pots and ring nets, which impact habitat on a much more localized scale. There are no data available that directly addresses the status of habitat in the park for these two fisheries, although GBP is generally considered to be in fairly pristine condition. There is a gap in data for WMS occurrence or prevalence in District 16, as well as incidence of BCS in the Tanner crab stock within GBP.

Overall Condition: Tanner Crabs

Fishing Effort and Harvest

The Tanner crab stock fishing efforts measure was assigned a S*ignificance Level* of 2. The reference condition for this measure is the level of fishing effort in 1968. In the Southeast Alaska stocks fishing effort has been reduced since the 1990s as regulatory changes sought to reduce the harvests to ensure that the stocks can recover. Although the Tanner crab fishery is considered relatively stable, fishing effort has fallen in the last 5-10 years (Stratman et al. 2011). Due to this decline in fishing vessels, CPUE, and harvest numbers, the *Condition Level* has been assigned a 1, or of low concern.

Stock Abundance

The stock abundance of Tanner crabs was assigned a *Significance Level* of 3. The reference condition for this measure is the estimated biomass assessed for GBP and Excursion Inlet starting in 1999 and 1997, respectively (Bishop et al. 2013). Tanner crab fisheries in Alaska are considered relatively stable, and the stock abundance estimate in GBP increased to an all-time high in 2006 (Neilsen et al. 2007, Bishop et al. 2013). The Excursion Inlet stock decreased from 1997 through 2003, but subsequently stabilized for four successive years (2003-2007); this stock has since been increasing. As there are not long-term declining trends in the estimated abundance of GBP and Excursion Inlet stocks of Tanner crab, the *Condition Level* has been assigned a 1.

Health and Incidence of Disease

The measure addressing the health of Tanner crab stocks in Statistical Area A and the incidence of disease was assigned a *Significance Level* of 3. The reference condition is from 1986 when there were not any incidences of BCS detected in GBP, with a small percentage (5.6%) found in Excursion Inlet. More recent surveys (2001-2008) did find BCS in Tanner crabs in GBP as well as Excursion Inlet (Bednarski et al. 2010). The appearance of BCS in GBP merits a *Condition Level* of 2, or of moderate concern.

Overall Condition: Weathervane Scallops

Fishing Effort and Harvest

The weathervane scallop stock in District 16 was assigned a *Significance Level* of 3 for the fishing effort and harvest measure. The reference condition is the fishing effort and harvest (lbs of meat per dredge hour and retained catch in lbs) of 1994 which has since fluctuated drastically. The GHL for scallops was reduced to ease pressure on the stock in District 16 in response to significant drops in CPUE for that area (NPFMC 2014). In District 16, the past 2 years of recorded harvest catch in pounds of meat exceeded the 25,000 pound GHL by 255 and 510 lbs in 2012 and 2013, respectively. Because of both the decision to reduce GHL starting in 1999 and the unstable and hard to predict catch sizes, the *Condition Level* for fishing effort was assigned a 2, or of moderate concern.

Shell Height Distribution

Weathervane scallop shell size distribution was assigned a *Significance Level* of 2. The most recent scallop shell size distributions in District 16 show a decrease in size distribution and an increase in smaller scallops measuring <110 mm (<4.5 in) (NPFMC 2015). Declining shell sizes and heights is of great concern to park managers, especially over the longer term; however, this metric must be evaluated in the context of the species' life cycle (i.e., size at first reproduction). The *Condition Level* has been assigned a 3, or of high concern, due to declining shell height distribution.

Health and Incidence of Disease

The health and incidence of disease in weathervane scallops in District 16 was assigned a *Significance Level* of 2. This measure mainly addresses the incidence of WMS and is considered a data gap. Without data on the health of weathervane scallops in District 16 or other areas, it has been deemed a data gap and a *Condition Level* cannot be assigned.

Habitat Status

This measure was assigned a *Significance Level* of 3. The habitat status for District 16 is not currently known, or rather, hasn't been reported and is considered a data gap. GLBA staff is primarily concerned about the impact of the weathervane scallop fishery to the overall habitat of the species. While the impact is unknown, the fishing dredges typically have a substantial impact on the benthic habitat and the benthic community. Of the three marine harvest removal components in this NRCA (Anadromous Fish and Halibut representing the other two), GLBA staff has indicated that the current status of the weathervane scallop fishery/habitat is most concerning at this time (Soiseth, written communication, 4 October 2015). Despite a limited amount of data regarding this measure, professional judgment and opinion was used to determine a *Condition Level* of 3 for the weathervane scallop habitat status measure.

Weighted Condition Score (WCS)

Because of the unique life histories and reported statistics and metrics for the two priority harvested marine invertebrate species, the NRCA team (SMUMN GSS and GLBA staff) have determined that two unique *Weighted Condition Scores* should be assigned: one for the Tanner crab condition and one for the Weathervane scallop condition.

The Tanner crab *WCS* was determined to be 0.46, indicating that this resource is of moderate concern for stock in GLBA. A declining trend was assigned to this resource due to recent trends in the selected measures. However, due to the nature of GLBA having a future without harvest once lifetime permit holders have retired, the stock within GLBA will be out from under the pressures of commercial harvest. This may bring a unique opportunity for research to monitor this stock's recovery.

Tanner Crabs			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.46**
Fishing Effort and Harvest	2	1	
Estimated Stock Abundance	3	1	
Health and Incidence of Disease	3	2	

The Weathervane scallop *WCS* was determined to be 0.88, indicating a resource of high concern. Declines in harvest levels and degradations to shell height distributions, which may be a result of commercial harvest pressure, are currently of concern to park managers. However, despite limited amounts of data, the park is currently very concerned about potential degradations to scallop habitat quality. Additional monitoring of the habitat status is needed to verify these concerns. Due in part to the limited knowledge of the habitat status of weathervane scallops, a trend arrow was not assigned. Although a declining trend may be warranted for several of the measures, this trend would be conjecture for two of the four measures (habitat status and health and incidence of disease).

Weathervane Scallops			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.88**
Fishing Effort and Harvest	3	2	
Shell Height Distribution	2	3	
Health and Incidence of Disease	2	n/a	
Habitat Status	3	3	

4.3.6. Sources of Expertise

- Chad Soiseth, GLBA Fisheries Biologist

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4.4. Murrelets

4.4.1. Description

Glacier Bay represents a major population center for the marbled murrelet and Kittlitz's murrelet (Photo 14). The ancient murrelet (*Synthliboramphus antiquus*) occurs in the park infrequently (NPS 2015) and is not discussed in this component. The marbled murrelet is the most abundant and common of the three murrelet species in GLBA, and can be found across the southern coastline of Alaska, the western coast of British Columbia, and the western coast of the continental U.S. (Nelson 1997); the species can also be found along the coastline of far eastern Asia. The marbled murrelet was listed as a threatened species in the continental U.S. under the Endangered Species Act (Washington, Oregon, and California), and as threatened under the Canadian Species at Risk Act (Kirchhoff et al. 2010). The Alaskan population of marbled murrelets has not been listed federally, but the species is identified as a red watchlist species by Audubon Alaska, indicating the population in places may be vulnerable, declining, and depressed (Kirchhoff and Padula 2010). The estimated global population of marbled murrelets was 360,000 in 2010, with approximately 75% of that global population occurring in Alaska (Kirchhoff and Padula 2010).

Photo 14. A marbled murrelet (left; NPS photo) compared to a Kittlitz's murrelet swimming in Glacier Bay (right; photo used with permission by Richard Nelson).

The marbled murrelet is a small seabird (size is approximate to a dove) that spends the majority of its time at sea (Photo 15) (Ralph et al. 1995). The species' diet consists primarily of small fish and invertebrates, and feeding occurs in nearshore marine areas (Ralph et al. 1995). While the species will typically forage individually or in pairs, it is not uncommon to see large aggregations in areas of high prey abundance. Primary forage fish include Pacific sand lance (*Ammodytes hexapterus*), Pacific herring (*Clupea pallasii*), Pacific capelin (*Mallotus villosus socialis*), Pacific sandfish (*Trichodon trichodon*), euphasiids, amphipods, and small crustaceans (Vermeer et al. 1987, Sanger 1987, Day et al. 1999).

Photo 15. A group of both marbled and Kittlitz's murrelets in Glacier Bay (NPS photo).

Marbled murrelets are fast flying (Nelson 1997, Elliot et al. 2004), which makes identification in flight difficult. Further complicating visual identification is the species' plumage, which is virtually identical between males and females, mottled and marbled in appearance, and closely resembles the Kittlitz's murrelet (Hoekman et al. 2011a). The first nest location of the marbled murrelet was not documented until 1974 (Binford et al. 1975). The species appears to prefer to nest in the upper branches of old growth coniferous forests (Ralph et al. 1995). Unlike many seabirds, the marbled murrelet nests in solitary pairs or very loose aggregations (Ralph et al. 1995), and also inland, up to 75 km (46 mi) from the shore (Day et al. 1999).

The Kittlitz's murrelet is a less abundant species, and is endemic to Alaska and northeastern Asia (Day et al. 1999). The species was previously a candidate for Endangered Species protection by the U.S. Fish and Wildlife Service (USFWS) (USFWS 2010), but was removed from the list in recent years (USFWS 2013; 50 CFR Part 17). Similarly, the International Union for Conservation of Nature (IUCN) downlisted the species from critically endangered to near threatened in 2014, due in part to uncertainty regarding the rate of population decline (BirdLife International 2014). The Kittlitz's murrelet has also been identified as a red watchlist species by Audubon Alaska (Kirchhoff and Padula 2010). The most recent estimated minimum global population of Kittlitz's murrelets is 33,583 individuals, with a large proportion of this population (37%) being supported by Glacier Bay annually (USFWS 2013).

While similar in size and appearance to the marbled murrelet, the Kittlitz's murrelet can be distinguished during the summertime in GBP by white outer rectrices (tail feathers), smaller bill, mottled coloration, whiter flanks, and a nasal one-note “MWAH” call (Hoekman et al. 2013c). Foraging strategies and prey species overlap with the marbled murrelet, although in summer Kittlitz’s murrelets can sometimes be observed foraging off of glacial faces and in close proximity to glacial outflows (Day and Nigro 2000). The Pacific capelin, a common prey species for the Kittlitz’s murrelet, is sometimes more likely to occur near the face of tidewater glaciers compared to other areas of Glacier Bay (Arimitsu et al. 2008).

The Kittlitz’s murrelet is a cryptic nester and typically lays a single egg per clutch. Nests were initially believed to be restricted to sites near the tops of mountains, faces of cliffs, or near retreating glaciers (Day et al. 1999), but nests have since been found on the ground in the Aleutian Islands (e.g., areas of high acidity such as mats of lichens and mosses) (Kaler et al. 2009). Approximately 60% of the Kittlitz’s murrelet breeding and foraging habitat in Alaska is protected within or adjacent to National Parks and the National Wildlife Refuges (Walton et al. 2013).

GLBA represents a critical habitat area for both Kittlitz’s and marbled murrelets. The Kittlitz’s murrelet was selected as a Vital Sign in the SEAN, and has been monitored by the network annually since 2009. According to Sergeant et al. (2014, p. 1), the Kittlitz’s murrelet’s “…use of glacially-influenced habitats link this species to dynamic physical habitat conditions such as glacial extent and oceanography that are subject to chronic climate-induced changes.” Continued monitoring of both murrelet species will provide park managers with information on the population status of murrelets in the park and to some extent the health of the surrounding marine ecosystem these birds depend on.

4.4.2. Measures

- Abundance in surveyed areas
- Spatial distribution of Kittlitz’s murrelet within surveyed areas

4.4.3. Reference Conditions/Values

The reference condition for the murrelet component will be the average abundance estimates from the annual SEAN monitoring surveys in GLBA (2009-2015). While these averages serve as a basis for comparison, it needs to be noted that annual abundance estimates during SEAN monitoring are highly variable, with annual changes ranging from -56% to 120% each year. This variation is unlikely to reflect solely intrinsic population dynamics in GLBA, but the ecological mechanisms and potential survey biases impacting this variability are poorly understood at this time (Sergeant et al. 2014). The average abundance and maximum and minimum values observed for both species in GLBA from 2009-2015 are presented in Table 18.

Table 18. Minimum, maximum, and average abundance estimates for murrelet species in GLBA from 2009-2015.[A]

Abundance Estimate	Abundance[B]	
	Kittlitz's Murrelet	Marbled Murrelet
Minimum	7,210	28,978
Maximum	16,469	84,428
Average	11,255	60,959

[A] Data obtained from Sergeant et al. (2015), which is the most recent annual monitoring report for murrelets in GLBA.

[B] Abundance estimates were extrapolated over 1,170 km^2 of sampled waters with the exception of 2009 which was extrapolated over 1,092 km^2 of sampled waters.

4.4.4. Data and Methods

Previous surveys of murrelets in GLBA, and across their range in general, have utilized different sampling designs with varied objectives (Kirchhoff 2011). Bay-wide surveys in Glacier Bay were conducted during many years from 1987-2010 and each utilized unique survey design and objectives, making comparisons between years and studies difficult. The following assessment will focus on the most recent murrelet survey efforts completed by the SEAN. Appendix B is reproduced from Kirchhoff et al. (2011) and represents a summary of the surveys that have occurred in GLBA from 1987-2010. Many of the data are presented in Appendix B for reference only, and will not be utilized in the overall condition assessment.

While this assessment focuses on current (2009-2015) estimates of abundance and density for murrelets in GLBA, Kirchhoff et al. (2010) also provides valuable information regarding the historic status of murrelets in GLBA. This study provided insight and potential explanations for the perceived decline in murrelet abundance in Southeast Alaska since 1991. Kirchhoff et al. (2010) repeated a 1993 GLBA survey to develop a greater understanding of how distribution and abundance patterns may shift over time. The 1993 survey was chosen mainly because it was the first survey to use straight line transects with starting and ending points that were known, so it could be easily replicated (Kirchhoff et al. 2010). The study concluded that Kittlitz's murrelet populations have been relatively steady since 1993, and these findings are generally in agreement with the most recent murrelet monitoring conducted by the SEAN.

As part of a long-term monitoring effort, the SEAN has monitored both marbled and Kittlitz's murrelets in 1,170 km^2 (452 mi^2) of GLBA waters annually since 2009 and will continue indefinitely (Hoekman et al. 2011a, 2011b, Hoekman et al. 2013a, 2013b, Hoekman et al. 2014, Sergeant et al. 2014, 2015). Surveys are conducted primarily in early July, and utilize a generalized random tessellation stratified sampling design (GRTS; Stevens and Olsen 2004) in order to provide a random sample that is still spatially-balanced throughout GBP. Line transects are oriented perpendicular to the prevailing shorelines in order to avoid instances where transects may be parallel to the observed density gradient of murrelets as observed in Drew et al. (2008) and Kirchhoff et al. (2011). In areas of the Bay where waters are more restricted (e.g., Rendu Inlet, Reid Inlet in the West Arm) a zig zag pattern is used. Observers record murrelet group size, species (marbled, Kittlitz's, or unidentified), and an estimate of distance and angle in a straight line forward from the bow of the survey vessel (Sergeant et al. 2014).

4.4.5. Current Condition and Trend

<u>Abundance in Surveyed Areas</u>

Kirchhoff et al. (2010)

Using similar methodologies across years, marbled murrelet abundance in 2009 was estimated at 25,288 individuals, which was a decrease of approximately 5,000 individuals when compared to 1993 (Table 19). The estimated global population of marbled murrelets was approximately 360,000 individuals in 2010 (Kirchhoff and Padula 2010). The estimated population in GLBA in 2009 comprised approximately seven percent of the global population.

Table 19. Observed, on-water density and extrapolated population size (abundance) of both marbled and Kittlitz's murrelets as observed during Lindell's (2005) 1993 survey of GLBA and Kirchhoff et al. (2010).

Species	Year	Population Size (Abundance)
Marbled Murrelet	1993	30,042
	2009	25,288
Kittlitz's Murrelet	1993	2,657
	2009	4,570

Kittlitz’s murrelet abundance in 2009 was approximately 4,570 individuals, which was an increase of approximately 2,000 individuals when compared to 1993 (Table 19). The most recently estimated minimum global population of Kittlitz’s murrelets was 33,583 individuals (USFWS 2013), and the approximate GLBA population in 2009 comprised nearly 14% of the global minimum population of this species.

SEAN Monitoring of Murrelet Species in GLBA (2009-present)

Murrelet abundance was estimated during SEAN monitoring by extrapolating the observed on-water density of murrelets (birds/km^2) over the total study area (1,170 km^2 [452 mi^2]). In 2009, which was the pilot year of the long-term monitoring project, the total study area was slightly smaller (1,092 km^2 [422 mi^2]) (Hoekman et al. 2011a).

The estimated abundance of marbled murrelets observed in GLBA during SEAN monitoring has been highly variable, ranging from 28,978 (2009) to 84,428 (2013) individuals (Table 20) and with annual change ranging from -51% to 120% (Sergeant et al. 2015). The average estimate of abundance for marbled murrelets in GLBA from 2009-2015 was 60,959 individuals per year (Table 18). Marbled murrelet abundance in 2015 was 83,793 individuals, which was the second largest estimate on record (Table 20); abundance in 2015 was above the 7-year average abundance in GLBA (Table 18, Table 20).

Table 20. Estimates of on-water abundance of Kittlitz's and marbled murrelets in GLBA during SEAN monitoring efforts from 2009-2014 (Sergeant et al. 2014).

Kittlitz's Murrelet			Marbled Murrelet	
Year	Abundance[A]	SE[C]	Abundance	SE[C]
2009[B]	13,124	4,062	28,978	4,077
2010	13,308	1,357	61,717	5,372
2011	7,477	1,119	73,766	7,055
2012	16,469	2,581	52,560	5,216
2013	7,210	2,046	84,428	15,394
2014	10,422	1,522	41,474	3,998
2015	10,778	2,598	83,793	12,044

[A] Abundance was projected across surveyed waters only.

[B] Note that the pilot surveys in 2009 differed in survey area (1,092 km^2) and methods (Hoekman et al. 2011a).

[C] Standard Error of the abundance.

Kittlitz's murrelet abundance estimates have also been highly variable during SEAN monitoring, with annual estimates ranging from 7,210 (2013) to 16,469 (2012) individuals (Table 20) (Sergeant et al. 2015). The average abundance estimate for Kittlitz's murrelets in GLBA from 2009-2015 was 11,255 individuals (Table 18). Kittlitz's murrelet abundance in 2015 was estimated at 10,422 individuals, which was near the abundance estimate from 2014, and slightly below the 6-year average (Table 18, Table 20).

Spatial Distribution of Kittlitz's Murrelets within Surveyed Areas

Spatial distribution of Kittlitz's murrelets in GLBA has shown substantial annual variation during SEAN surveys from 2009-2015. However, spatial distributions are only qualitatively described at this point, and there are not any statistics to report at this time. Kittlitz's murrelets have been most prevalent in areas near small fjords and narrower portions of the upper West arm and the east side of GBP (Beardslee entrance to upper East arm) (Figure 30) (Sergeant et al. 2015). Sergeant et al. (2015) notes that Kittlitz's murrelet concentrations are most consistent in the Reid Inlet, which has been the only tidewater glacier location where murrelets have been observed in consistently high numbers. Aggregations of Kittlitz's murrelets also occur intermittently at other hotspots in GBP, such as the Hugh Miller-Scidmore Complex (documented aggregations from 2000-2002 and from 2010-2013), the Beardslee Entrance, near Russell Island, and the upper east side of the main bay.

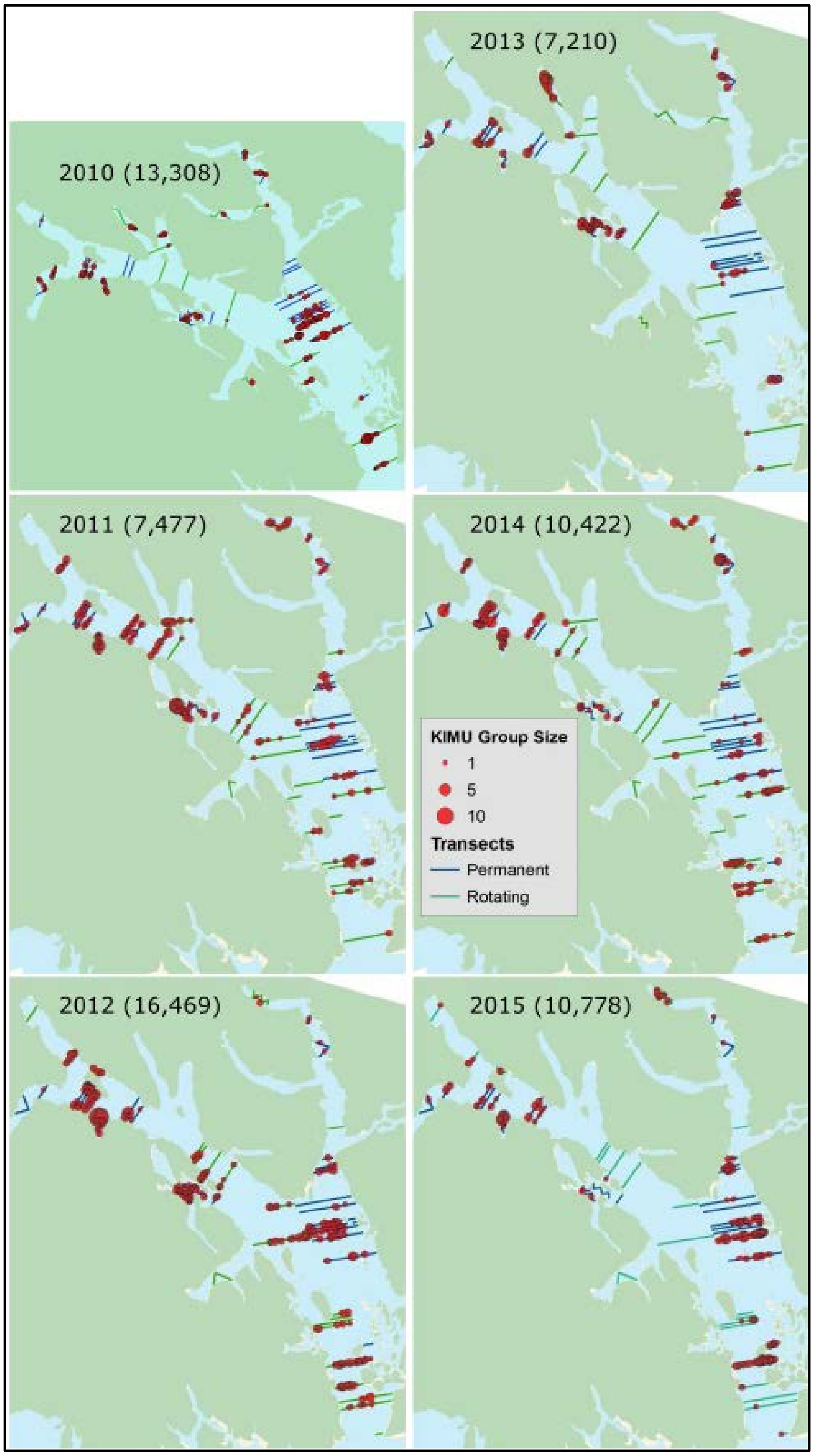

Figure 30. Comparison of Kittlitz's murrelet spatial distributions from 2010-2015 in GBP. Maps are labeled with survey year and abundance estimate in parentheses. 2010 and 2013 represent Panel 1 of the 3-year rotation panel design, 2011 and 2014 represent Panel 2, and 2012 and 2015 represent Panel 3. Due to the use of a different data collection tool, the 2010 map appears in a slightly different format (Sergeant et al. 2015).

Evidence exists that breeding populations of Kittlitz's murrelets throughout their range are closely linked to tidewater glaciers (Kuletz et al. 2003, Arimitsu et al. 2011, Kissling et al. 2011); however, outside of Reid Inlet in GLBA, Kittlitz's murrelets use of tidewater glaciers has been moderate to low (Sergeant et al. 2015). Breeding aggregations of murrelets have been observed in recent years in areas far from glacially-influence habitats (Sergeant et al. 2015). These observations may suggest that the species has broader habitat affinities than previously thought and additional research into Kittlitz's murrelets distribution in GLBA is warranted. Future assessments of murrelet distribution should investigate the relationships that prey availability, temperature and salinity gradients, bathymetry, tidal currents and proximity to nesting habitat have on murrelet distribution (Sergeant et al. 2015).

Threats and Stressor Factors

GLBA staff identified several threats and stressors to the murrelet community in the park, including: altered habitats resulting from climate change, varied availability of forage species, and disturbances from marine vessels. Species that are linked to ice-associated habitats during a specific stage of their life history, such as the breeding season of the Kittlitz's murrelet, are typically the first to respond to climate change (Walther et al. 2002, Root et al. 2003, Kuletz et al. 2003). In GLBA, rapid glacial recession and changes in the number of tidewater glaciers may positively and/or negatively impact breeding and foraging habitats for murrelet species, especially during the summer (Kuletz et al. 2003, Piatt et al. 2011), but existing data are unable to resolve these questions.

Food availability for murrelets is likely sensitive to temporal variations in climate (Saether 2000), which can shift the timing of peak food resource availability (Becker et al. 2007). A mismatch between reproductive timing and peak prey availability could lead to a reduction in population productivity (Thomas et al. 2001, Becker et al. 2007). Murrelet species are adaptable in regards to focal forage species, and typically will shift to forage species that are most abundant or energetically valuable (Becker et al. 2007). This adaptability was evident in the late 1970s when murrelets altered their diets range-wide following a major climatic shift in the North Pacific (Francis et al. 1998, Anderson and Piatt 1999, Piatt et al. 2011). Murrelets have been shown to be most productive during periods of lower oceanic temperatures (e.g., the 1998-1999 PDO Index shift to cooler temperatures), likely due to more productive oceanic conditions and an increased number of forage fish (Becker et al. 2007).

The magnitude and persistence of the perceived population declines in murrelet species is uncertain and is the subject of much debate (Kirchhoff et al. 2010, Piatt et al. 2011, Kirchhoff et al. 2014, Marcella 2014), and many factors have been described as potential contributing sources to the apparent regional decline (low recruitment: Kaler et al. 2009, USFWS 2011; direct mortality due to oil spills: van Vliet and McAllister 1994, Piatt and Anderson 1996, Kuletz et al. 2003; and disturbance from vessels: Day et al. 1999, 2003, Agness et al. 2008, 2013, USFWS 2011). In GLBA, continued summertime abundance monitoring will help clarify these species' population status and trend.

Other research in GLBA has focused on cruise ship disturbance to murrelets. In an effort to protect the park's critical glacially influenced ecosystems, the park finalized an EIS and ROD in 2003 (NPS

2003). Part of this EIS established a quota for vessels operating in the park. The maximum number of cruise ships allow to enter GBP during the peak season (June-August) is 153 cruise ships, with no more than two cruise ships permitted to enter on a given day.

Disturbance to murrelets by vessel traffic in GLBA has been studied in the past by Agness et al. (2008, 2013) and Marcella (2014). Both studies have documented that breeding and non-breeding murrelets are disturbed by vessels. Agness et al. (2013) found that Kittlitz's murrelets were more likely to fly away from vessels compared to marbled murrelets, which results in significant energy expansion. Added energy requirements resulting from repeated disturbances could impact the population dynamics of murrelets by lowering reproductive success and or survival, although these fitness-related variables were not recorded during the study. Marcella (2014) found that up to 72% of murrelets passing within 850 m (2788 ft) of a cruise ship were disturbed (dove or flew in response) although murrelets were also disturbed when approached by ships at distances >1000 m (>3280 ft).

Data Needs/Gaps

Murrelets in GLBA have been well-studied over the past few decades. Multiple projects have used unique survey methodologies, which makes comparison between efforts and years difficult. This is especially evident when trying to make sense of the perceived decline in murrelets in the GLBA region during the 1990s. Current SEAN monitoring efforts are long-term (2009-present) and have utilized the same methodology each year since 2010. Continuation of these efforts will allow for accurate comparisons between survey years and can serve as a baseline for future comparisons.

Murrelet diets and preferred forage species within GLBA are poorly understood. This relationship will be especially important as climate change modifies oceanographic dynamics such as temperature regimes and primary productivity, potentially shifting the abundance, distribution, and community composition of murrelet prey species. Shifts in prey species could result in additional direct competition for resources with other top predators in the Bay (e.g., Steller sea lions). Continued research regarding vessel disturbance and habitat use within GLBA is needed in order to more accurately determine the level of impact that vessel traffic may have on nesting and non-breeding birds.

Because the Kittlitz's murrelet is a cryptic, solitary nester, it is difficult to fully understand the annual reproductive success of the species. Additional investigations into this species' nesting requirements in the park may shed some light on particular areas of high importance, or high priority resources in the marine and terrestrial ecosystems.

Overall Condition

Abundance in Surveyed Areas

During project scoping, the abundance in surveyed areas measure was assigned a *Significance Level* of 3. Murrelet species have long been assumed to have undergone a major population decline in the 1990s. During Kirchoff et al.'s (2010) replication of a 1993 survey, the authors found no evidence of either murrelet species undergoing a large population decline in the GLBA area after 1993. This is in contrast to Piatt et al. (2011), who estimated that the Kittlitz's murrelet declined in GLBA by 85-95%

between 1991 and 2008. As mentioned previously, the status of the murrelet species during the late 1990s and early 2000s may never be resolved due to differences in sampling methodology.

During annual SEAN monitoring in GLBA, the 2014 and 2015 abundance estimates for Kittlitz's murrelets were similar, with 10,422 and 10,778 individuals, respectively. These estimates were slightly below the 7-year average of 11,225 individuals, but still represented a 45% (2014) and 49% (2015) increase when compared to 2013 (Table 20). Marbled murrelet abundance was estimated at 83,793 individuals, which was the second highest abundance estimate on record and fell well above the 7-year average of 60,959 individuals (Table 18, Table 20).

The abundance estimates obtained for both species throughout the duration of the SEAN monitoring have been highly variable, with annual changes ranging from -56% to 120% for Kittlitz's murrelets and from -51% to 113% for marbled murrelets (Sergeant et al. 2014). Based on the life history traits of both murrelet species, the variations and changes in abundance between years appear to be larger than what could plausibly be attributed to only the GLBA murrelet population. This suggests that a good deal of immigration and emigration may be occurring in GLBA waters on an annual basis (Day et al. 1999; USFWS 2013, Sergeant et al. 2014). The current condition of murrelet abundance was assigned a *Condition Level* of 1, indicating low concern.

Spatial Distribution of Kittlitz's Murrelets within Surveyed Areas

The spatial distribution of Kittlitz's murrelets within selected areas of GLBA was assigned a *Significance Level* of 3. While there are only qualitative discussions regarding this measure, there does not appear to be cause for substantial concern in GLBA. Monitoring results from 2010-2015 have shown that murrelets are well distributed across GBP, with frequent substantial aggregations found in Reid Inlet and in areas near small fjords and narrower portions of the upper West arm and the east side of GBP (Figure 30). Additional research is still needed to determine why distribution patterns shift annually and an increased understanding of the species' dependence (or lack of) on tidewater glaciers is needed. A *Condition Level* of 1 was assigned to this measure.

Weighted Condition Score

The *Weighted Condition Score* for murrelets was determined to be 0.33, which is indicative of good condition. A trend arrow was not assigned to this component for a few reasons. First, there have been high levels of annual variation observed for both the reported abundance and the reported density measures during SEAN monitoring. These high annual shifts make determining a realistic trend difficult. Second, there is considerable uncertainty regarding the actual population size of these species, especially in regards to whether or not there was a substantial decline in Southeast Alaska and across their ranges. Continued monitoring of both species in the park will provide managers with a better approximation of trend as more years of data become available.

Murrelets			
Measures	Significance Level	Condition Level	WCS = 0.33
Abundance in Surveyed Areas	3	1	
Spatial Distribution of Kittlitz's Murrelets in Surveyed Areas	3	1	

4.4.6. Sources of Expertise

- Chris Sergeant, SEAN Ecologist

4.4.7. Literature Cited

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4.5. Sea Ducks

4.5.1. Description

Sea ducks belong to the waterfowl Tribe *Mergini* (Anatidae subfamily Merginae). GLBA is home to 13 of the 18 extant sea duck species throughout the year, although the hooded merganser (*Lophodytes cucullatus*), common eider (*Somateria mollissima*), and king eider (*Somateria spectabilis*) are considered rare visitors to the park (NPS 2015; Dan Esler, Research Wildlife Biologist, USGS Alaska Science Center, written communication, 16 January 2015). Sea ducks within GLBA have been sporadically surveyed during the last few decades, and more research is needed to quantify distribution, abundance, and ecology within GLBA. Despite the lack of current abundance estimates, it is known that the park (and southeast Alaska as a whole) provides critical habitat for wintering and molting sea duck species (Bodkin et al. 2001); there are three species for which GLBA is globally important: Barrow's goldeneye, harlequin duck (*Histrionicus histrionicus*), and surf scoter (*Melanitta perspicillata*). Substantial proportions of the global population of these three species are found within GLBA waters (Esler, written communication, 1 October 2015).

Sea ducks are primarily diving species, and have diets that consist mainly of shellfish, mollusks, small aquatic invertebrates, and fish (Johnsgard 1978). Sea duck species feature several anatomical adaptations to survive in coastal waters. Sea ducks have legs that are located near the rear of the body and also have very large feet (Johnsgard 1978). These adaptations allow for deep, powerful dives, but sacrifice the species' ability to maneuver effectively on land. Several species have specialized bill shapes that allow for more efficient prey capture; the surf scoter and black scoters (*Melanitta americana*, hereinafter collectively referred to as scoters; Photo 17) have large, angled bills that allow for shellfish capture, while the common merganser (*Mergus merganser*; Photo 16) has an elongated, thin bill that contains hundreds of ridges along it that allow for the grasping of fish.

Photo 16. Common merganser; note the long, thin bill used for catching fish (NPS photo).

Photo 17. Surf scoter in GLBA; notice the large bill with the slightly angled tip that is used to pry shellfish from rocks (NPS photo).

Sea ducks are found within GLBA throughout the year, although few sea ducks breed within the park. Use of GLBA is largely for nonbreeding stages of the annual cycle, particularly wing molt, wintering, and migration. Sea ducks represent an abundant part of the marine bird community; in fact, sea ducks constituted between 47% and 56% of all birds observed during surveys in March, June, and November from 1999-2001 (Robards et al. 2003).

Sea ducks, like other waterfowl, molt their flight feathers simultaneously, usually after the breeding season, and experience a prolonged period of flightlessness (scoter feather growth takes >40 days; Dickson et al. 2012, Uher-Koch et al. 2014). Often sea duck species will migrate to specific areas before the molt takes place; these areas tend to have high prey availability and low predator density (Salomonsen 1968, Savard et al. 2007, Uher-Koch et al. 2014). The molt-ecology of sea ducks is poorly understood. Recent research indicated that scoter mortality during molt was low (Uher-Koch et al. 2014), and that the molt period was protracted over many months across all of the age and sex cohorts (Dickson et al. 2012). The sheltered coastal habitat of GLBA is home to large congregations of molting harlequin ducks, surf scoters, and white-winged scoters (*Melanitta deglandi*), and additional research is needed to determine the distribution, abundance, mortality, and ecology of these species during this annual cycle stage.

4.5.2. Measures

- Species richness
- Abundance of molting harlequin ducks, surf scoters, and white-winged scoters
- Abundance of wintering Barrow’s goldeneyes, harlequin ducks, and surf scoters
- Winter Age Ratio of Barrow’s goldeneyes, harlequin ducks, and surf scoters

- Foraging effort of molting and wintering sea ducks (Barrow's goldeneyes, harlequin ducks, surf scoters, and white-winged scoters)

4.5.3. Reference Conditions/Values

A reference condition for the sea duck community in GLBA does not exist at this time. There are few data quantifying sea duck abundance, distribution, or performance in Glacier Bay, and any historical resource conditions would be speculative at best.

4.5.4. Data and Methods

Duncan and Climo (1991a) reported results from a 1991 survey of the birds and mammals of the Beardslee Islands in GLBA. Ten surveys were conducted in the Beardslee Islands between 4 June and 20 September 1991; each survey sampled between five and16 transects. Observers reported all bird and mammal species observed within 100 m (328 ft) on either side of the skiff, within 50 m (164 ft) straight ahead of the skiff, and within 100 m (328 ft) above the skiff (Duncan and Climo 1991a). This study was the fourth year of a survey of this area; however, only data for 1991 are available in this report. Data regarding the 1987-1990 surveys are only depicted graphically in Duncan and Climo (1991a), and without the data/discussion of those results, interpretation of those surveys would be inaccurate. This assessment summarizes only the 1991 survey results.

Piatt et al. (1991) investigated marine bird distribution in GLBA between 13 June and 15 July 1991. The primary objective of the study was to document murrelet population and distribution throughout the Bay; however, surveys also recorded the number of sea ducks (among other marine predators) that were encountered on transects. Surveys followed USFWS protocols at the time, and used a 4.9 m (16 ft) skiff to survey 166 transects in the Bay. Two observers recorded all species observed within 200 m (656 ft) on either side of the boat. Birds observed outside of the 200 m (656 ft) buffer also were recorded when possible.

Robards et al. (2003) summarized results of USGS marine predator surveys in 1999 and 2000, while Bodkin et al. (2001) summarized results of 2001 surveys. The objective of these surveys was to quantify distribution and abundance of marine predators (including sea ducks) in Glacier Bay. Vessel surveys were used to survey all coastline areas of GLBA in June 1999, 2000, and 2001, with several pelagic surveys also being conducted during this time (transects were spaced 2.5 km [1.5 mi] apart). A subset of June surveys (approximately 30% of the transects) were resampled in November 1999, and March 2000 and 2001 to investigate seasonal patterns of use in the Bay (Robards et al. 2003). Surveys counted all birds within 150 m (492 ft) of either side of the larger boats, and within 100 m (328 ft) of the smaller boats. Birds within 300 m (984 ft) of the front of both sized boats also were recorded; birds flying above transects also were counted.

There have been several brief surveys and censuses that focused exclusively on scoters in the Hugh Miller complex of Glacier Bay. Carter (1984) represents one of the earliest scoter-specific censuses to take place in the park; this census focused on obtaining a population estimate of molting scoters in the Bay. Surveys took place from 1-3 August, and again on 19 August 1984; census data from 1 August were considered an accurate representation of scoters in the study area, and subsequent surveys were hampered by inclement weather. Censuses were taken predominantly from shore, and observation points were opportunistic (i.e., a census took place when the observers found a raft of scoters in a bay while they kayaked). Priority areas that were censused included High Miller Inlet, Charpentier Inlet, and Scidmore Bay. Observers recorded location, number of birds, species of birds, and whether or not the birds were molting.

Fister and Widdice (1989) surveyed Scidmore Bay, Weird Bay, and Charpentier Inlet for Canada geese (*Branta canadensis*), scoters, mergansers, harlequin ducks, and other species from 14-15 July 1989. Surveys were conducted from kayaks and from shore; smaller groups of birds were counted from the water using 8X binoculars, while larger groups were counted from shore using a high powered spotting scope. Data recorded included species, number of individuals, location, species composition of scoter groups (i.e., percentage of white-winged vs. surf), and the percentage of birds molting.

The last of the scoter surveys in the late 1980s and early 1990s took place in 1991 when Duncan and Climo (1991b) surveyed the Hugh Miller complex. Surveys took place from 6-7 August 1991, and were conducted primarily from shore. Two observers used a tandem kayak to access the study area (Weird Bay, Hugh Miller Inlet, Charpentier Inlets, and Blue Mouse Cove), and when a large raft of scoters was observed, the kayak was brought to shore and the birds were counted with the aid of binoculars and spotting scopes. Strong winds prevented access to Scidmore Bay, and only the entrance of the bay was surveyed.

Gustavus, AK is located just outside of GLBA and has been the site of an annual Christmas bird count since 1968. The CBC conducted in Gustavus is part of the International Christmas Bird Count (CBC), which started in 1900 and is coordinated by the Audubon Society. Data from the Gustavus CBC are available for 39 years: 1968-71, 1973-75, 1977, 1979-83, 1985-87, 1989, 1992-1998, and 2000-2014.

Multiple volunteers surveyed a 24-km (15-mi) diameter area on one day, typically between 14 December and 5 January, by foot, boat, and car. The center point of the 24-km diameter was 58.450595ºN, -135.872083ºW (Figure 31). Unlike surveys that occur during the breeding season (such as the North American Breeding Bird Survey [BBS]), the CBC surveys overwintering and resident birds that are not territorial and singing. The total number of species and individuals were recorded each year; data from the Gustavus CBC have been modified for this assessment to only include sea bird species (previously defined above). Species records for oldsquaw and long-tailed duck (*Clangula hyemalis*) were combined and treated as a single species in accordance with AOU (2000).

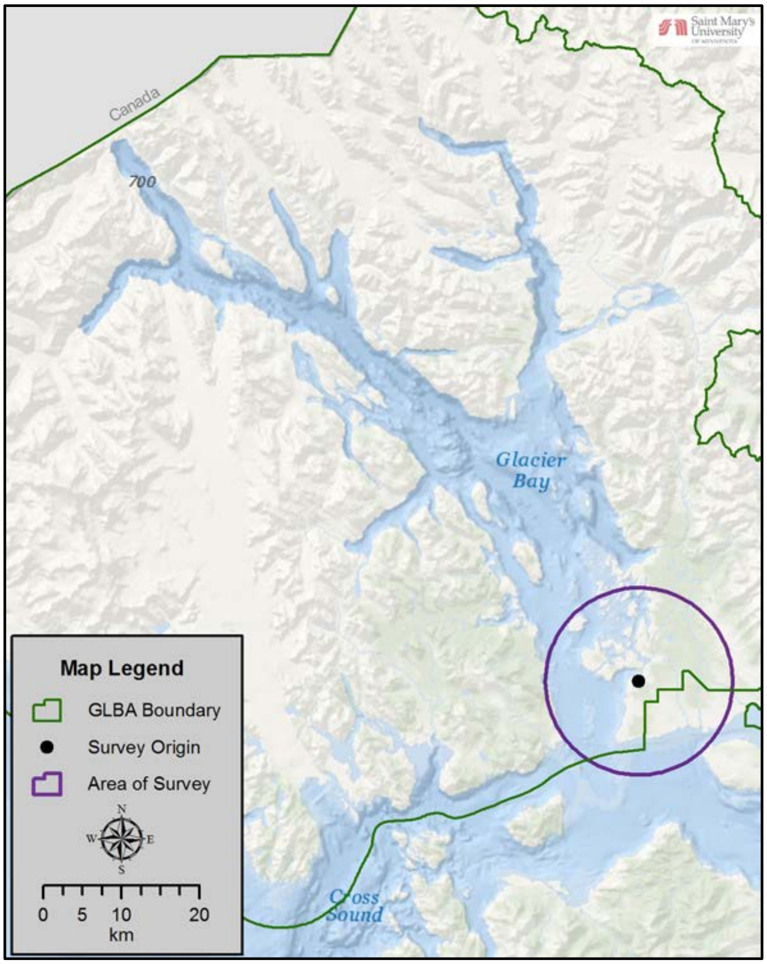

Figure 31. Center point and coverage area of the Gustavus, AK Christmas Bird Count.

4.5.5. Current Condition and Trend

Species Richness

The species richness measure can indicate overall habitat suitability for breeding, wintering, and molting sea ducks. Species richness values are unlikely to change in regards to the park's sea duck community without extreme changes to the environmental conditions found within the park (Esler, written communication, 8 May 2014).

Sea duck species identification is often problematic, as surveys typically occur in difficult observation conditions (i.e., wave action, glare, observation distances >100 m [>328 ft]). Observations are further complicated when identification requires use of high-magnification spotting scopes or binoculars. In many surveys, if a sea duck could not be identified to a species level, it was identified to the genus level. This assessment summarizes species richness using only species that have been identified to the species level; instances of unidentified species (i.e., unidentified merganser species) are noted in the text and tables/figures below.

NPS Certified Species List (NPS 2015)

The NPS Certified Bird Species List contains 257 species, 13 (5%) of which are sea duck species (Table 21). This list, however, does not allow for a specific annual analysis of species richness, as no data are collected yearly. The list simply represents a compilation of all species identified in the park by the various studies that have occurred. The king eider is listed as a confirmed species by NPS (2015), although this species was not observed during any of the surveys summarized in this report. It is likely that this species, along with the hooded merganser, and common eider are rare visitors to the park.

Table 21. Sea duck species identified on the NPS Certified Species List (NPS 2015) as well as species identified during surveys and counts in GLBA from 1968-2001.

Common Name	NPS (2015)	Duncan and Climo (1991a)	Piatt et al. (1991)	Bodkin et al. (2001)	Robards et al. (2003)	CBC (1968-2014)
bufflehead	X			X	X	X
common goldeneye	X			X	X	X
Barrow's goldeneye	X		X	X	X	X
oldsquaw (long-tailed duck)	X	X	X	X	X	X
harlequin duck	X	X	X	X	X	X
hooded merganser	X					X
white-winged scoter	X	X	X	X	X	X
black scoter	X			X	X	X
surf scoter	X	X	X	X	X	X
common merganser	X	X	X	X	X	X
red-breasted merganser	X		X	X	X	X
common eider	X					X
king eider	X					
Unidentified Scoter sp.		X	X	X	X	X
Unidentified Merganser sp.		X	X		X	X
Unidentified Goldeneye sp.		X	X	X	X	X
Species Richness	**13**	**5**	**7**	**10**	**10**	**12**
Species Richness Including Unidentified Sp.	**13**	**8**	**10**	**12**	**12**	**15**

Duncan and Climo (1991a)

Duncan and Climo (1991a) observed five sea duck species during 1991 surveys in the Beardslee Islands. Three unidentified species were recorded during the surveys (unidentified scoter sp., merganser sp., and goldeneye sp.) (Table 21).

Piatt et al. (1991)

During a short (1 month) marine bird distribution survey, during the summer of 1991 in GLBA, Piatt et al. (1991) documented seven sea ducks species in the park (Table 21). Unidentified scoter, merganser, and goldeneye species also were observed during the survey.

USGS Marine Predator Surveys (Robards et al. 2003; Bodkin et al. 2001)

Ten sea duck species were observed in GLBA from 1999-2000 (Robards et al. 2003), while 10 species were observed in 2001 (Bodkin et al. 2001; Table 21). The ten species observed during the USGS marine predator surveys represent 77% of all sea duck species confirmed in GLBA (only the hooded merganser, common eider, and king eider were not observed). Unidentified scoter, merganser, and goldeneye species also were reported during USGS studies.

Christmas Bird Count (1968-2014)

From 1968-2014, 12 species of sea ducks were observed during Gustavus' annual CBC (Table 21). Twice in the history of this count there have been 12 species observed in a single count year (1977, 2001). The lowest sea duck species richness value during the CBC has been nine species, which has been reported four times since 1968. The only species from NPS (2015) that has not been observed during the CBC is the king eider (Table 21).

Sea duck species richness values in GLBA have ranged from five species (Duncan and Climo 1991a) to 12 (CBC 1968-2014) (Table 21). Long-tailed ducks, harlequin ducks, white-winged scoters, and common mergansers were observed in all of the surveys summarized in this document (Table 21). The timing of surveys is likely important when discussing species richness, as several species that winter or molt in GLBA may be missed during surveys conducted during early summer. For example, even though hooded mergansers are a rare species in the park, they were only observed during winter CBCs (albeit in low numbers). Most of the surveys that have taken place in the park have occurred in June and July, although the USGS marine predator surveys also sampled in March and November. Differences in sampling/surveying methodology (e.g., location, timing, coverage) may account for the variations observed in species richness; there does not appear to be any cause for concern regarding sea duck species richness in GLBA at this time.

<u>Abundance of Molting Harlequin Ducks, Surf Scoters, and White-winged Scoters</u>

Sea ducks typically migrate to remote locations following the breeding season to molt. These molting habitats are usually in areas that are low in predator abundance, and offer abundant populations of forage species (Hohman et al. 1992, Cooke et al. 1997). Sea ducks experience a prolonged period of flightlessness during molt. Additionally, the quality and quantity of available food is important to molting sea ducks, as previous studies have shown that food-stressed or starved birds experience poorer quality molts (Pehrsson 1987).

There does not appear to be an optimum molt initiation date in many sea duck species (Dickson et al. 2012), and the timing and duration of molt can vary greatly between species or years. Molting can take place anywhere between late June and early September in the GLBA region; this time range coincides with survey dates from nearly all of the sea duck surveys that are summarized in this document. The Hugh Miller complex within GLBA is a high use feeding and molting area for scoters

(Duncan and Climo 1991a), and several scoter surveys took place in this area in the late 1980s and early 1990s (Carter 1984, Fister and Widdice 1989, Duncan and Climo 1991b).

Because many of the surveys that took place during the molting window did not seek to identify whether birds were molting, it is impossible to say for certain that the data reported includes only molting sea ducks. Due to the lack of specific molting abundance, the current condition of this measure is unknown. However, reported below are the abundance values for all harlequin ducks, surf scoters, and white-winged scoters that were observed in the park during each study that took place within the broad molting timeline (late-June to mid-September). Instances where the number of molting individuals was identified are specified below.

Carter (1984)

Carter (1984) surveyed scoter populations in Hugh Miller Inlet, Charpentier Inlet, and Scidmore Bay in early August of 1984. While one of the objectives of the study was to generate a population estimate of molting white-winged and surf scoters, the author had difficulty ascertaining if observed birds were in fact molting. At the conclusion of the study, 4,331 scoters were observed (1,317 surf scoters, 2,750 surf/white-winged scoters [mixed], and 264 unidentified scoters); these observations fell within the typical molting period, but the exact molting status of these individuals cannot be known for certain.

Fister and Widdice (1989)

During mid-July surveys of Scidmore Bay, Weird Bay, and Charpentier Inlet in 1989, Fister and Widdice (1989) observed multiple species of sea ducks. Twenty-one harlequin ducks were observed during the surveys, but no determination of molt status was made. White-winged and surf scoters were not identified to species during this survey, but were grouped together as “scoters.” Over the duration of the survey, Fister and Widdice (1989) observed 5,312 scoters in the study area; no judgement was made regarding molt status.

Duncan and Climo (1991a)

Harlequin ducks, white winged scoters, and surf scoters were among nine indicator species chosen for the Beardslee Islands by Duncan and Climo (1991a). Duncan and Climo (1991a) surveyed birds and mammals of the Beardslee Island complex from June-September 1991; no indication of molting status was made. Ten surveys were conducted in the study area, with the number of transects utilized ranging from five to 16.

Scoters were most abundant in the Beardslee Islands during late-August and early-September (Figure 32). Surf scoters were more commonly observed than were white-winged scoters over the duration of the study (1,991 surf scoters compared to 1,533 white-winged scoters); the number of scoters unable to be identified to species accounted for over half of all scoter species observations (3,685). Total scoter abundance values did not exceed 180 individuals in the study area until 30 July 1991, when 966 individual scoters were observed (Figure 32). Harlequin duck abundance did not follow the same general trend as did scoters, as harlequin ducks were most abundant in mid-July, and had abundance values that were comparably high throughout the duration of the survey (Figure 33).

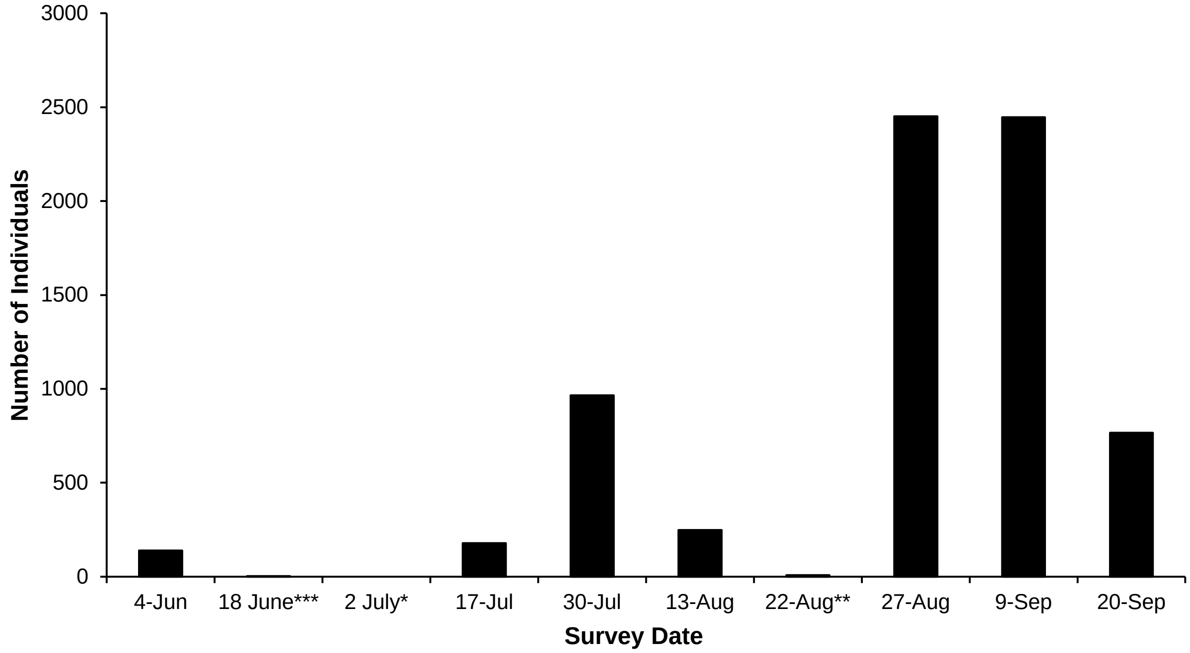

Figure 32. Scoter abundance during Duncan and Climo (1991a) surveys of the Beardslee Island complex in GLBA. * indicates a survey date where only five transects were surveyed, ** indicates a date where only six transects were surveyed, and *** indicates a date where 16 transects were surveyed. All other dates had 13 transects surveyed.

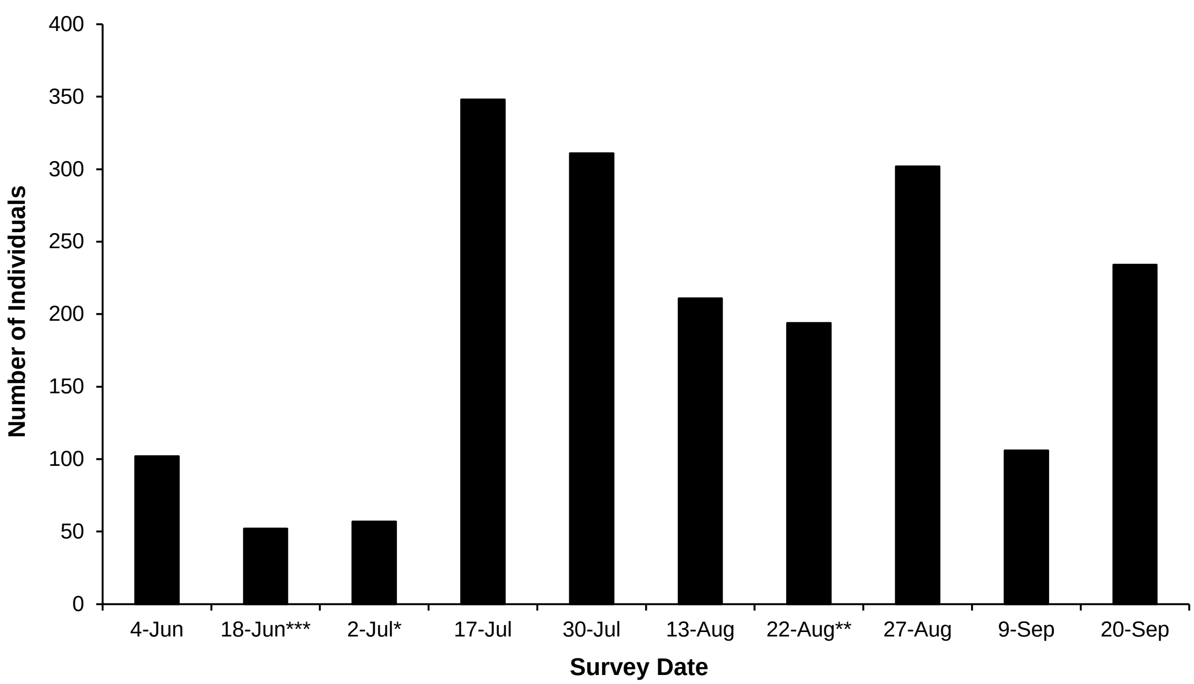

Figure 33. Harlequin duck abundance values observed during Duncan and Climo (1991a) surveys of the Beardslee Island complex in GLBA. * indicates a survey date where only five transects were surveyed, ** indicates a date where only six transects were surveyed, and *** indicates a date where 16 transects were surveyed. All other dates had 13 transects surveyed.

Duncan and Climo (1991b)

Duncan and Climo (1991b) surveyed the scoter population of the Hugh Miller complex in early August of 1991. The number of scoters (not identified to species) observed over the two day survey was 1,983. Molt status was not determined, as most of the ducks were observed at a distance and were on the water when observed (Duncan and Climo 1991b).

Piatt et al. (1991)

From mid-June to mid-July 1991, Piatt et al. (1991) surveyed the shoreline along most of GLBA's coast. The purpose of the survey was to document murrelets, but observers also reported numbers of all marine bird species that were encountered. Observers did not report molt status of observed species. Table 22 displays sea duck abundance values observed during Piatt et al. (1991) surveys. Scoters accounted for over 80 percent of all sea duck observations, with white-winged scoters being observed three times as frequently as surf scoters. Scoters were most abundant in Scidmore Bay, Queen Inlet, and near Composite Island (Piatt et al. 1991). Harlequin ducks were observed in lower numbers when compared to scoter species, and comprised only 5% of all sea duck observations (Table 22). Harlequin ducks were observed in highest numbers in Scidmore Bay, Tlingit Point, and near Muir Point/Muir Inlet (Piatt et al. 1991).

Table 22. Abundance of sea ducks observed on shoreline transects in GLBA from 13 June - 15 July 1991. Numbers include birds observed both on and off transects (Piatt et al. 1991, J. Piatt et al. unpublished data).

Species	Number of Individuals	% of Total
white-winged scoter	6233	49.08
unidentified scoter	2518	19.83
surf scoter	1820	14.33
common merganser	1305	10.28
harlequin duck	705	5.55
unidentified merganser	80	0.63
unidentified goldeneye	22	0.17
red-breasted merganser	10	0.08
Barrow's goldeneye	4	0.03
oldsquaw (long-tailed duck)	2	0.02
All Goldeneyes	**26**	**0.20**
All Scoters	**10571**	**83.24**
All Mergansers	**1395**	**10.99**
Total Seaducks	**12699**	**100**

USGS Marine Predator Surveys (Robards et al. 2003; Bodkin et al. 2001)

As part of USGS marine predator vessel surveys in GLBA from 1999-2001, Bodkin et al. (2001) and Robards et al. (2003) documented abundance of scoters and harlequin ducks in the park. Observers recorded the number of individuals, density, and approximate location of species (sea ducks were distributed almost exclusively within nearshore habitat; Bodkin et al. 2001). Surveys were conducted in winter (November to March) and summer (June); June surveys were the only surveys that approached the molting window for these priority species, and will be the only surveys summarized in this measure. Winter surveys are summarized below in the "Abundance of Wintering Barrow's Goldeneyes, Harlequin Ducks, and Surf Scoters" measure.

Harlequin duck abundance values ranged from 1,192 individuals (1999) to 1,645 individuals (2000; Figure 34), and annual variations appeared to be minimal, although from only 3 years of data it is impossible to determine any absolute trends. Harlequin ducks were among the most abundant sea duck species during the USGS vessel surveys, trailing only the scoter species and the common merganser in annual June abundance (Bodkin et al. 2001, Robards et al. 2003). Harlequin duck density during the study, which was reported as number of ducks observed per square km, was lowest in 1999 (2.93 ducks/km^2). Density values in 2000 and 2001 were comparable, being estimated at 4.33 and 4.43 ducks/km^2, respectively (Figure 35).

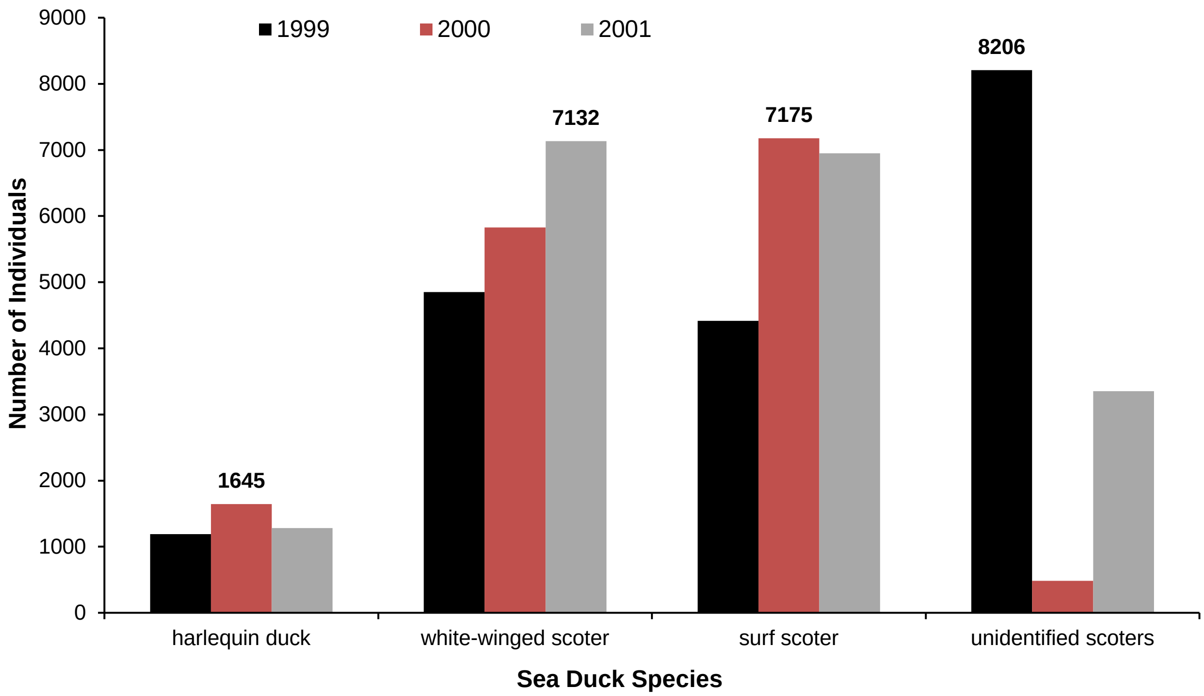

Figure 34. Harlequin duck, white-winged scoter, surf scoter, and unidentified scoter abundance values documented during USGS vessel surveys in GLBA from 1999-2001 (Bodkin et al. 2001, Robards et al. 2003). Surveys were conducted during March, June, and November.

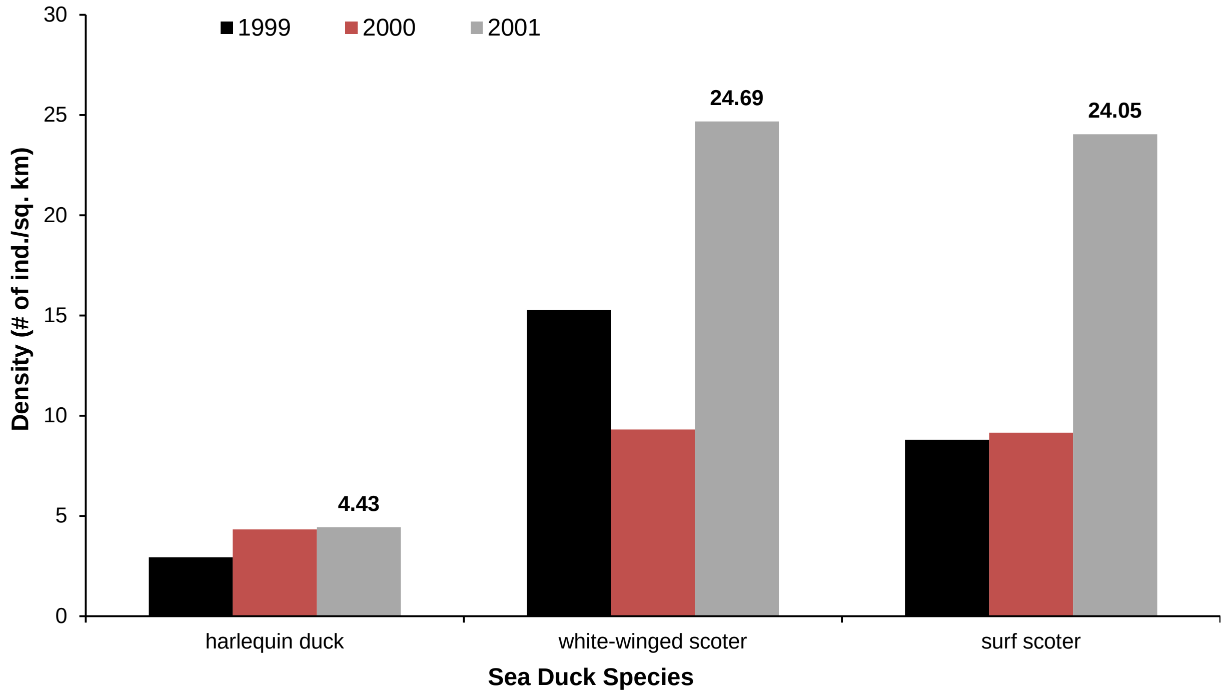

Figure 35. Harlequin duck, white-winged scoter, and surf scoter density estimates obtained during USGS vessel surveys in GLBA from 1999-2001 (Bodkin et al. 2001, Robards et al. 2003). Surveys were conducted during March, June, and November.

Surf scoter abundance values during the USGS vessel surveys ranged from 4,414 individuals in 1999 to 7,175 individuals in 2000 (Figure 34). 1999 had the lowest abundance out of any year by a relatively large margin, with abundance estimates in 2000 and 2001 being 1.5 times greater than 1999 values. White-winged scoter abundance was similarly low in 1999 (4,849 individuals), but abundance values increased each year of the survey (Figure 34); the 7,132 individuals observed in 2001 was among the highest observed abundance values for any species (second to only the 7,175 surf scoters observed in 2000). The number of unidentified scoters was variable by year (potentially due to observer bias/skill or observation conditions) with peak abundance values in 1999 (8,206 individuals; Figure 34).

Surf scoter density during the study (scoters/km^2) was consistent between 1999 and 2000, with values of 8.8 and 9.15 scoters/km^2, respectively (Figure 35). 2001 marked a noticeable increase in surf scoter density with 24.1 scoters/km^2 observed (Figure 35). White-winged scoter density was more variable between 1999 and 2000 when compared to trends observed in surf scoters (15.27 and 9.31 scoters/km^2, respectively), but much like the surf scoter the density estimate for 2001 was the highest observed for the species during the study (24.1 scoters/km^2; Figure 35).

Abundance of Wintering Barrow's Goldeneyes, Harlequin Ducks, and Surf Scoters

As has been previously mentioned, Glacier Bay represents a critical wintering habitat for several duck species, most notably Barrow's goldeneyes, harlequin ducks, and surf scoters. These species, exhibit high levels of winter site fidelity (Iverson and Esler 2006), and likely return to the same winter range each year (Kirk et al. 2008). Monitoring abundance of these wintering species provides not only valuable information regarding trends in their populations, but it also provides information regarding the health of their forage species in the Bay. According to Fretwell and Lucas (1970), the ideal free distribution model predicts that the density of predatory species tends to be positively linked with the density of their prey species. Changes in the abundance of these sea duck species will likely indicate a change in the abundance and density of their targeted prey species.

Few studies have documented winter abundance of these priority species. In GLBA, only Bodkin et al. (2001), Robards et al. (2003), and the annual CBC efforts have documented sea duck abundance in the winter months. In 1999, the USGS study in GLBA (Bodkin et al. 2001, Robards et al. 2003) documented winter abundance in November; winter surveys shifted to March for 2000 and 2001.

USGS Marine Predator Surveys (Robards et al. 2003; Bodkin et al. 2001)

Winter marine bird observations were dominated by a few sea duck species, namely harlequin ducks, scoters, goldeneyes, bufflehead, and long-tailed ducks. These species accounted for over 65% of all bird observations during the USGS surveys (Bodkin et al. 2001). Harlequin duck winter abundance ranged from 325 individuals (1999) to 463 individuals (2000; Figure 36). Trends in abundance are difficult to determine, as data were collected for only three winter seasons. Harlequin duck density during the study period (reported as number of ducks observed per square km) was relatively low and stable, with the peak value occurring in 2000 (4.57 ducks/km^2) and a range in values between 3.64 and 4.57 ducks/km^2 (Figure 37).

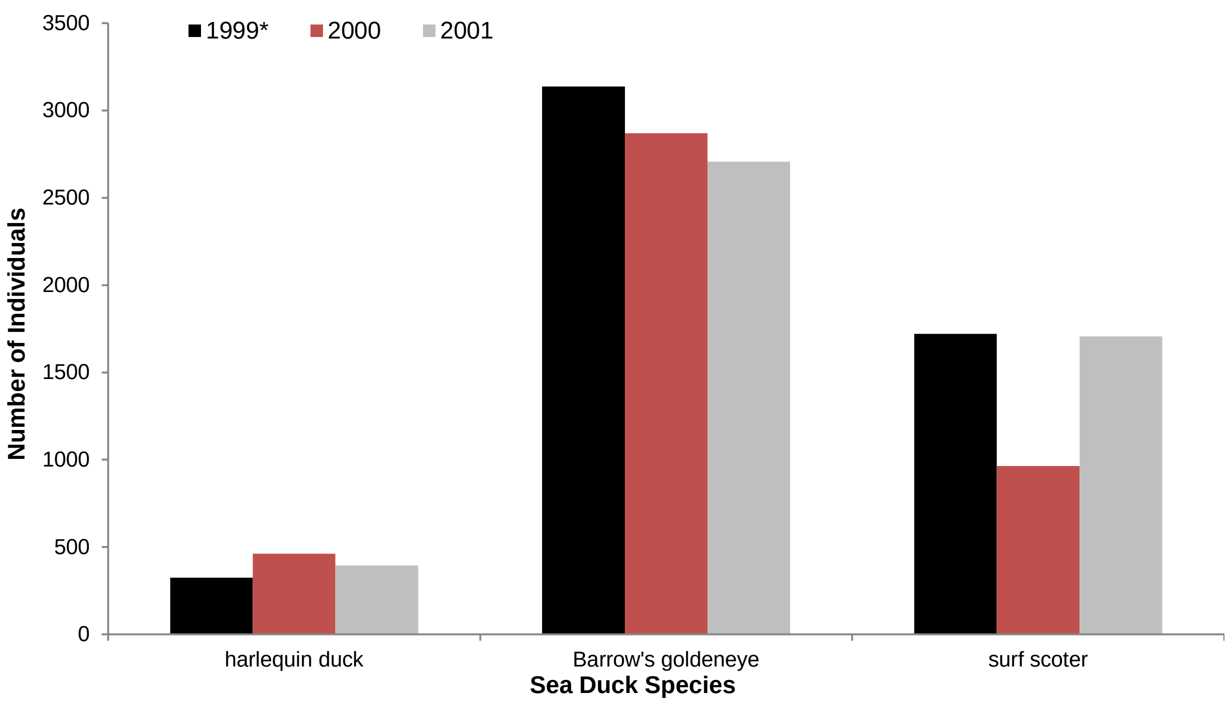

Figure 36. Harlequin duck, Barrow’s goldeneye, and surf scoter winter abundance values documented during USGS vessel surveys in GLBA from 1999-2001; *1999 surveys were conducted in November, all other surveys were conducted in March. Unidentified scoter species are not included in this figure (Bodkin et al. 2001, Robards et al. 2003).

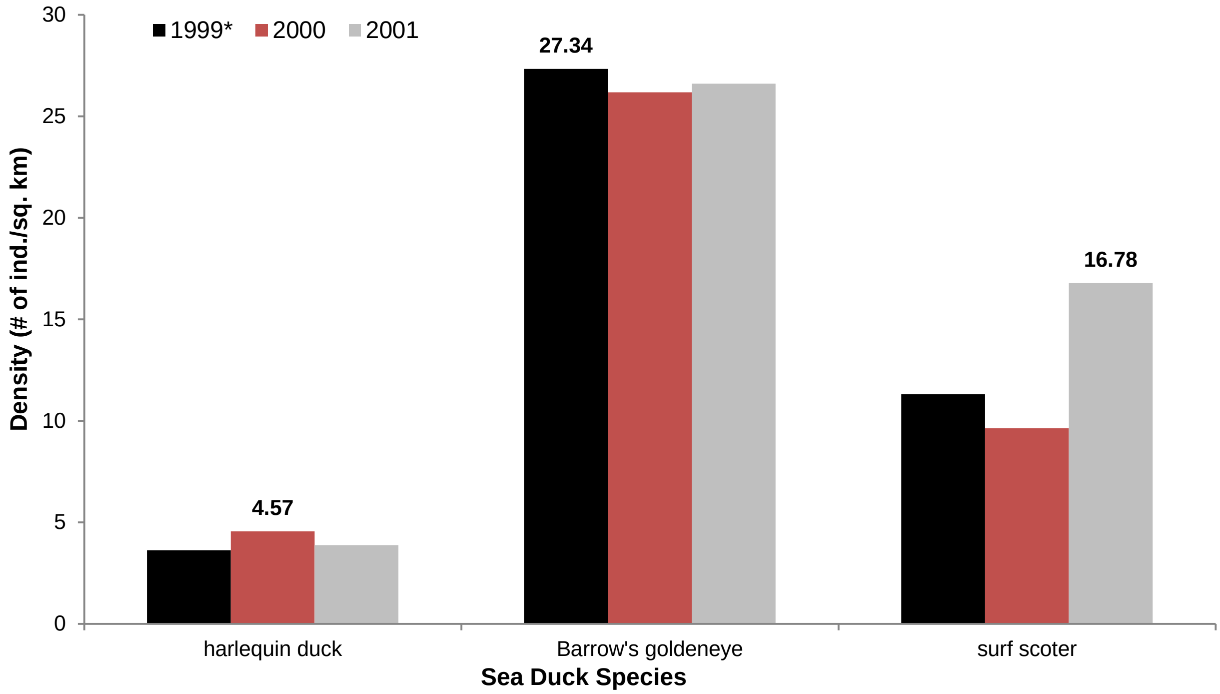

Figure 37. Harlequin duck, Barrow's goldeneye, and surf scoter density estimates obtained during USGS vessel surveys in GLBA from 1999-2001; *1999 surveys occurred in November, while the remaining surveys occurred in March. Unidentified scoter species are not included in this figure (Bodkin et al. 2001, Robards et al. 2003).

Barrow's goldeneyes were among the most abundant sea duck species during the USGS winter surveys, averaging over 2,900 observations each year (Bodkin et al. 2001, Robards et al. 2003). Abundance values declined during each year of the USGS study (Figure 36); abundance estimates peaked during November 1999 surveys (3,138 individuals observed), and declined to 2,706 individuals observed during the March 2001 survey. Barrow's goldeneye density during the study period was high and relatively stable, with peak estimates observed during November surveys in 1999 (27.34 ducks/km^2) and annual variations not exceeding ±1.15 ducks/km^2 (Figure 37).

Surf scoter annual abundance was the most variable among the three priority species identified in this measure. While the most surf scoters were observed in 1999 (1,721 individuals), 2001 surveys yielded a very similar number of birds (1,706); however, only 965 individuals were observed in 2000 (Figure 36). Density estimates for surf scoters ranged from 9.64 ducks/ km^2 (2000) to 16.78 ducks/ km^2 (2001) (Figure 37).

Gustavus, AK Christmas Bird Count Data (1968-2014)

The Gustavus, AK CBC represents the most continuous source of bird data in the GLBA region, with counts occurring almost every year from 1968-present. The CBC methodology is an example of an index count, which is a methodology that tallies the number of bird detections during surveys of points, transects, or other defined regions (Kendeigh 1944, Verner 1985, Bibby et al. 1992, Ralph et al. 1995, Rosenstock et al. 2002). Index counts quantify bird species' distribution, occurrence, habitat relationships, and population trends (Rosenstock et al. 2002).

The Gustavus CBC surveys only a portion of GLBA (Figure 31), so results from the survey may not be completely indicative of the winter abundance trends for these priority sea duck species in the entire Bay. Counts such as the CBC (or other index counts, e.g., breeding bird surveys) are neither censuses nor density estimates (Link and Sauer 1998). Possible bias of count locations and the number of observers limit the overall usefulness of index count data, and it is often not advisable to estimate overall population sizes from these data alone (Link and Sauer 1998); these biases may influence how many individuals are observed in a given year, and may potentially explain the annual variation observed in species each year.

Of the three priority species discussed in this measure, harlequin ducks had the lowest annual average abundance values (51.9 ducks/year; Figure 38). Abundance values were highly variable, with peak abundance for the species being observed in 1998 (190 individuals) and the lowest number of individuals observed in 1970 (3 individuals). Recent CBCs have reported below average numbers of harlequin ducks, with 34 and 26 ducks observed in 2013 and 2014, respectively; six of the past seven CBC efforts have yielded below average numbers of harlequin ducks (Figure 38).

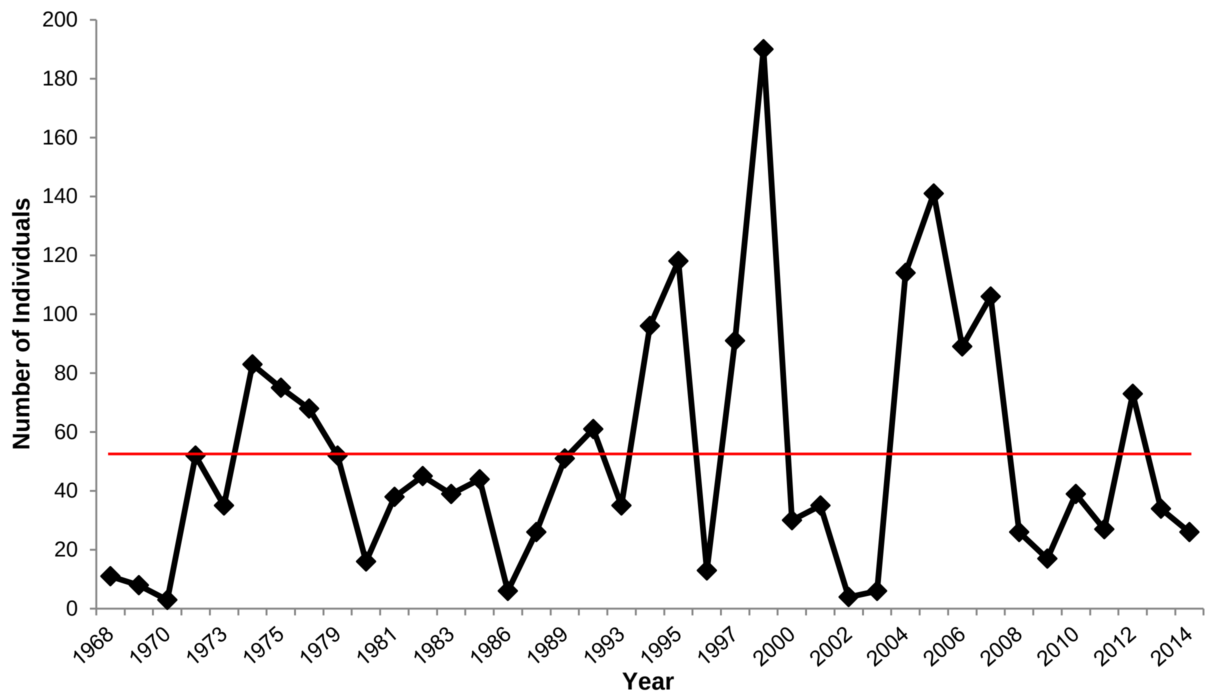

Figure 38. Annual abundance of harlequin ducks in the GLBA area as reported by the Gustavus, AK CBC count from 1968-2014. The solid red line indicates the 39-year average for abundance, which was 51.9 individuals.

From 1968-2014, an average of 346.3 Barrow's goldeneyes were observed during the Gustavus CBC (Figure 39). As was the trend for most species observed during the CBC, annual abundance values were highly variable for the Barrow's goldeneye. For example, in a 4-year span from 2002-2005, Barrow's goldeneyes had abundance values of 878 (count-high), 136, 70, and 832, respectively. Six of the past 10 years have had above average abundance values, with two of those years being only slightly below average (2009: 337 individuals; 2012: 333 individuals) (Figure 39). The last two years (2013, 2014) of the survey have reported above average abundance values for Barrow's goldeneyes (Figure 39).

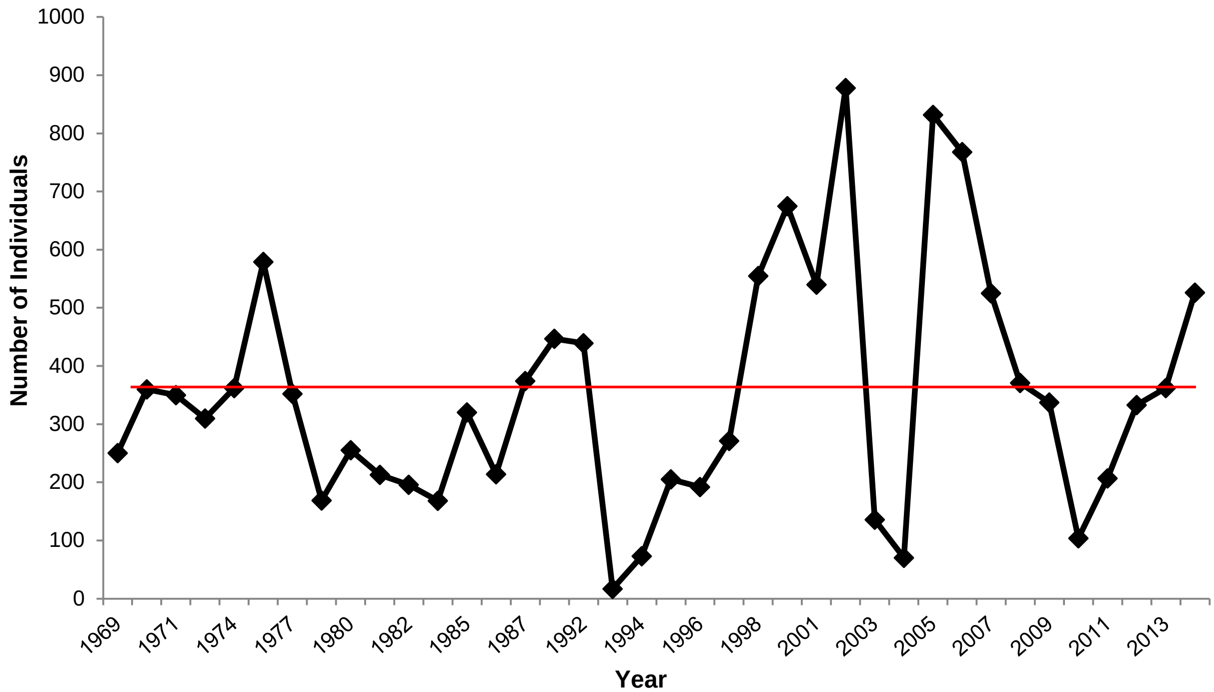

Figure 39. Annual abundance of Barrow's goldeneyes in the GLBA area as reported by the Gustavus, AK CBC count from 1968-2014. The solid red line indicates the 39-year average for abundance, which was 346.3 individuals.

Of the three species discussed in this measure, surf scoters were the most abundant, averaging 455.6 individuals/count (Figure 40). Instances of >1,000 individuals observed occurred in 1974, 1989, 2005, 2007, and 2014 (2012 was close to 1,000 observations, with 988 individuals reported). Outside of these peak years, average abundance was typically between 100-300 individuals. Peak surf scoter abundance was observed in 2005, when 1,992 individuals were observed, while the lowest number of surf scoters was in 1996 (45 individuals; Figure 40). In the last 10 years, surf scoter abundance has been above average nine times, failing to exceed the average abundance value only in 2008 (Figure 40). Additionally, three of the four highest surf scoter counts have occurred in the last 10 years (2005:1,992 individuals; 2007: 1,394 individuals; 2014: 1,288 individuals) (Figure 40).

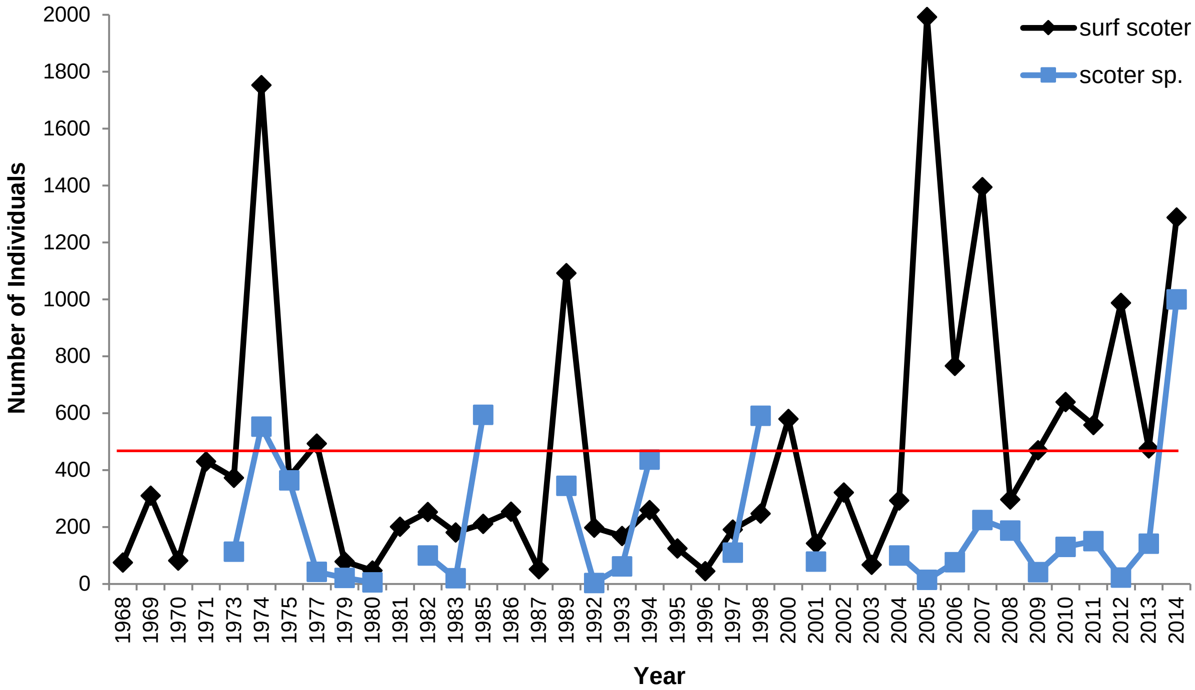

Figure 40. Annual abundance of surf scoters and unidentified scoter species in the GLBA area as reported by the Gustavus, AK CBC count from 1968-2014. The solid red line indicates the 39-year average for abundance for surf scoters, which was 455.6 individuals.

Surf scoters can be difficult to distinguish from other scoter species (e.g., white-winged scoters), especially when observed at a distance. For this reason, Figure 40 also includes the number of unidentified scoter species observed during each year of the Gustavus CBC. The number of unidentified scoters is highly variable, ranging from no reports of unidentified scoters (multiple years), to years where the number of unidentified scoters exceeds the number of identified scoters (e.g., 1998), and also to when the number of unidentified scoters is almost as high as the number of identified scoters (e.g., 2014). 2014 represented an unusually high count for both surf scoters and unidentified scoters; it is likely that the actual number of surf scoters observed in the CBC area was even higher than the 1,288 surf scoters reported. The total number of unidentified scoters reported is likely dependent upon the number of observers involved in the CBC each year, the skill of the observers involved, and the distance that the scoters are located from observation areas (as scoters observed at a great distance are less likely to be identified to species).

Winter Age Ratio of Barrow's Goldeneyes, Harlequin Ducks, and Surf Scoters

Many sea duck species have been declining globally, as 10 of North America's 15 sea duck species have exhibited declining trends (USFWS 1993, Goudie et al. 1994, Iverson et al. 2004), although the cause and extent of these declines are poorly understood. While the most recent data for many of these sea ducks species show stable or increasing trends, numbers still remain below what was reported historically (Flint 2012). Demographic data, such as recruitment or productivity of sea ducks, would provide researchers and managers with a tool to monitor variations in sea duck populations and to understand what mechanisms may be driving population declines. These data may also help to understand what life stages of sea ducks are being affected the most (Iverson et al. 2004).

Estimating recruitment in sea duck species has proven problematic in the past, as the typical methods for estimating recruitment in duck species involves pair:brood ratios, nesting success rates, and age ratios using hunter-returned wings. Iverson et al. (2004, p. 253) notes that these methodologies work poorly with sea duck species because "…most [sea duck species] breed in low densities in remote portions of the continent (Bellrose 1980) and rarely appear in hunter bags (Bartonek 1994)."

An alternative to the traditional productivity and recruitment methods is to use age ratios of non-breeding (in this case, overwintering) sea ducks obtained from count data. These age ratios allow for a comparison of the number of young birds to adults in overwintering areas. Winter age ratios are useful as they can estimate survival and recruitment during the previous breeding season(s) for many North American sea duck species, including goldeneyes (Duncan and Marquiss 1993), harlequin ducks (Smith et al. 2001, Rodway et al. 2003, Robertson 2008), and surf scoters (Iverson et al. 2004). Winter age ratios are able to be calculated as male sea duck species have plumage maturation patterns that allow age cohorts to be estimated (Palmer 1976). Female age ratios can also be estimated, though not through direct field observations (Robertson 2008). Iverson et al. (2004, p. 253) explains:

> *…female age classes cannot be distinguished alone and must be calculated from male age ratios. Calculation of female age ratios is straightforward, so long as secondary sex ratios are equal (Blums and Mednis 1996) and prebreeding mortality rates are the same for subadult males and females; however, any sex-related distributional biases must be known.*

Despite the utility of winter sex ratios in understanding sea duck population trends and dynamics, no study has taken place within GLBA that has investigated this parameter. Until such a study occurs, no assessment of condition or trend can be made regarding the winter age ratios of Barrow's goldeneyes, harlequin ducks, or surf scoters.

Foraging Effort of Molting and Wintering Sea Ducks (Barrow's goldeneyes, harlequin ducks, surf scoters, and white-winged scoters)

The primary prey species for most sea ducks are benthic invertebrates that are found in intertidal areas. An assortment of taxa are of particular importance for sea ducks, including mussels, limpets, clams, snails, and amphipods. Adequate food availability is important during periods of remigial molt, as daily energy and nutrient requirements increase during the feather production process (Murphy and King 1992, Murphy 1996, Hogan 2012, Hogan et al. 2013). Despite the limited ability to move during their molt, some sea duck prey species require only limited effort to capture, as epibenthic species such as mussels are readily captured from the ocean floor and are typically found in high densities. However, other prey species such as clams require a substantial amount of energy to excavate from the sediment on the ocean floor and are found in much lower densities (Hogan et al. 2013b). Due to characteristics such as these, sea duck species must adjust their forage species and strategies during these critical stages.

Foraging effort of molting and wintering sea ducks has been an area of particular importance and emphasis in recent years, although data are lacking in the GLBA area. There does not appear to be an optimal foraging strategy universally utilized during remigial molt, as inter- and intraspecific variations have been driven by location (VanStratt 2011, Hogan et al. 2013b), environmental conditions (e.g., weather, predation risk; Brown and Saunders 1998, Zimmer et al. 2010), and age/sex classes (Morton et al. 1990, Fischer and Griffin 2000, Badzinski and Petrie 2006, Hogan et al. 2013b). As Hogan et al. (2013) notes, many aspects of a species' molt strategy (specifically Barrow's goldeneye in Hogan et al. 2013) are shaped annually by environmental conditions. Molting is not the only critical foraging period for these birds, as the availability of food during the winter is also important. Additional research is needed during this period to observe foraging success and effort, as well as prey species selection

Data regarding Barrow's goldeneyes, harlequin ducks, surf scoters, and white-winged scoters in GLBA are confined to abundance counts; no study has investigated the foraging effort of wintering or molting sea ducks. Until such a study takes place, a statement of condition cannot be made regarding this measure.

<u>Threats and Stressor Factors</u>

Sea ducks prey primarily upon benthic invertebrates in intertidal and subtidal zones, and likely would be affected by changes in abundance of their forage species (e.g., mussels, limpets, snails, amphipods, polychaetes, clams, and urchins). While there is natural temporal variation in invertebrate communities, shifts in abundance or diversity due to climate change are an increasing threat to the invertebrate community. Much of the research pertaining to climate change in marine ecosystems has dealt with temperature variations (Harley et al. 2006); however, for invertebrate species that produce carbonate structures (e.g., clams, mussels), reduction of oceanic pH levels and the accompanied reduction in CO_2 levels could have pronounced impacts (Pörtner and Langenbuch 2005, Harley et al. 2006). Sea ducks would presumably be impacted by any such changes in abundance or quality of invertebrate prey species; however, links between prey conditions and effects on sea duck populations requires more empirical evaluation (Esler, written communication, 8 May 2014).

As has been mentioned previously, abundance of several sea duck species has declined globally over the past decades. While causes of these declines are poorly understood, it has been hypothesized that contaminants may play some role. Many of the prey species that sea duck species focus on (e.g., clams, mussels) are filter feeders and accumulate and concentrate contaminants. Some studies have documented elevated selenium (Henny et al. 1995, Franson et al. 2004, Mallory et al. 2004), copper (Mallory et al. 2004), and cadmium (Henny et al. 1995, Mallory et al. 2004) in sea duck species. The effects of contaminants on sea duck populations are in need of additional research to determine if contaminants are at all linked to the population declines, or if sea duck species naturally carry high levels of contaminants in their system.

Sea ducks are closely tied to the intertidal and subtidal benthic communities, as some species are year-round residents of this community and rely upon it for food, habitat, and nesting locations. These communities are affected by many factors, including natural (e.g., increases in sea otter abundance and foraging) and anthropogenic impacts (e.g., vessel traffic, human use, and pollution). The expansion, and subsequent population growth of sea otters in Glacier Bay is likely to have direct impacts to many sea duck communities, as otters and sea ducks have diets that overlap (both consuming many clam and mussel species; Sanger and Jones 1982, Goudie and Ryan 1991), and sea otters are known to dramatically impact the structure and function of nearshore ecosystems (Estes and Palmisano 1974).

Humans have the potential to disturb sea duck species in a variety of ways, and in GLBA vessel traffic is likely the primary source of anthropogenic disturbance. Vessel traffic has been shown to alter behavior of adults and lower survival of ducklings in nesting sea ducks species (Åhlund and Götmark 1989, Keller 1991) and increase the likelihood of gull predation (Mikola et al. 1994). The effects of vessel traffic during the molting and wintering periods is poorly understood/studied, but studies of other duck species during the nonbreeding season have indicated vessel traffic has the ability to alter a duck's behavior, habitat selection, and energy expenditure (Tuite et al. 1983, Bell and Austin 1985, Galicia and Baldassarre 1997). Due to the increased vessel traffic in the summer season, potential impacts of vessel traffic on sea ducks in GLBA would occur mainly in the summer during the molting period.

Sea ducks are particularly vulnerable to the effects of both catastrophic and low-level oil spills and pollution. Sea ducks typically forage in nearshore intertidal and subtidal habitats, and these habitats are the areas that are typically hardest hit by oil spills, as was seen during the Exxon-Valdez oil spill (EVOS) in 1989 (ADEC 1992, Neff et al. 1995, Lanctot et al. 1999); approximately 40% of the 42 million liters of crude oil that was lost during the EVOS accumulated in the nearshore environment of Prince William Sound (Galt et al. 1991). While it is likely that all sea duck species would be affected at some level by an oil spill, evidence from the EVOS suggests that species such as the harlequin duck and Barrow's goldeneye may be affected at a greater level. Recovery of oiled and dead harlequin ducks was problematic after the EVOS, but estimates that take into consideration collection and recovery problems indicate that between 1,298 and 2,650 harlequin ducks were killed during the immediate weeks following the spill (ECI 1991, Piatt and Ford 1996). Post spill recovery monitoring (1989-1998) showed very limited signs of population recovery in harlequin ducks (Lance et al. 2001), and populations in oiled regions had not yet fully recovered more than a decade after the spill (Esler et al. 2002).

Data Needs/Gaps

Sea ducks within GLBA have been understudied in the past few decades, with the last active monitoring occurring in 2001. Sea ducks utilize Glacier Bay during various life stages throughout the year, and the establishment of an annual molting period survey and an overwintering survey would provide valuable information for several species. With the global decline in sea duck populations, monitoring abundance trends in the GLBA region would help provide managers with a more accurate picture of local population health. It will also be important to monitor the spatial and temporal variation and trends in prey species of sea ducks. Prey sources for sea ducks experience natural variations, are affected by climate change, and are shared with other carnivorous species such as sea otters and other marine mammals. Monitoring of these shared resources is needed for not just sea ducks, but all predators in the nearshore intertidal system.

Outside of the species richness measure, which is unlikely to change outside of extreme changes to park conditions, none of the measures in this document have enough data to assess current condition. Two measures (winter age ratio of three priority duck species, and foraging effort of molting and wintering sea ducks) have not had any data collected within the park, and are in need of established monitoring protocols. Effects of various identified threats and stressors are poorly understood as well, especially in the GLBA region. The effect of human disturbances during the molting period (e.g., vessel traffic, kayaks, human use of shorelines) is not well known, and an investigation into the effects of contaminant concentration in prey species, similar to those discussed in Chapter 4.22 of this document, may help to inform managers of this potential threat (or may provide information regarding the amount of contaminants that sea ducks harbor without experiencing detrimental effects).

Overall Condition

Species Richness

The species richness measure was assessed a *Significance Level* of 2. The assignment of a slightly lower *Significance Level* was due in part to the fact that the species richness levels observed in the park are unlikely to change in the foreseeable future, barring some extreme changes to environmental conditions within Glacier Bay.

The NPS Certified Species List (NPS 2015) identifies 13 sea duck species; however, three of those species (king eider, black scoter, and common eider) are considered rare visitors to the park. The most recent (and most in depth) study to take place in GLBA was the USGS marine predator survey (Bodkin et al. 2001, Robards et al. 2003); this study reported 10 sea duck species within GLBA, and did not observe any of the rare species mentioned previously (Table 21). The Gustavus, AK CBC represents the longest continuous source of data for sea ducks in the region. Despite not being a sea duck-specific survey, the count has identified 12 species in the region during its 39-year history. The only species from NPS (2015) not observed during the CBC was the king eider.

There does not appear to be any cause for concern regarding the species richness measure in GLBA. The sea duck avifauna in GLBA is what should be expected for protected waters within coastal Alaska and British Columbia (Esler, written communication, 8 May 2014). Because of this, a *Condition Level* of 0 was assigned to the species richness measure.

Abundance of Molting Harlequin Ducks, Surf Scoters, and White-winged Scoters

The abundance of molting harlequin ducks, surf scoters, and white-winged scoters was assigned a *Significance Level* of 3. Despite the fact that there has not been a study in the park that focused exclusively on molting sea duck abundance, several of the sea duck surveys completed in the area have taken place during the molting period and serve as an adequate, although not precise, estimate of molt abundance.

The abundance of sea ducks during the molting period has been high in GLBA, especially for harlequin ducks and scoters (Table 22, Figures 32-35). During the most recent molting period surveys in the park (Bodkin et al. 2001, Robards et al. 2003), abundance values were high for the three species (Figure 34), although the scoter species were nearly five times as abundant as the harlequin duck, annually. Harlequin duck abundance remained relatively consistent for the duration of the study (1999-2001), while scoter abundance values increased from 1999-2001. White-winged scoters continued to increase in abundance in 2001 and experienced peak abundance values (7,132; Figure 34). Surf scoters slightly declined from their peak value of 7,175 in 2000, to 6,949 individuals in 2001 (Figure 34).

While the results of the USGS marine predator monitoring, coupled with older surveys of abundance (Carter 1984, Fister and Widdice 1989, Duncan and Climo 1991a, b, Piatt et al. 1991), indicate low to no concern regarding the abundance of molting sea ducks, there have been no data collected in the park that relate to this measure for nearly 15 years. Trend comparison from historic surveys and the USGS surveys seem to indicate that the abundance of these species remained stable, or even increased, in the 10 years between efforts. However, there was a considerable degree of variation between survey methodology between all of the efforts prior to 2001, which may account for increases in abundance in 2001 (USGS surveys were more intense and involved more survey dates, transects, and observers). Furthermore, surveys were located in different areas of the park, which likely influenced the number of species and individuals observed, depending on time of year the survey was conducted. Due to the lack of recent (within 15 years) data, any assessment of condition for this component would be speculative. Because of this, a *Condition Level* was not assigned to this measure. Assuming future studies of molting abundance in the park use methodologies similar to the USGS marine predator surveys, comparisons could be made to assess trends in abundance and distribution. The data gathered from future studies could also be used to evaluate molting distributions relative to habitat features and human activities (e.g., vessel and kayak use).

Abundance of Wintering Barrow's Goldeneyes, Harlequin Ducks, and Surf Scoters

During project scoping, the abundance of wintering Barrow’s goldeneyes, harlequin ducks, and surf scoters measure was assigned a *Significance Level* of 3. This measure is of high importance to the park, as Glacier Bay represents a vital overwintering area for many sea duck species. Unfortunately, only a few studies have documented the winter abundance of these focal species in the park (Bodkin et al. 2001, Robards et al. 2003, Gustavus, AK CBC).

Barrow’s goldeneyes were among the most abundant overwintering sea duck during the USGS marine predator monitoring (Figure 36). Peak winter abundance values were observed in 1999 (3,138 individuals), and values declined in 2000 (2,870 individuals) and again in 2001 (2,706 individuals; Figure 36). Harlequin duck abundance values were lower and relatively stable during the USGS study, while surf scoter species experienced a low year in 2000 before rebounding to abundance levels that nearly equaled 1999 values (Figure 36). It is important, however, to recognize that these are not abundance estimates for the park, as the surveys only cover a sample of GLBA. Expanded estimates for Barrow’s goldeneyes could be in the 10s of thousands (Esler, written communication, 1 October 2015).

Of the three priority species discussed in this measure, harlequin ducks had the lowest annual average abundance values from CBCs (51.9 ducks/year; Figure 38). Recent CBCs have reported below average numbers of harlequin ducks, with 34 and 26 ducks observed in 2013 and 2014, respectively; six of the past seven CBC efforts have yielded below average numbers of harlequin ducks (Figure 38). These numbers may not be truly indicative of the actual abundance trends for sea duck species in the winter in GLBA, though, as CBC efforts will vary in effort and coverage from year to year.

Barrow’s goldeneyes had an average annual abundance of 346.3 ducks/year, although abundance values were highly variable per year (Figure 39). Six of the past 10 years have had above average abundance values, with two of those years being only slightly below average (2009: 337 individuals; 2012: 333 individuals) (Figure 39). The last two years of the survey (2013, 2014) have reported above average abundance values for Barrow’s goldeneyes (Figure 39).

Surf scoters were the most abundant of the priority sea ducks observed on the CBC, averaging 455.6 individuals/count (Figure 40). Instances of >1,000 individuals observed occurred in 1974, 1989, 2005, 2007, and 2014 (2012 was close to 1,000 observations, with 988 individuals reported). Outside of these peak years, average abundance was typically between 100-300 individuals. In the last 10 years, surf scoter abundance has been above average nine times, failing to exceed the average abundance value only in 2008 (Figure 40). Additionally, three of the four highest surf scoter counts have occurred in the last 10 years (2005:1,992 individuals; 2007: 1,394 individuals; 2014: 1,288 individuals) (Figure 40).

A *Condition Level* for this measure was not assigned due to a lack of recent monitoring/survey data. The USGS marine predator survey represents the only intensive sea bird survey that has occurred in the park in the last 15 years. Based on those surveys, sea ducks constituted a significant portion of the marine bird community throughout the year, although more recent data would be useful. Although a good deal of data exists from the Gustavus, AK CBC, there are inherent biases that exist in the survey’s methodology that makes assessment of current condition problematic. The Gustavus CBC surveys only a portion of GLBA (Figure 31), so results from the survey may not be completely indicative of the winter abundance trends for these priority sea duck species in the entire Bay. Count locations and the number of observers limit the overall usefulness of index count data, and it is often not advisable to estimate overall population sizes from these data alone (Link and Sauer 1999); these biases may influence how many individuals are observed in a given year, and may potentially explain the annual variation observed in species each year. While the data provide a useful glimpse into the abundance trends of sea ducks in GLBA, the data may not accurately describe the current trends and condition for the park as a whole.

Winter Age Ratio (productivity) of Barrow's Goldeneyes, Harlequin Ducks, and Surf Scoters

During project scoping, the decision to assign this measure a *Significance Level* of 2 was based on the fact that this measure (along with the foraging effort of molting and wintering sea ducks measure) required more expertise and effort than quantifications of abundance and distribution. Unfortunately, no data exist for GLBA that directly relate to winter age ratios of sea ducks. Until such a study occurs in the park, no assessment of current condition or trend can be made; a *Condition Level* was not assigned to this measure.

Foraging Effort of Molting and Wintering Sea Ducks (Barrow's goldeneyes, harlequin ducks, surf scoters, and white-winged scoters)

Similar to the winter age ratio measure, this measure was assigned a *Significance Level* of 2 during project scoping. No study has taken place in the GLBA area that documents the foraging effort of molting or wintering sea ducks; until a study takes place that documents foraging effort, a *Condition Level* cannot be assigned for this measure.

Weighted Condition Score (WCS)

The sea duck component was not assigned a *Weighted Condition Score*, due to lack of recent (within 15 years) data for several of the measures. Until a survey or study collects new data relating to the measures identified in this component, condition assessment would be speculative at best.

Sea Ducks			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Species Richness	2	0	
Abundance of molting focal species	3	n/a	
Abundance of wintering focal species	3	n/a	
Winter age ratio of focal species	2	n/a	
Foraging effort of molting and wintering focal sea duck species	2	n/a	

4.5.6. Sources of Expertise

- Dan Esler, USGS Alaska Science Center Research Wildlife Biologist

4.5.7. Literature Cited

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4.6. Breeding Landbirds

4.6.1. Description

Bird populations often act as excellent indicators of an ecosystem's health (Morrison 1986, Hutto 1998, NABCI 2009). Birds are typically easy to observe and identify, and bird communities often reflect the abundance and distribution of other organisms with which they co-exist (Blakesley et al. 2010). This component will focus specifically on the breeding landbirds of GLBA; a landbird is defined as a bird species that has a principally terrestrial life cycle (Rich et al. 2004).

GLBA is home to a wide variety of habitats, and during the breeding season many species of migratory birds (e.g., Wilson's warbler [*Cardellina pusilla*], hermit thrush [*Catharus guttatus*]) come to Glacier Bay to nest (Photo 18). Breeding landbirds are found in highest numbers during the spring and summer months in the park, and the largest congregations tend to occur in lower elevation shrub and deciduous forest communities (Willson and Gende 1998). Isostatic rebound and vegetation succession continue to alter the landscape of GLBA, and preferred landbird habitat locations are constantly shifting. Monitoring avian population health and diversity in their preferred habitats will be important for detecting ecosystem changes or populations trends.

Photo 18. Wilson's warbler (*Cardellina pusilla*) (left) and hermit thrush (*Catharus guttatus*) (right) (NPS photos).

4.6.2. Measures

- Species richness
- Species abundance
- Trends in abundance of species of conservation concern

4.6.3. Reference Conditions/Values

A reference condition has not been assigned to the breeding landbirds component. A lack of historic data related to this component makes it difficult to determine a point in time to compare to the current population. The current condition of this resource will be determined by the best professional judgment of the identified subject experts.

4.6.4. Data and Methods

The NPS Certified Bird Species List for GLBA (NPS 2014) and the GLBA Bird Checklist (Paige and Drumheller 2013) were the primary species lists used for this assessment; these lists represent all of the confirmed bird species that have been found in the park. For this component, only bird species considered landbirds (as defined by Rich et al. 2004) were included. Breeding status of landbirds was determined using the designations provided in NPS (2014) and Paige and Drumheller (2013). For instances where breeding status was uncertain, the Birds of North America online database (http://bna.birds.cornell.edu) was consulted to determine a species' breeding range and whether or not it included the GLBA area. The studies described below have been adjusted so that only data regarding breeding landbirds are discussed in this document (i.e., instances of non-breeding landbirds have been removed and are not discussed).

Trautman (1966) investigated the length of time needed for the development of habitat types and their associated bird assemblages post-glacial retreat. Transects were established in Muir Inlet near Casement Glacier during the breeding season of 1965, and were surveyed frequently from 10 June - 2 August 1965. Trautman (1966, p.121) describes the survey area in detail:

> *The detailed ornithological studies in the present report [Trautman 1966] were conducted … on the eastern side of Muir Inlet, from opposite Garforth Island northward 22 kilometers along the east coast of the Inlet, including Sealers Island; to approximately 1.5 kilometers north of The Nunatak; east 5 kilometers to the lower half of Red Mountain; Southwest 19 kilometers along the retreating front of Casement Glacier to, and including, the large island near the head of Adams Inlet; continuing south-westward along the south shore of Adams Inlet, around the lower slopes of Mt. Case, thence southward along the lower slopes of Mt. Wright, to the starting point opposite Garforth Island.*

The majority of transects radiated outwards from the receding edge of Casement Glacier and crossed sections of land where the date of deglaciation was known. In addition to transect data, opportunistic bird observations were also documented.

Wik (1967) compiled one of the earliest bird checklists produced for GLBA, and documented all species that had been reliably reported within GLBA. Wik (1967) used previous work by Bruce W. Black in 1954 (checklist now unavailable) and Jacot (1962), and also utilized species lists obtained from Trautman (1966) and NPS field rangers from 1967 to populate the checklist. The Wik (1967) checklist increased the number of birds observed in the monument from 110 (Jacot 1962) to 173.

Muldoon (1986) conducted a brief avian survey in the Burroughs Glacier area in 1986. Surveys took place from 5-15 June, and were designed to document bird species distribution in the Burroughs Glacier area, and traversed five major habitat types: ice, early pioneer to open thicket, dense thicket, shore, and the inlet. The habitat types were selected to roughly approximate the age of glaciation in the study area. The dense thicket habitats were deglaciated prior to 1960, while the early pioneer and open thickets were deglaciated between 1960 and 1986.

Willson and Nichols (1997) used fixed-radius point counts to survey breeding landbirds in the east arm of Glacier Bay during the 1997 breeding season. Thirteen study locations were identified in the east arm of Glacier Bay, with an additional location identified along the road to Bartlett Cove. At each location, five 50 m (164 ft) radius plots were established approximately 150 m (492 ft) from each other. Each plot was surveyed one time for 8 minutes during the study. Sampling locations were situated along a vegetation gradient to identify potential distribution trends regarding successional habitats. Five categories of habitat structural diversity were identified during the surveys:

- Category 1 – sites with a few shrubs scattered across wide open spaces (dryas mats, lichens, gravel);
- Category 2 – represents sites of shrub thickets with small openings;
- Category 3 – adds small trees to shrub thickets that were almost completely closed;
- Category 4 – consists of sites with dense thickets overtopped by young cottonwoods;
- Category 5 – tall forest of mixed deciduous and coniferous trees, usually with a well-developed shrub layer (Willson and Nichols 1997, p.2).

In 1998, a breeding landbird survey was conducted in the midbay region of Glacier Bay by Willson and Gende (1998). The primary objective of Willson and Gende (1998) was to compare the breeding landbird communities along streams that were thought to support salmon runs to landbird communities along streams that did not support salmon runs. Sample sites were located in the Geikie Inlet area (11 sites), Fingers Bay (2 sites), Spokane Bay area (2 sites), Good River (1 site), and the Bartlett River (1 site). Censuses were conducted twice at each site, and lasted for 8 minutes each. Each site had five points that were located parallel to the associated stream, and all birds seen and heard within 50 m (164 ft) were recorded.

Saracco and Gende (2004) investigated landbird abundance and species composition in GLBA during a 2004 survey. Nine survey transects within Glacier Bay, and one transect on Excursion Ridge were sampled during the 2004 breeding season (Figure 41). Saracco and Gende (2004) used variable circular plot point counts, and established 57 points along the 10 survey transects in the park. Points were separated from each other by a minimum of 200 m (656 ft), and were located at least 150 m (492 ft) from any woody vegetation edge habitats. Surveys were completed within 4 hours of sunrise. Additionally, Saracco and Gende (2004) used non-metric multidimensional scaling (NMS) to investigate variation in observed community composition at transects across the study region. NMS was used to investigate the potential relationship that habitat type (early and late coniferous and deciduous forests) and east-west orientation may have on avian community structure.

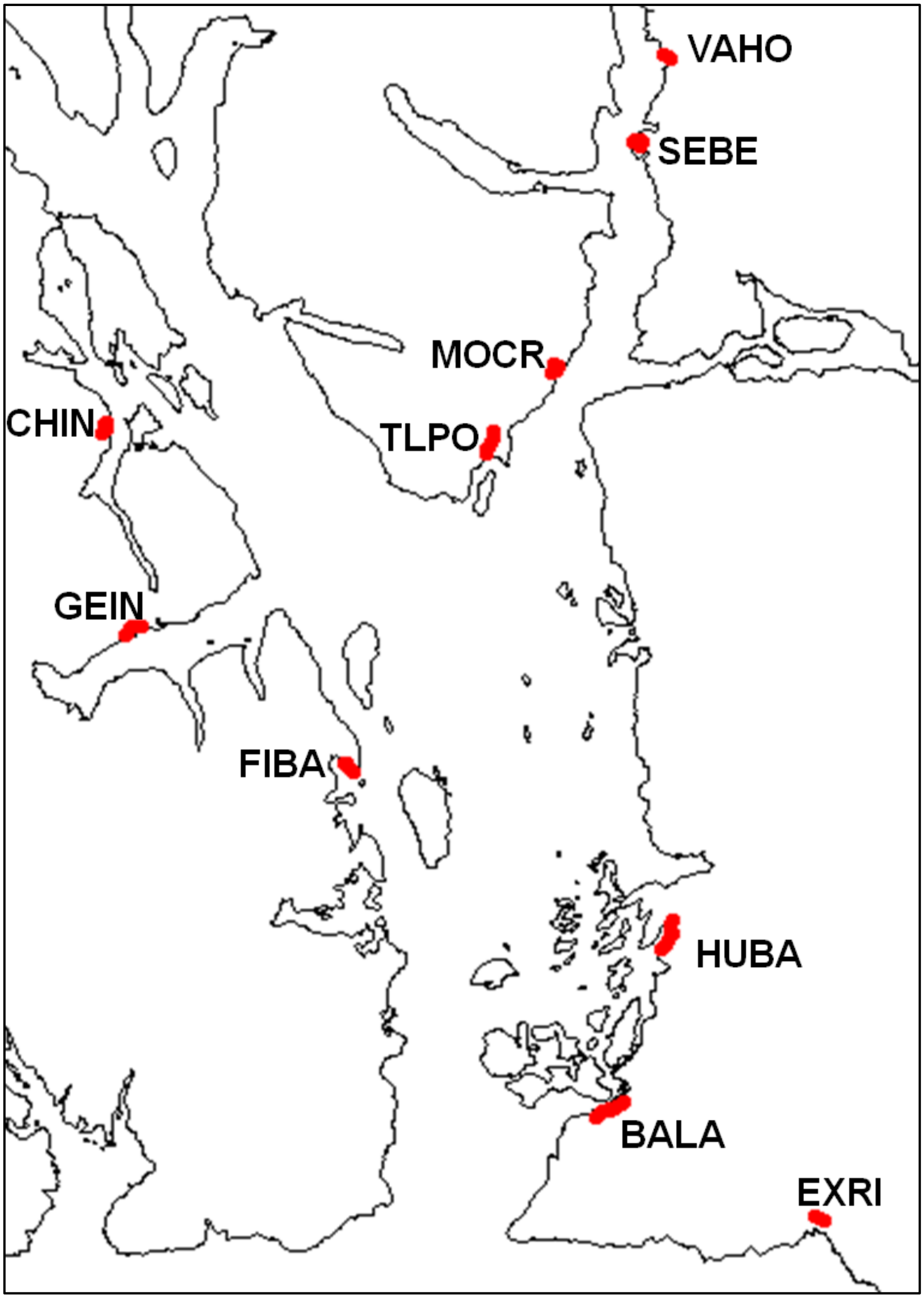

Figure 41. Transect locations during Saracco and Gende (2004). Acronyms include: CHIN (Charpentier Inlet), GEIN (Geikie Inlet), FIBA (Finger Bay), EXRI (Excursion Ridge), BALA (Bartlett Lake Trail), HUBA (Hutchins Bay), TLPO (Tlingit Point), MOCR (Big Creek), SEBE (Sealers beach), VAHO (Van Horne) (Saracco and Gende 2004).

4.6.5. Current Condition and Trend

<u>Species Richness</u>

The species richness measure can indicate overall habitat suitability for breeding birds, and is vital to understand the effects of changing landscapes on native biodiversity. The various studies that have occurred in GLBA since 1965 have all used unique methodologies, occurred at varying times of the year, and have been conducted in locations that often do not overlap. These variations make comparisons across studies problematic, and it is likely impossible to draw accurate conclusions by comparing each study to each other. Also of note is the fact that none of the studies summarized in this assessment accounted for detectability differences among species and habitats or sampling sites. Because of this, species richness as reported here reflects only the number of species observed and not the actual number of species that were present.

NPS Certified Species List (NPS 2014)

The NPS Certified Bird Species List contains 257 species, 59 (23%) of which are breeding landbird species (Figure 42, Appendix C). This list, however, does not allow for a specific analysis of annual species richness, as no data are collected yearly, and the list only documents the presence (or historic presence) of the identified species. The species documented on this list are the product of the various avian surveys in the park that are summarized in this document.

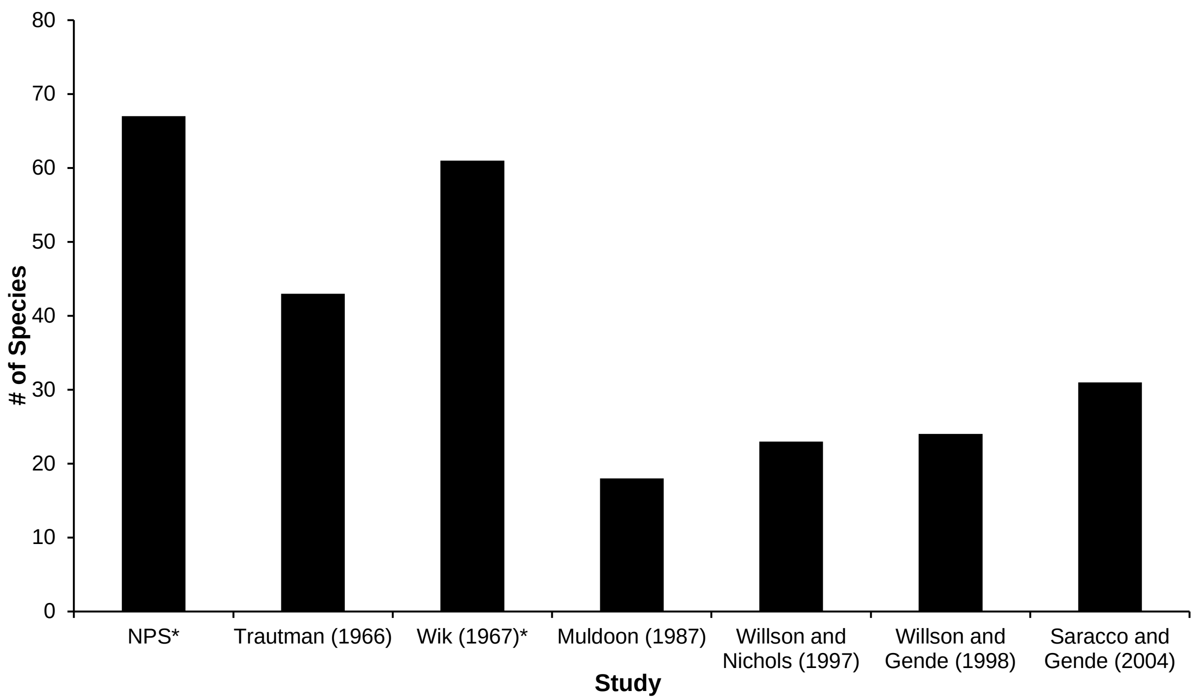

Figure 42. Observed species richness values during landbird surveys in GLBA from 1965-2004. Studies denoted with an * indicate a species checklist and did not conduct on the ground surveys.

Trautman (1966)

Trautman (1966) represents the earliest bird survey in the park, and documented the presence of 43 breeding landbird species in the east arm (Muir Inlet) of Glacier Bay (Figure 42). Trautman (1966) established monitoring transects along a successional gradient in Muir Inlet in order to observe the distribution of bird species in habitats in varying degrees of succession. Species richness values tended to increase as the age/successional stage of the habitat increased. However, Trautman (1966, p. 129) noted that "…a species of bird moved into an area immediately or shortly after the establishment of its habitat, provided that there was a nearby population of that species. It remained a nesting species as long as its habitat remained." Many breeding bird species are habitat specialists, and rely on a certain habitat type in the park. Species such as the snow bunting (*Regulus calendula*) and gray-crowned rosy finch (*Leucosticte tephrocotis*) were commonly observed near or on glaciers during Trautman (1966); however, these species were uncommon in later successional habitats (Bartlett Cove) of the park.

Wik (1967)

Wik (1967) is one of the earliest, and most thorough bird checklists produced for GLBA, and provides an approximation of what breeding landbird species could be expected to occur in the park in a given breeding season. Sixty-one breeding landbird species were identified on the checklist (Figure 42).

Muldoon (1987)

During a brief avian distribution survey, Muldoon (1987) documented 18 breeding landbird species in the Burroughs Glacier area in 1986 (Figure 42). Only two species were observed on or near the ice of Burroughs Glacier (gray-crowned rosy finch, snow bunting), and both species were observed preying on ice worms on the glacial ice. The habitat zone with the highest observed species richness was the early pioneer to open thicket habitat type; commonly observed species included the fox sparrow (*Passerella iliaca*), savannah sparrow (*Passerculus sandwichensis*), and yellow warbler (*Setophaga petechia*). The dense thicket habitat type proved difficult for the researcher to navigate, and a limited amount of time was spent in this habitat. Two breeding landbird species were found exclusively in the dense thicket habitat: the American Pipit (*Anthus rubescens*) and the orange-crowned warbler (*Oreothlypis celata*). The two other habitat zones surveyed by Muldoon (1987) (Shore and Inlet) consisted of primarily shorebird and waterfowl observations.

Willson and Nichols (1997)

Willson and Nichols (1997) observed 23 breeding landbird species during surveys of the east arm of Glacier Bay in 1997 (Figure 42). Surveys were distributed across a vegetation gradient (Table 23) to relate avian distribution to vegetative successional stage. Species richness was lowest in habitats that had low vegetative cover (i.e., the first part of the successional gradient), and appeared to increase along the vegetation gradient (Figure 43). The highest species richness values occurred in Bartlett Cove, an area that was deglaciated approximately 250 years ago (Trautman 1966), while the lowest values were observed at Wachusett Inlet 1 (Appendix D, Figure 43), a recently deglaciated area (the inlet was glaciated as recently as 1929).

Table 23. Survey site names and habitat type sampled within each site by Willson and Nichols (1997). Sites are ordered from early succession (top of table) to later succession (bottom of table).

Site	Habitat Type
Wachusett Inlet 1	scattered alder, rock, and sand
Wachusett Inlet 4	willow alder scrub <30%
Adams Inlet 1	dry habitat - alder, willow, and openings
Wachusett Inlet 2	alder and willow - early deciduous
Wachusett Inlet 3	upland alder
Goose Cove	scrub willow and alder with openings
Adams Inlet 2	scattered scrub with cottonwoods and spruce
Stump Cove	alder willow scrub
Hunter Cove 1	young cottonwood
Hunter Cove 4	young cottonwood
Hunter cove 2	mature spruce and cottonwood
Hunter Cove 3	mix - old spruce, cottonwood, and alder
Ice Valley	mixed mature spruce and cottonwoods
Bartlett Cove	mature spruce

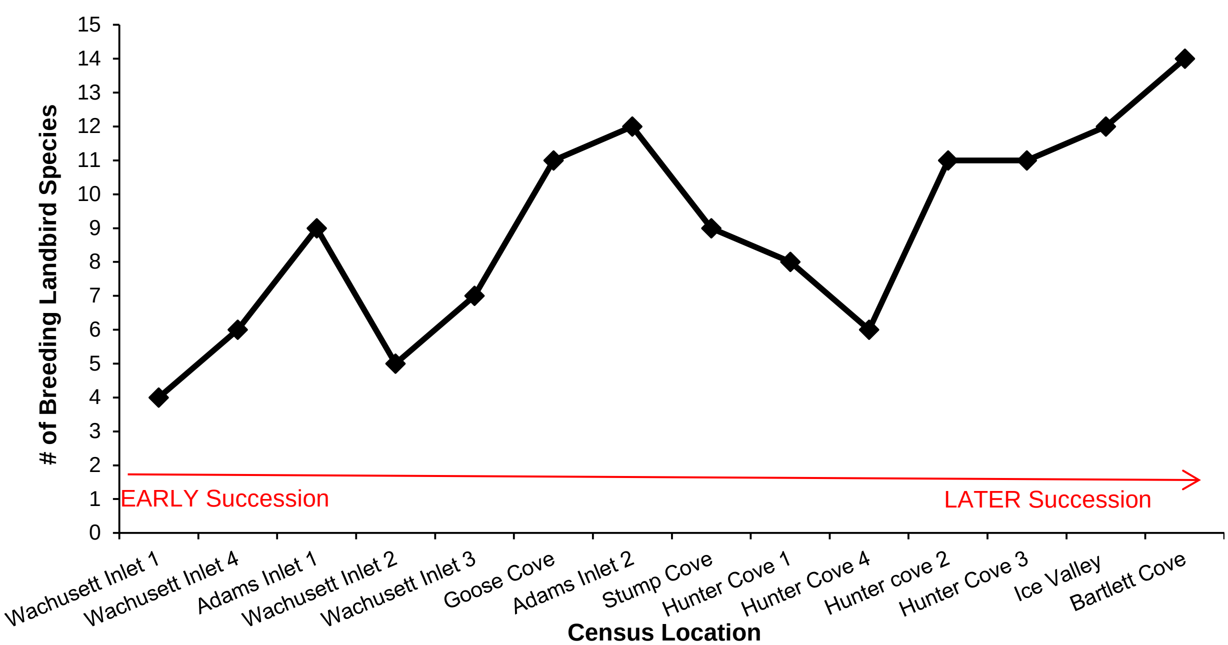

Figure 43. Species richness by survey site location during Willson and Nichols (1997). Census locations are arranged on the x-axis according to successional stages of the habitat (Willson and Nichols 1997).

Willson and Gende (1998)

Willson and Gende (1998) observed 24 species during surveys conducted in the first three weeks of June 1998 (Figure 42). Twenty-one species were observed at non-salmon streams in GLBA, with values ranging from 6 species (Wood Lake Creek West) to 14 species (Good River, Upper Wolf

Creek) (Table 24). Total species richness for streams with salmon in Willson and Gende (1998) was 22 species, with values ranging from eight species (Tyndall Point Creek) to 16 species (Bartlett River) (Table 24). Willson and Gende did not find any significant differences in species richness values between salmon and non-salmon streams ($p > 0.05$), and did not find any relationship between species richness and the percent of deciduous trees along the stream (t-test on regression line and intercepts had $p > 0.50$).

Table 24. Species richness and abundance of breeding landbirds observed during Willson and Gende (1998) surveys of salmon and non-salmon streams in GLBA (Willson and Gende 1998).

Site	Species Richness	# of Individuals
Salmon Streams		
Bartlett River	16	60
Geikie Rock Creek	13	60
Tyndall Point Headwaters	13	45
Crosscamp	9	43
Lower Wood Lake Creek	12	63
Tyndall Point Creek	8	43
Lower Wolf Creek	12	61
North Fingers Bay Creek	16	58
Non-salmon Streams		
Big Rock Creek	10	44
Good River	14	52
Wood Lake Creek East	7	25
Geikie End Creek	10	50
Upper Wolf Creek	14	41
Point Creek	8	37
Spokane Creek	12	54
Wood Lake Creek Tributary	11	45
Wood Lake Creek West	6	31
South Fingers Bay Creek	12	63
Non-salmon Stream Totals	21	442
Salmon Stream Totals	22	433
Grand Total	**24**	**875**

Saracco and Gende (2004)

Saracco and Gende (2004) observed 31 breeding landbird species during bird survey efforts in 2004 (Table 25; Figure 42). When including all observation distances (including flyovers), the Bartlett Lake Trail (BLTR) transect had the highest species richness value of any transect (17 species), while the Excursion Ridge (EXRI) transect had the lowest value (10 species; Table 25).

Table 25. Species richness and abundance of breeding landbirds observed during Saracco and Gende (2004). Values include observations made at all recording distances (including flyovers); acronyms are previously defined in Figure 41 (Saracco and Gende 2004).

Location	# of Species	# of Individuals
BLTR	17	100
FIBA	16	64
GEIN	18	112
CHIN	13	75
HUBA	12	80
EXRI	10	38
TLPO	15	90
MOCR	13	52
SEBE	12	51
VAHO	11	12
Total	**31**	**674**

Saracco and Gende (2004) used NMS to investigate variation in observed community composition along transects across the study region (complete NMS methodology available in Saracco and Gende 2004). The first two NMS axes accounting for the greatest variation in avian community composition corresponded to: 1) a vegetation gradient from older coniferous forests in the Lower Bay to the younger deciduous habitat of the Upper Bay (NMS axis 1); and 2) a geographical east-west gradient in the Bay (NMS axis 2). Species richness values (which included only species observed <50 m from the transect, excluding flyovers), increased along NMS axis 1 (i.e., species richness increased as transect locations transitioned to early deciduous forests), although this trend was not statistically significant (Saracco and Gende 2004). Two sites (Sealers Beach [SEBE], Van Horn [VAHO]) were notable outliers to this trend (Figure 44); these sites were located in early successional deciduous habitats yet produced relatively low species richness values (Figure 44). It should be noted that all NMS-related data were not adjusted to include only breeding landbird species, and a few species that are not considered landbirds (e.g., common merganser) are included in these figures and discussions.

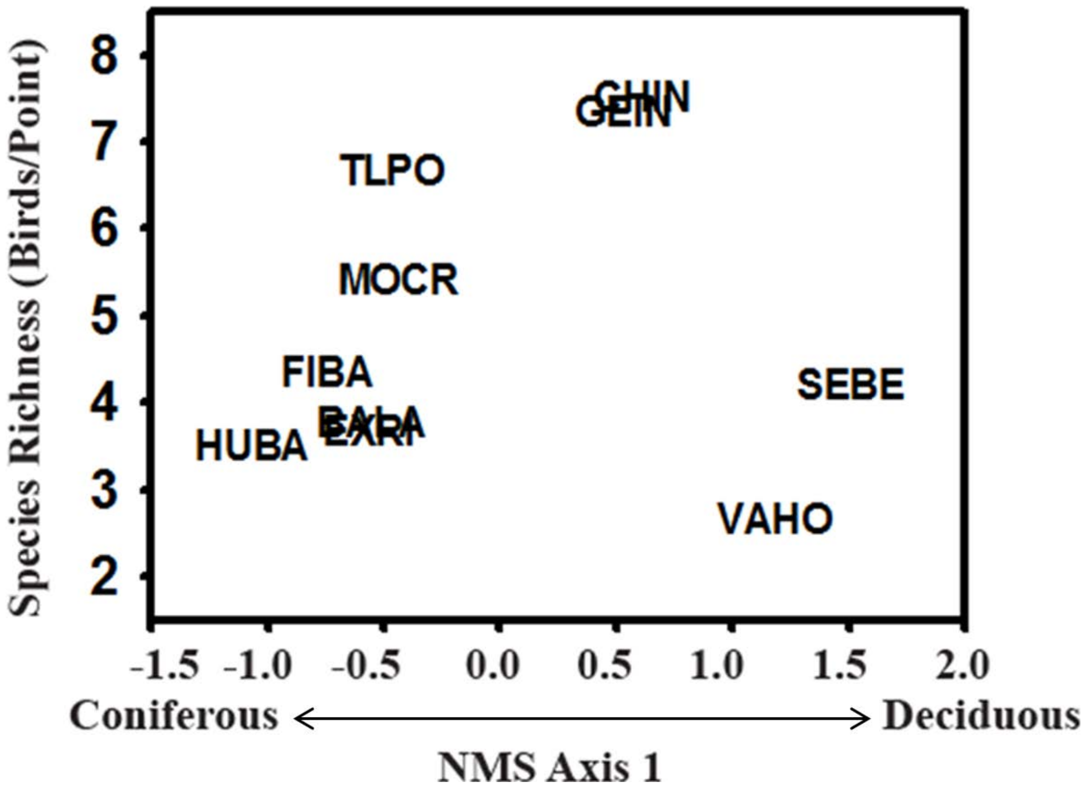

Figure 44. Scatterplot of species richness observed during Saracco and Gende (2004) against the NMS ordination axis 1. Acronyms follow the abbreviations in Figure 41; overlaid in the bottom left are BALA and EXRI, and in the upper right are CHIN and GEIN (Saracco and Gende 2004).

Species richness values also tended to increase along NMS axis 2 (i.e., as transects moved from east to west), although this trend was not statistically significant ($p = 0.11$; Figure 45) (Saracco and Gende 2004). Similar to NMS axis 1, only species observed within 50 m (164 ft) of a point on a transect were included in Figure 45.

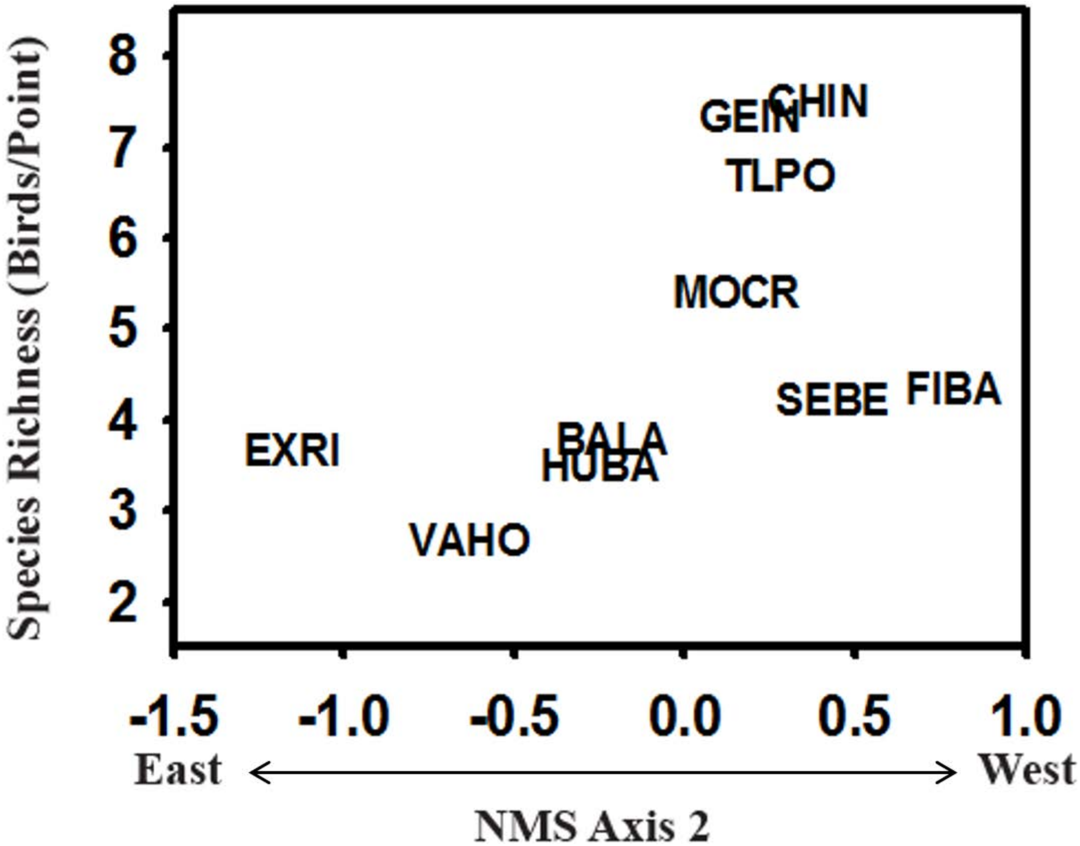

Figure 45. Scatterplot of species richness observed during Saracco and Gende (2004) against the NMS ordination axis 2 (r = 0.53), P = 0.11. Acronyms follow the abbreviations in Figure 41 (Saracco and Gende 2004).

<u>Species Abundance</u>

Species abundance documents how many breeding landbird individuals are documented in given survey/monitoring period. It needs to be noted, however, that all species have different detection probabilities, and measures of abundance reported here should be considered "naïve" estimates as they do not account for these variable detection probabilities.

Trautman (1966)

Trautman (1966) observed 5,439 individual birds of 43 species during an in depth survey of the Muir Inlet region of GLBA (Appendix E, Appendix F). The most frequently observed species during the survey included the common redpoll (*Acanthis flammea*, 962 individuals), fox sparrow (862 individuals), savannah sparrow (443 individuals), and the northwestern crow (*Corvus caurinus*, 422 individuals). These four individuals accounted for 49% of all breeding landbird observations during Trautman (1966).

Muldoon (1987)

Muldoon (1987) only documented species richness and distribution during the 1986 survey, and a discussion of this measure is not possible.

Willson and Nichols (1997)

Willson and Nichols (1997) documented 442 individual birds of 23 species during a 1997 survey of portions of the east arm of Glacier Bay. Peak abundance values appeared to increase as census sites progressed from early to late successional habitats, although Goose Cove (early-mid succession) had the highest abundance value (58 individuals) out of any site in the study (Figure 46). Bartlett Cove also had high abundance values (54 individuals), which was likely attributed to the later successional habitat, and the abundance of edge habitat for edge-specialist species (e.g., American robin [*Turdus migratorius*], dark-eyed junco [*Junco hyemalis*]).

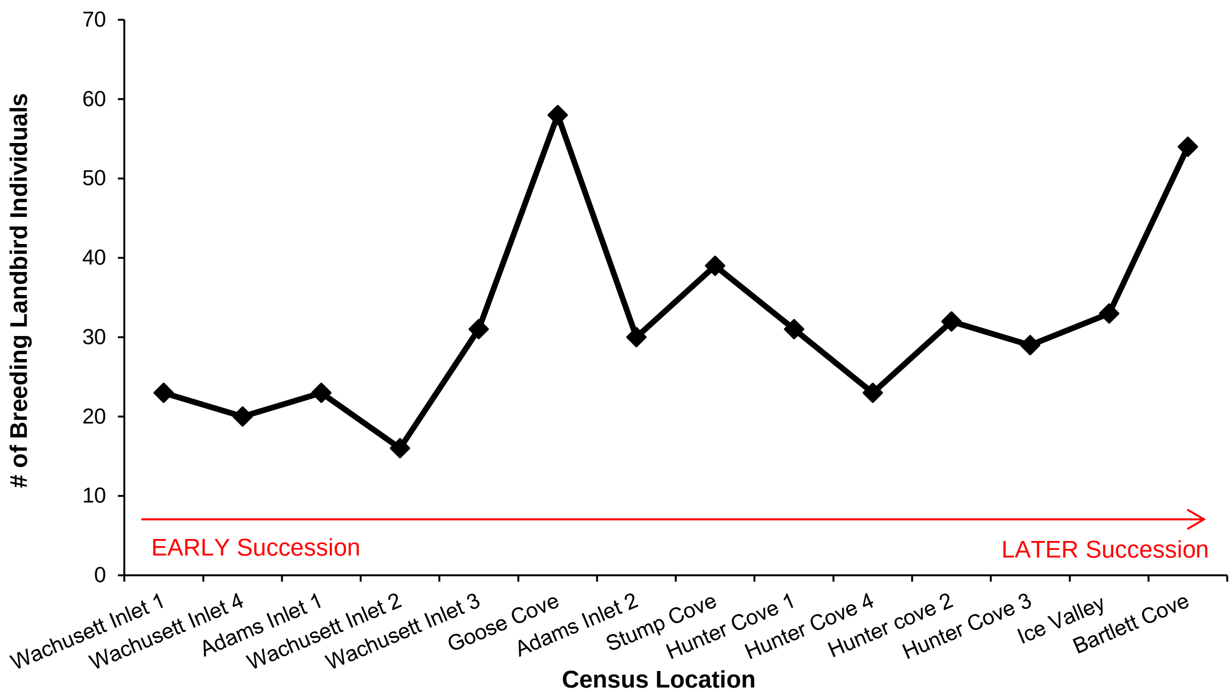

Figure 46. Breeding landbird abundance by survey site location during Willson and Nichols (1997). Census locations are arranged on the x-axis according to successional stages of the habitat (Willson and Nichols 1997).

The most commonly observed species during Willson and Nichols (1997) were the fox sparrow (91 individuals), Wilson's warbler (*Cardellina pusilla*, 51), hermit thrush (*Catharus guttatus*, 47), and the orange-crowned warbler (40) (Appendix D). These four species accounted for 52% of all breeding landbirds observed during Willson and Nichols (1997) (Appendix D).

Willson and Gende (1998)

During a 1998 survey of the mid-bay region of Glacier Bay, Willson and Gende (1998) documented 875 individual breeding landbirds of 24 species (Table 24). Comparisons between streams with and without salmon runs revealed no significant differences in abundance or densities (densities ranged from 5.1 to 8.9 birds/point/day, $p > 0.05$; Willson and Gende 1998). In streams that supported salmon runs, abundance values ranged from 43 individuals (Crosscamp, Tyndall Point Creek) to 63 individuals (Lower Wood Lake Creek) (Table 24), and the average number of individuals observed at a salmon supporting stream was 54.13. Abundance values in streams that did not support salmon runs ranged from 25 individuals (Wood Lake Creek East) to 63 individuals (South Fingers Bay Creek) (Table 24).

Across both study groups, the most commonly observed species were the Wilson's warbler (112 individuals), hermit thrush (106 individuals), yellow warbler (95 individuals), and the ruby-crowned kinglet (70 individuals). These four species accounted for 44% of all breeding landbird observations during Willson and Gende (1998).

Saracco and Gende (2004)

When including all observation distances (including flyovers), Saracco and Gende (2004) observed 674 breeding landbird individuals of 31 species. The most abundant species (i.e., most commonly observed) during the 2004 landbird surveys were the varied thrush (*Ixoreus naevius*), hermit thrush, Pacific-slope flycatcher (*Empidonax difficilis*), golden-crowned kinglet (*Regulus satrapa*) and the winter wren (*Troglodytes hiemalis*). These species were detected on more than 35 percent of the survey points that were sampled (Table 26). Table 26 identifies the frequency of occurrence for 25 species during the 2004 survey of 57 points in GLBA; only species that were detected within 50 m (164 ft) of a point are included in Table 26.

Table 26. Frequency of occurrence (proportion of survey points observed) for breeding landbirds during Saracco and Gende (2004). Table includes breeding landbirds observed within 50 m of a survey point. Table modified from Saracco and Gende (2004) to only include breeding landbird species.

Species	Frequency (proportion of points)
varied thrush	0.53
hermit thrush	0.49
Pacific-slope flycatcher	0.42
golden-crowned kinglet	0.39
winter wren	0.37
fox sparrow	0.32
ruby-crowned kinglet	0.3
yellow warbler	0.3
orange-crowned warbler	0.28
Wilson's warbler	0.28
chestnut-backed chickadee	0.23
pine siskin	0.19
gray-cheeked thrush	0.14
Townsend's solitaire	0.12
yellow-rumped warbler	0.12
American robin	0.11

Table 26 (continued). Frequency of occurrence (proportion of survey points observed) for breeding landbirds during Saracco and Gende (2004). Table includes breeding landbirds observed within 50 m of a survey point. Table modified from Saracco and Gende (2004) to only include breeding landbird species.

Species	Frequency (proportion of points)
common redpoll	0.07
pine grosbeak	0.05
red-breasted sapsucker	0.04
dark-eyed junco	0.04
rufous hummingbird	0.02
downy woodpecker	0.02
brown creeper	0.02
Swainson's thrush	0.02
Lincoln's sparrow	0.02

Similar to what was done for species richness, Saracco and Gende (2004) used NMS to investigate variation in observed abundance at transects across the entire study region. NMS axis 1 corresponded again to a vegetation gradient from older coniferous forests in the Lower Bay to the younger deciduous habitat of the Upper Bay, while NMS 2 correlated to an east-west gradient in the Bay. Abundance values in 2004 followed a similar pattern as the species richness values did, as the number of individuals per point increased as sample sites moved from older coniferous to younger deciduous habitats. Once again, VAHO and SEBE were notable outliers in this trend (Figure 47). Despite being located in an early successional deciduous habitat, VAHO had a comparatively low number of individuals observed during surveys (12 individuals). This may be partially explained by the presence of rain and thunderstorms during the survey that was conducted at this site.

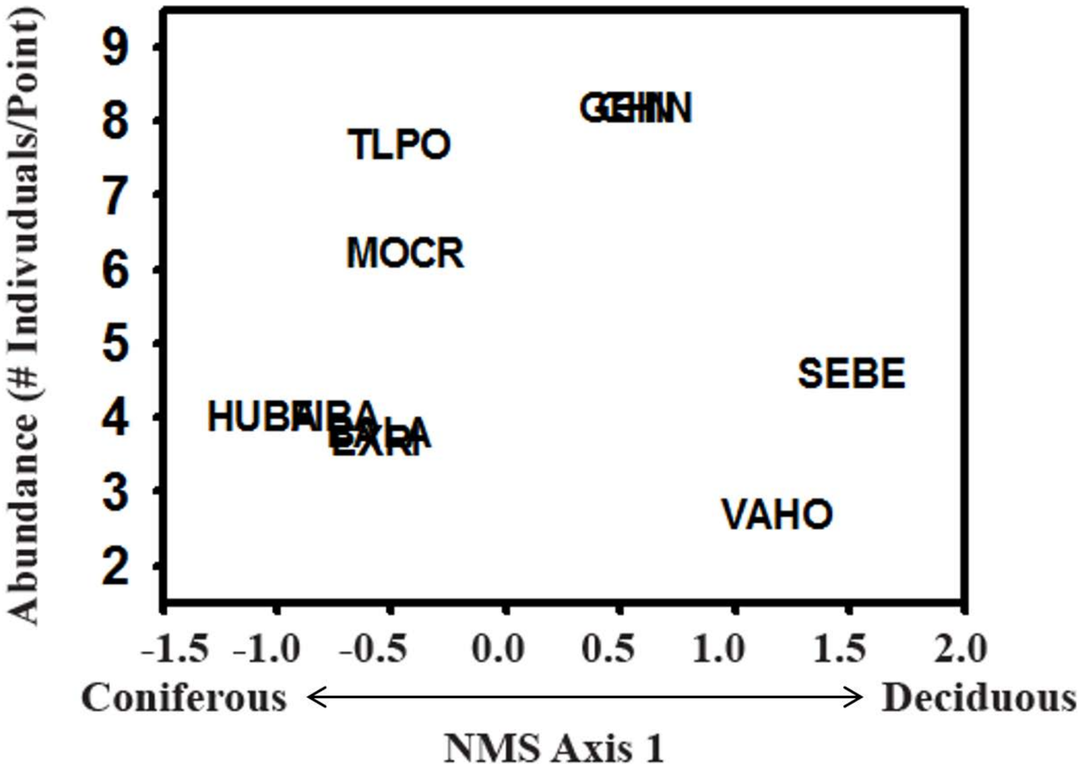

Figure 47. Scatterplot of abundance observed during Saracco and Gende (2004) against the NMS ordination axis 1. Acronyms follow the abbreviations in Figure 41, abbreviations overlaid in the bottom left are HUBA and FIBA (horizontally near 4 ind/pt) and BALA and EXRI vertically near 0.5 on NMS Axis 1). Overlaid in the upper left of the figure are GEIN and CHIN (Saracco and Gende 2004).

Species abundance appeared to increase along NMS Axis 2, but this trend was not significant (P = 0.14; Figure 48). Similar to NMS axis 1, only species observed within 50 m (164 ft) of a point on a transect were included in Figure 48.

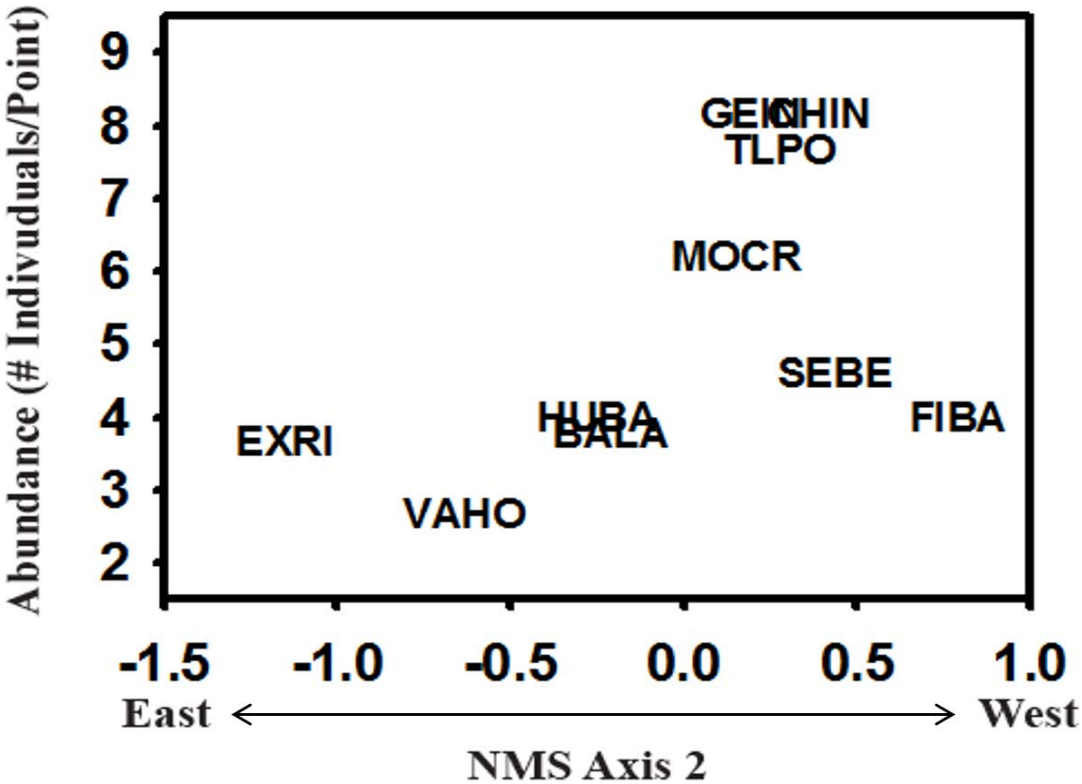

Figure 48. Scatterplot of abundance observed during Saracco and Gende (2004) against the NMS ordination axis 2 (*P* = 0.14); Acronyms follow the abbreviations in Figure 41 (Saracco and Gende 2004).

Trends in Species of Conservation Concern

For this measure, a species was considered a species of conservation concern if it appeared on one of the following conservation lists:

- USFWS Bird Species of Conservation Concern (BCC) for Bird Conservation Region (BCR) 5 (Northern Pacific Rainforest) (USFWS 2008);
- Listed by Partners in Flight (PIF) on the:
    - North American Landbird Conservation Plan (NALCP) (Rich et al. 2004);
    - Saving our Shared Birds (SOS) shared species list (Berlanga et al. 2010);
- USFWS Endangered Species List;
- Bureau of Land Management (BLM) Sensitive Species List;
- Audubon Alaska's WatchList.

Twenty-two breeding landbird species of concern (as defined by the lists above) have been observed within GLBA (Appendix G). However, without repeated breeding bird surveys that document changes in the presence/absence or abundance of priority species in the GLBA area, current trends in species of conservation concern cannot be determined at this time. The most recent survey data from GLBA is over 10 years old (Saracco and Gende 2004); with the constantly changing and evolving habitats in the park, more recent data are needed to accurately assess this measure.

Threats and Stressor Factors

GLBA identified several threats and stressors that are likely to affect the breeding landbird community in the park, these threats and stressors included: climate change, externally derived environmental contaminants, diseases, sources of direct mortality, and habitat alteration and destruction during the non-breeding season (for migratory breeding landbirds). Climate change is one of the major forces affecting bird communities across the globe; this threat is becoming better understood as research and data continue to become available. Changes in the temperature and precipitation norms in the park could have both direct and indirect effects on the breeding bird community of GLBA. Examples of direct impacts to the breeding landbird community would include shifts in the timing of spring plant phenology, and an altered snowmelt pattern in breeding locations in the park. Indirect impacts resulting from shifts in temperature and precipitation could include effects on the frequency, extent, and severity of insect outbreaks, particularly the spruce bark beetle (*Dendroctonus rufipennis*). These insect outbreaks often have lasting effects on communities, as tree mortality can influence the overall habitat structure and species composition of areas for many years.

Another threat facing breeding landbird populations is the shifting of species' reproductive phenology. Several bird species depend on temperature ranges or weather cycles to cue their breeding. As global temperatures change, some bird species have adjusted by moving their home range north (Hitch and Leberg 2007). Other species have adjusted their migratory period and have begun returning to their breeding grounds earlier in the spring; American robins in the Colorado Rocky Mountains are now returning to their breeding grounds 14 days earlier compared to 1981 (NABCI 2009). A concern is that this shift in migration may be out of sync with food availability and could ultimately lead to lowered reproductive success and population declines (Jones and Cresswell 2010).

The North American Bird Phenology Program (BPP) is currently analyzing the migration patterns and distribution of migratory bird species across North America (USGS 2008). Information from this analysis will provide new insights into how bird distribution, migration timing, and migratory flyways have changed since the later part of the 19th century. This information may also be applied to estimate changes in breeding initiation periods in specific habitats.

Externally derived environmental contaminants affect almost every ecosystem in GLBA, but little is known about how these contaminants specifically affect the breeding landbird community of GLBA. The sources of these contaminants can be distant, as Jaffe et al. (1999) notes that Asian anthropogenic emissions make up a significant source of air pollution that arrives in North America in the spring. The air quality and pollution in GLBA is discussed in depth in Chapter 4.21 of this report. Water quality and pollution is also of concern to the breeding landbird community. Water quality and pollutants are discussed in detail in Chapter 4.22 of this report. An additional water-related issue that may exist in GLBA is tied to the availability of nutrients in the marine environment. A theory is that the success and health of the terrestrial communities (including birds) is tied closely to the health of the salmon runs in the rivers and streams (Gende et al. 2002). This was briefly investigated in GLBA by Willson and Gende (1998); however, this study used only 1 year of data and did not find a difference in avian density when comparing salmon and non-salmon streams. A decline in the number of streams that support salmon runs, a decline in the number of returning salmon, or both, could contribute to the overall health of the breeding landbird communities in these riparian areas.

Disease is a threat common to most ecological communities; however, not much is known about the effects of disease in wild bird populations. West Nile Virus has been linked to population declines in wild birds in recent years, with the Corvid family being particularly hard hit. LaDeau et al. (2007) indicated that the American crow (*Corvus brachyrhynchos*) population has declined up to 45% since West Nile Virus was first discovered. Monitoring of population trends in the park will be important to identify species that may potentially be vulnerable to disease outbreaks. Instances of suspected outbreaks could be sampled for disease if needed. A large number of birds in south-central Alaska have been documented with beak abnormalities (USGS 2014); however, the northwestern crow is the only one of these species to have been documented in relatively high numbers in Southeast Alaska. The exact cause of these deformities has not been determined, although researchers have investigated nutritional deficiencies, disease/parasites, and genetics as causes and have found limited to no evidence. There has been some evidence that contaminants may play a role, although more research is needed (USGS 2014).

Migratory breeding landbirds are faced with additional threats during the migratory periods, such as direct mortality due to collisions with man-made structures, predation by feral/stray cats, and habitat alteration and destruction at locations outside of the breeding area. Recent efforts to develop alternative energy sources have resulted in more wind farm development across the planet (de Lucas et al. 2008). Collisions with wind farms are likely more frequent among raptors and Neotropical migrants. However, the exact effects that these wind farms have on birds are still poorly understood. Some studies have found that wind farms are responsible for no more mortalities than other human-made structures (e.g., buildings, communication towers) (Osborn et al. 2000), while other studies have found that turbines are responsible for unusually high numbers of bird mortalities (Smallwood and Thelander 2007). Bird collisions with buildings, power lines, communication towers, and windows are one of the top anthropogenic threats to birds and may result in between 365-988 million bird deaths across North America (Loss et al. 2014). While there are few buildings and towers in the immediate GLBA area, species migrating to and from the park are sure to encounter such obstacles during annual migrations.

Domestic cats (*Felis catus*) are one of the largest causes of bird mortality in the United States. According to Loss et al. (2012), annual bird mortality caused by outdoor cats is estimated to be between 1.4 and 3.7 billion individuals. The median number of birds killed by cats was estimated at 2.4 billion individuals, and almost 69% of bird mortality due to cat predation was caused by un-owned cats (i.e., strays, barn cats, and completely feral cats) (Loss et al. 2012).

One of the major threats facing landbird populations across all habitat types is land cover change (Morrison 1986). Land cover change is not restricted to the breeding habitat; many species depend on specific migratory and wintering habitat types that are also changing. Most of the birds that breed in the United States spend winters in the Neotropics (MacArthur 1959); deforestation rates in these wintering grounds have occurred at an annual rate up to 3.5% (Lanly 1982). While forest and habitat degradation does occur in the United States, it does not approach the level of degradation seen in the tropics (WRI 1989). Furthermore, Robbins et al. (1989) supported the suggestion that deforestation in the tropics has a more direct impact on Neotropical migrant populations than deforestation and habitat loss in the United States.

Data Needs/Gaps

There is currently no annual monitoring program that documents the presence, abundance, trends, or demography of breeding landbirds in GLBA. The establishment of an annual monitoring program would greatly benefit park managers, and would provide valuable information regarding the overall health of the Glacier Bay area, and would also provide important abundance information regarding the potential species of conservation concern in the area. Sampling in GLBA is likely problematic due to the variety of habitats and the sheer size of the park, but a monitoring program would likely need to balance providing a representative sample frame of interest and be able to be implemented practically over time. Most of the park is difficult to access, but the sampling points should cover a large region that covers a variety of habitat types and not just the areas that are most accessible.

As has been documented by several studies in the park, the younger deciduous forests in the park have higher species richness values (Willson and Gende 1998, Saracco and Gende 2004). Continuation and expansion of the survey work done by Saracco and Gende (2004) could provide valuable insight into this trend, and potentially identify critical habitat areas for nesting landbird species in the park.

Christmas Bird Counts have taken place in the Gustavus, AK region since 1969, but these counts occur during the winter and do not survey many of the breeding species that come to the area in the spring and summer months. The establishment of a BBS in the Gustavus region would help to fill the current data gap regarding breeding landbird species richness and abundance in the Glacier Bay area, but is likely not possible due to the average length of these routes exceeding the available road lengths in the area. An analogous survey or route in the area that uses a comparable methodology at a smaller scale is one alternative possibility.

Overall Condition

Species Richness

The GLBA project team assigned the species richness measure a *Significance Level* of 3 during project scoping. After being adjusted to only include breeding landbird species (see Section 4.6.4 above), the NPS Certified Bird Species List identifies 67 breeding landbird species that occur in GLBA.

There have been five primary bird surveys of that have occurred in GLBA since 1965 (Appendix C). These surveys sampled different areas of the park, which makes comparisons between them difficult. Further complicating comparisons is the fact that survey methodologies has varied greatly between all studies. Trautman (1966) represents one of the most intensive bird surveys conducted in the park, and this study produced the highest species richness value on record (43 species). However, this study included repeat visits to transects, which has not been duplicated in other studies in the park. The most recent work completed in the park, Saracco and Gende (2004), sampled both coniferous and deciduous forests of eastern and western GLBA and recorded the second-highest breeding landbird species richness value for the park (31 species). Surveys conducted in the late 1980s and late 1990s reported breeding landbird species richness values ranging from 18-24, with an average value of 21.67 species/study.

There do not appear to be major concerns in GLBA regarding the current species richness of breeding landbirds. However, the most recent data source is >10 years old, and will likely no longer be indicative of the current status in species richness for the park. It is apparent that many breeding landbird species in the park are drawn to the younger deciduous forests (Willson and Gende 1998, Saracco and Gende 2004), and the habitats and point count locations sampled in 2004 by Saracco and Gende (2004) may no longer show the same patterns that they did >10 years ago. Additional research is needed that investigates the relationships between species richness and the age class and east/west orientation of sampling sites. Until more recent data becomes available, this measure cannot be accurately assessed; a *Condition Level* was not assigned to this measure.

Species Abundance

Species abundance was assigned a *Significance Level* of 3 during initial project scoping. While many of the summarized sources in this document recorded abundance, these data are out of date, with the most recent source being over 10 years old. Because of the lack of recent data, a *Condition Level* was not assigned to the species abundance measure (*Condition Level* = N/A).

As was mentioned, most of the studies that have occurred in GLBA have documented breeding landbird abundance. Trautman (1966) documented the most birds out of any study in GLBA (5,439 individuals); however, this study was also the longest in duration and included repeat sampling of transects, which may explain the high number of individuals observed. Diversity in the park is likely to be affected by habitat type, as breeding landbirds in the park appear to have higher species richness and abundance values in habitats that are earlier deciduous habitats (Willson and Gende 1998, Saracco and Gende 2004). This trend needs further research, particularly in regards to an expansion of survey sites to areas that have not previously been sampled in the park.

Trends in Species of Conservation Concern

The trends in species of conservation concern measure was assigned a *Significance Level* of 2 during project scoping. Appendix G identifies the breeding landbird species of conservation concern that have been confirmed in the park; however, this list is likely to change as additional species may be identified in the park and different conservation lists can be consulted. Current condition for this component was not determined (*Condition Level* = N/A), as there have been no recent breeding landbird surveys in the park. The 2004 surveys conducted by Saracco and Gende (2004) represent the most recent survey work that has been completed in GLBA. The dynamic nature of the GLBA ecosystem means that bird communities are continually changing in terms of distribution and abundance. As new habitat becomes available, or as habitats mature, habitat-specialist bird species are likely to shift. In order for this measure to be accurately assessed, a survey within the last 5 years would be optimal.

Weighted Condition Score

Due to a lack of recent data, a *Weighted Condition Score* was not assigned to the breeding landbirds component. While trends observed from the limited studies from 1965-2004 do not appear to indicate major changes in the community, a lack of continuity in the survey period, coupled with a lack of recent (within 10 years) data prevent a trend analysis at this time.

Breeding Landbirds			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Species Richness	3	n/a	
Species Abundance	3	n/a	
Trends in Species of Conservation Concern	2	n/a	

4.6.6. Sources of Expertise

- James Saracco, The Institute for Bird Populations Research Ecologist
- Tania Lewis, GLBA Wildlife Biologist.

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4.7. Glaucous-Winged Gulls

4.7.1. Description

The glaucous-winged gull is a common species along the northwestern coast of North America. The species' breeding range extends from Washington to the Alaska Peninsula and Aleutian Islands, and glaucous-winged gulls are year-round residents in GLBA. Glaucous-winged gulls are a large gull species and can range from 50-59 cm (19.7-23.2 in) in length, and weigh around 1,010 g (2.23 lbs) (Verbeek 1993, Sibley 2000). The glaucous-winged gull is an opportunistic feeder and frequently obtains food from the surface of the water or from along coastlines; it is not uncommon to observe this species stealing food from other gulls or shorebird species. In GLBA, the glaucous-winged gull appears to favor a diet that consists of primarily mussels, fish, urchins, and various gastropods (Zador and Piatt 1999).

Photo 19. The glaucous-winged gull, a common colonial nesting species found in GLBA (NPS photo).

The nesting habits of the glaucous-winged gull were studied at South Marble Island from 1999-2001 (Zador and Piatt 1999, Zador 2001, Zador and Piatt 2002) and a comprehensive shoreline survey of ground nesting birds (including glaucous-winged gulls) was conducted from 2003-2005 (Arimitsu et al. 2007). In the park, glaucous-winged gulls usually nest in concentrated colonies, with nests in close proximity to each other (Arimitsu et al. 2007). Colonies are located near un-forested shorelines, and are often hidden in ryegrass (*Leymus arenarius*) or built in scrapes on the shoreline's bedrock or cliffs (Zador 2001, Arimitsu et al. 2007).

Similar to other large gull species, the glaucous-winged gull is an indeterminate egg layer (Parsons 1976, Zador 2001). Indeterminate layers will replace lost eggs in a clutch, typically due to egg loss caused by predation or flooding, until the clutch contains a certain number of eggs; in glaucous-winged gulls, eggs will be laid every other day with laying typically terminating when the clutch reaches three eggs (Zador 2001, Zador and Piatt 2002). Once laying has initiated, the incubation period for the species is approximately 27 days (Verbeek 1993). A clutch may be replaced if an egg(s) is lost after incubation has begun; however, this process is energy and time consuming for the species, and it typically takes 12-13 days to resume follicle growth to initiate the laying of another egg (Zador and Piatt 2002).

Recently, efforts have taken place that would allow some harvest of glaucous-winged gull eggs in GLBA by the native Huna Tlingit tribe. The Huna Tlingit tribe's traditional territory included all of present-day GLBA (Hunn et al. 2002), and the tribe had a rich tradition of gull egg harvests in the park up until 1974 when restrictions on harvest began to be enforced. Harvests would occur on well-populated islands, and would occur up to two times early in the breeding season (before 15 June; NPS 2010b).

4.7.2. Measures

- Distribution (presence in a location)
- Population trends in select areas (Tlingit Point, South Marble, Lone Island, Geikie Rock, Flapjack, Boulder Island)
- Number of harvested eggs by location

4.7.3. Reference Conditions/Values

Park-wide monitoring of glaucous-winged gull colonies began recently (2012), and for many colonies and metrics an appropriate reference condition(s) does not exist. Data collected by GLBA staff during annual surveys will likely serve as a future reference condition for this resource. With the high probability of a gull egg harvest in the near future, gull population data from post-harvest periods can be compared to pre-harvest data to determine any trends in abundance, reproductive success, or distribution.

When discussing the number of nesting adults observed at a colony and the number of nests observed at a colony, the results from Arimitsu et al. (2007) will be used for all colonies except for South Marble Island, which will use Zador and Piatt (1999) and Zador (2001) as reference conditions as those studies predate Arimitsu et al. (2007). The surveys conducted by Arimitsu et al. (2007) in 2003 and 2005 counted adult glaucous-winged gulls using a comparable methodology to current GLBA surveys. Zador and Piatt (1999) and Zador (2001) performed a detailed survey of South Marble Island in 1999 and 2000, and the number of adults observed during those studies will serve as the reference condition for the number of nesting adults portion of the population trends measure.

4.7.4. Data and Methods

Zador and Piatt (1999) investigated the population size and productivity of seabirds at South Marble Island in GLBA from May to June 1999. Most of the data were collected during ground surveys and observations; two researchers were stationed on South Marble Island for 48 days in 1999, and opportunistic observations were also recorded during trips to and from the island. In addition to the ground surveys, Zador and Piatt (1999) used vessel surveys to census the glaucous-winged gull population of the island. Vessel surveys took place before egg laying began (mid-late May) and occurred during high tide.

Zador (2001) focused on glaucous-winged gull egg loss on South Marble Island, and investigated how the removal of eggs from a clutch on the island may affect reproductive and physiological responses of the species. Nesting areas were monitored from mid-May to late-June in 1999 and 2000. Data collected during nest observations included, but were not limited to, the number of nests, number of eggs, and date of first egg appearance. When possible, eggs were numbered in a nest based on the sequence that they were laid. In addition to monitoring nesting and reproductive success on the island, Zador (2001) also investigated the effects that a controlled egg harvest may have on the island's gull population. Three treatments were used during the egg harvest study: 1) the first egg laid in a nest was removed on the same day it was laid, 2) an entire clutch of three eggs was removed from a nest on the day the third egg was laid, and 3) no eggs were removed.

Arimitsu et al. (2007) documented the nesting distribution and abundance of several shoreline nesting bird species in GLBA from 2003-2005. Each survey year had a unique focus: 2003 surveys focused on areas of high visitor use (sites with >30 overnight visits from 1995-2002), 2004 surveys included the entire shoreline of the park (except for the Beardslee Islands), and 2005 surveys focused only on nesting areas identified in 2003-2004. High visitor use areas were surveyed from the ground, while areas that were classified as low visitor were surveyed from vessels approximately 3-15 m (10-49 ft) from shore. If nesting was observed during a vessel survey, observers would move the vessel to land and continue the survey on foot.

Initial glaucous-winged gull monitoring efforts began in GLBA in 2012 and have continued annually (data are current through 2015). Monitoring of the populations in the park is completed using repeated vessel and ground surveys, with surveys occurring from May-August. The frequency and duration of surveys are often affected by hauled out marine mammals; species such as harbor seals, Steller sea lions, or sea otters are protected under the Marine Mammal Protection Act of 1972 (16 U.S.C. §§ 1361-1407 [1994]) and the Endangered Species Act of 1973 (16 U.S.C §§1531 et seq.) and cannot be disturbed by vessels approaching their location. Nesting islands are usually accessed by vessel two to six times a year. Ground surveys consist of two to four visits per island and researchers attempt to determine the date of initial egg laying. Additionally, ground surveys attempt to complete a census of the nesting population (distribution, abundance, number of eggs). Vessel surveys occur directly before or after ground surveys, and identify the number of nesting adults and the number of nests visible at the sites.

4.7.5. Current Condition and Trend

Distribution

Glaucous-winged gulls are widely distributed in GLBA during the non-breeding season, as individual gulls can be observed at nearly every island or coastline in the park. During the breeding season, however, glaucous-winged gulls in GLBA congregate into relatively large nesting colonies which can range in size from a couple of pairs to over 1,000 pairs (Zador and Piatt 1999, NPS 2014b). Colonies are most frequently located on islands, and appear to be most successful when the island is located 1.6 km (1 mi) away from the mainland (Tania Lewis, GLBA Wildlife Biologist, phone conversation, 6 November 2014). Colonies in GLBA favor particular habitat types; Arimitsu et al. (2007) noted that most glaucous-winged gull colonies favored ryegrass or bedrock and were located in close proximity to the shoreline. Habitat selection by gulls can directly affect reproductive success and fitness (Pierotti 1982, Garcia-Boroglu and Yorio 2004). Predation, combined with any reductions in optimal nesting habitat could lead to decreased abundance and productivity at gull colonies (Cowles et al. 2012).

The availability of optimal nesting habitat is constantly changing in GLBA. Glacier Bay is a recently deglaciated ecosystem (approximately 250 years ago), and the ongoing vegetational succession and isostatic rebound continually creates and removes nesting habitat for the gulls. Ecological succession occurring on the islands in the park have resulted in many historic nesting locations transitioning from the gull-preferred open meadows and rocky habitats to closed conifer forests, which are not suitable for nesting colonies. This process is perhaps most evident at North Marble Island, where during the 1972 and 1973 breeding seasons, Patten (1974) observed roughly 1,000 breeding glaucous-winged gulls. Approximately 30 years later, the island had transitioned into a closed canopy conifer forest, and Arimitsu et al. (2007) observed nine gulls and only one nest.

Hunn et al. (2002) identified 25 sites in the park that supported glaucous-winged gull colonies that were historically large enough to support Huna Tlingit egg harvests in the 19th and early 20th centuries. Current NPS monitoring efforts indicate that there are only six of those colonies support a substantial number of nesting pairs (Boulder Island, Flapjack Island, Geikie Rock, Lone Island, South Marble Island, and Tlingit Islet; Figure 49) (NPS 2014a, b).

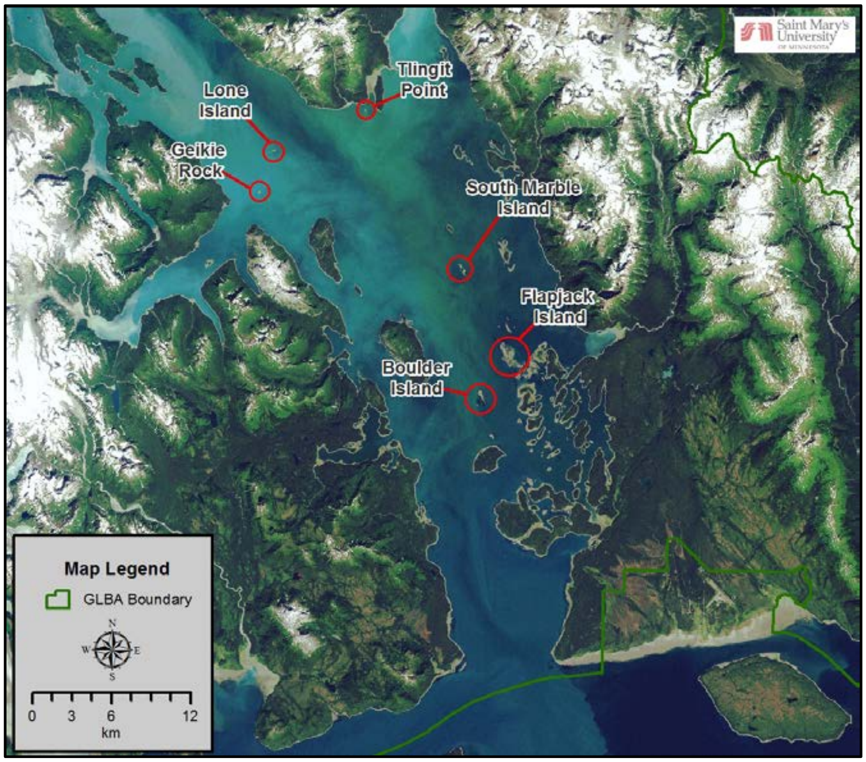

Figure 49. The six priority glaucous-winged gull nesting locations surveyed annually by GLBA staff (NPS 2014a, b).

Several isolated locations in the park have supported low numbers of nesting glaucous-winged gulls. In 2004, the north shore of Muir Inlet had relatively low numbers of nesting gulls on a mainland coastal cliff (77 adults and one nest observed; Arimitsu et al. 2007). GLBA surveys in 2012 recorded 26 adults and two nests at the site, and by 2014, there were no observations of adults at the site. Some notable locations that have historically supported gull colonies (or still support very few nesting gulls) include Leland Island, Sealers Island, Hugh Miller Inlet, and Sturgess Island.

<u>Population Trends in Select Areas (Tlingit Islet, South Marble Island, Lone Island, Geikie Rock, Flapjack, and Boulder Island)</u>

GLBA has identified six priority areas for glaucous-winged gulls: Tlingit Islet, South Marble Island, Lone Island, Geikie Rock, Flapjack Island, and Boulder Island. These areas are monitored annually with the goal of collecting necessary biological data to inform potential yearly gull harvests (NPS 2014b). Recently, the Gull Use Act has passed Congress. This Act allows native harvest of glaucous-winged gulls in the park, and these priority islands represent areas that support nesting colonies that may be large enough to support an egg harvest by the Huna Tlingit tribe. Legal harvest will not occur in the park until the park can promulgate regulations.

The metrics used to determine population trends at these islands include: number of nesting adults, number of observed nests, number of eggs, and number of eggs/nest at peak incubation. Of the reported metrics, the number of observed nests is often viewed as the best indicator of population trends at a colony, while the number of eggs per nest is likely the best indicator of yearly productivity

(Lewis, written communication, 24 April 2015). The total number of adults and total number of eggs may fluctuate greatly by year and time of year, which makes comparison between years/surveys difficult. Data were collected from 2012-2014 during the annual GLBA glaucous-winged gull surveys, and includes both vessel and ground survey results. In many instances, the value reported represents the highest value or count observed during monitoring efforts in a given year. This measure uses data summarized from both GLBA monitoring and from Arimitsu et al. (2007). While the two sources of data report population statistics, they are not always comparable due to differences in sampling timing.

Tlingit Islet

Arimitsu et al. (2007) was the first study to document the number of nesting adults near Tlingit Islet when the authors observed 28 nesting adults in 2005 (Figure 50). The number of nesting adults was not recorded at this location again until 2012, when the GLBA glaucous-winged gull monitoring project began. The number of nesting adults at Tlingit Islet remained relatively consistent from 2012-2014, with an average of 53 nesting adults observed each year. The high count for observed adults each year ranged from 40 (2013) to 68 adults (2012) (Figure 51).

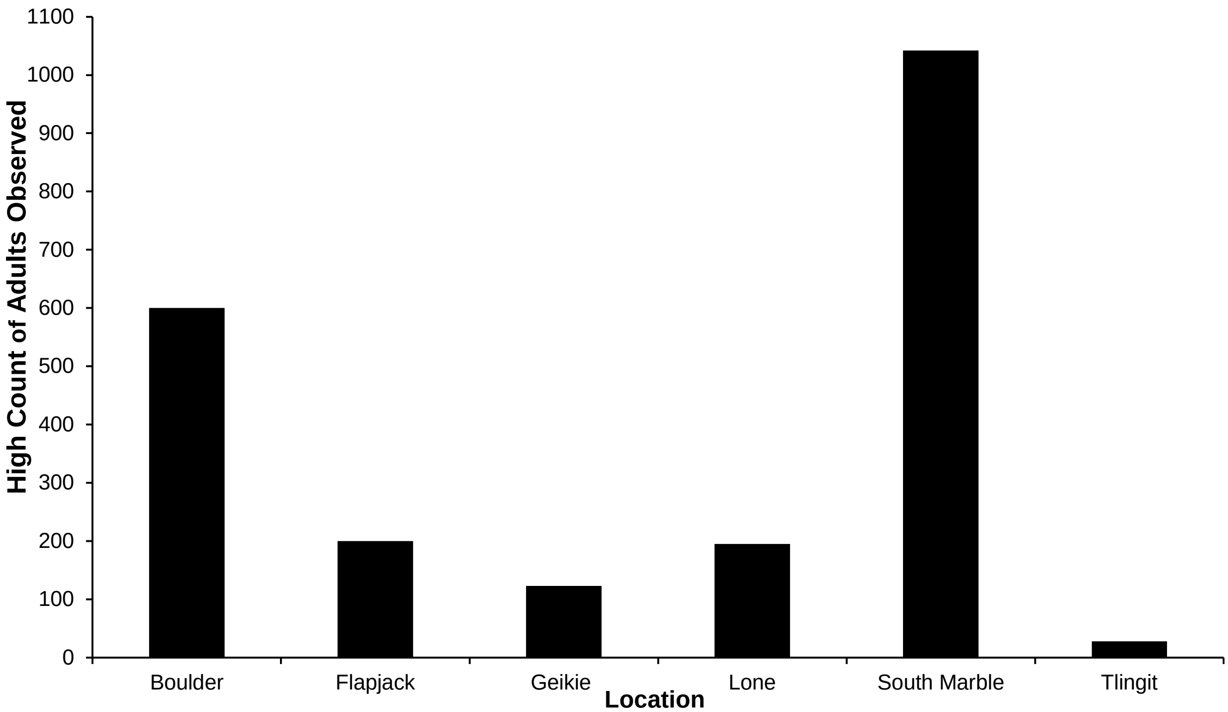

Figure 50. High count of number of glaucous-winged gull adults observed in GLBA during 2003 or 2005 surveys (Arimitsu et al. 2007).

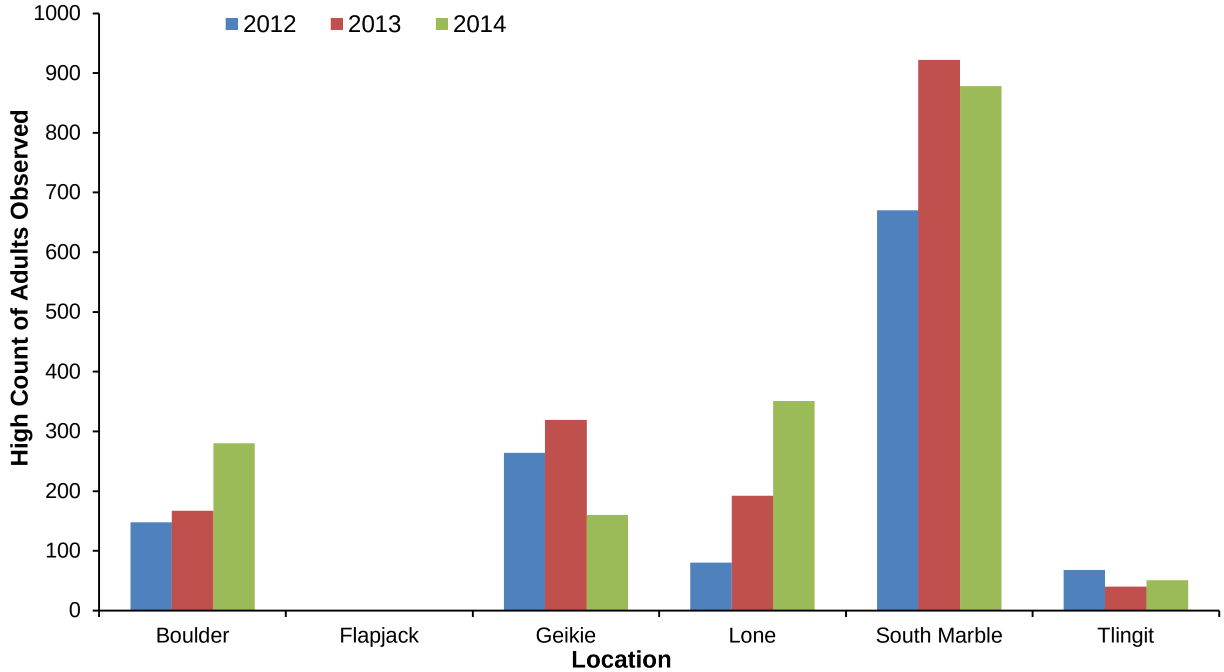

Figure 51. High count of number of glaucous-winged gull adults observed in GLBA during annual monitoring in the park from 2012-2014. Vessel surveys are not possible at Flapjack Island due to reefs that surround the island (NPS 2014a).

Arimitsu et al. (2007), which serves as the reference condition for this measurement, documented four nests at Tlingit Islet in 2005 (Figure 52). GLBA monitoring documented a high count of 51 nests in 2012 and this number has declined during each year of the survey (31 in 2013 and 22 in 2014) (Figure 52).

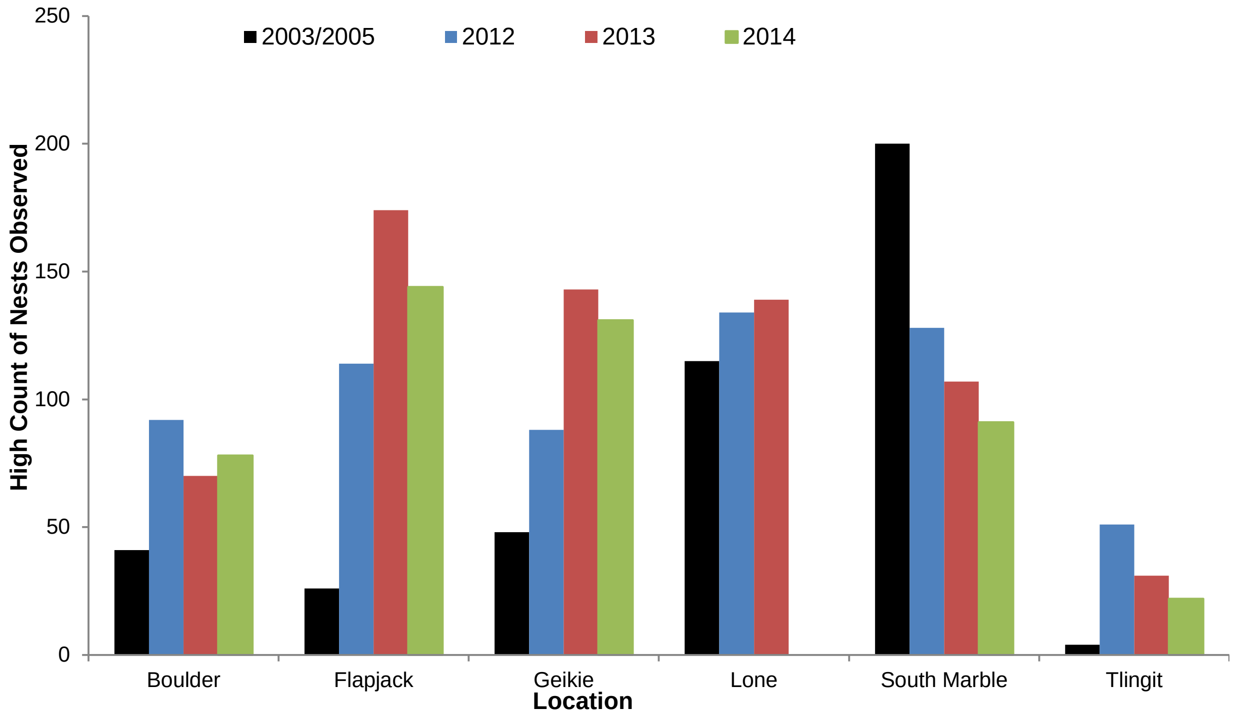

Figure 52. The high counts of glaucous-winged gull nests observed in GLBA during annual monitoring from 2012-2014 (NPS 2014a). The 2003/2005 values represent the reference condition from Arimitsu et al. (2007).

The GLBA annual monitoring project was the first to document the number of eggs at the Tlingit Islet site, and a high count of 11 eggs was reported in 2012. This number increased to 32 eggs in 2013 and to 56 eggs in 2014. 2012 was a record year in regards to precipitation and cold temperatures (Lewis, phone conversation, 6 November 2014), and record low egg counts were observed at each of the priority sites in the park (Figure 53).

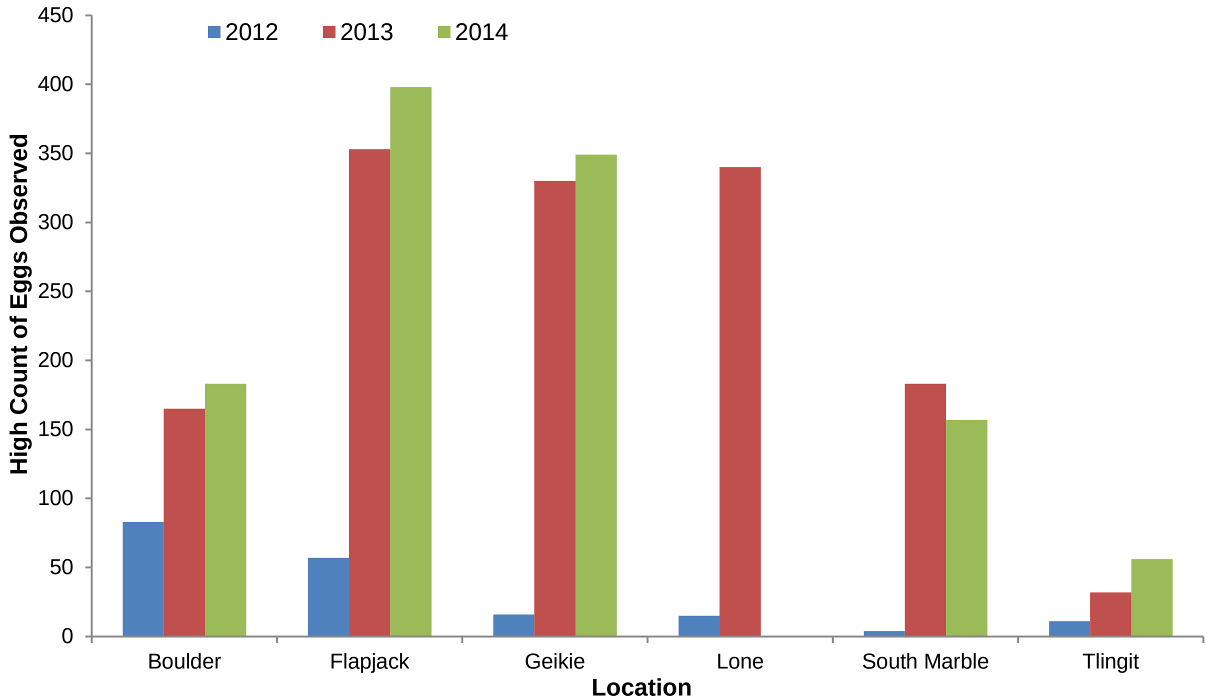

Figure 53. The high counts of glaucous-winged gull eggs observed in GLBA during annual monitoring from 2012-2014 (NPS 2014a).

Using the high count of eggs and nests at a location in a given year, it is possible to estimate the number of eggs per nest at the peak of incubation. These estimates were created for each nesting colony using only the GLBA monitoring data and are presented in Figure 54. Yearly averages were also created that summarized the annual number of eggs per nest park-wide (represented by a dashed line in Figure 54). Tlingit Islet had low numbers of eggs per nest from 2012-2013, but egg numbers rebounded to 2.545 eggs/nest in 2014. Values in 2012 and 2013 fell below park-wide averages, while 2014 was slightly above average (Figure 54).

Figure 54. The number of glaucous-winged gull eggs per nest observed during peak incubation at the six priority nesting locations. Dashed lines correspond to the yearly average of eggs/nest at peak incubation during each year. Data collected during GLBA annual monitoring from 2012-2014 (NPS 2014a).

South Marble Island

The breeding of glaucous-winged gulls has been studied and observed at South Marble Island more frequently than any of the other islands in the park (Jewet 1942, Patten 1974, Zador and Piatt 1999, Zador 2001, Arimitsu et al. 2007). South Marble Island has been deglaciated for approximately 170 years (Zador and Piatt 1999), and is composed of low cliffs and spruce forests on the western half of the island, and grassy hilltops and steep slopes on the eastern half of the island. The island itself is 9.7 km^2 (3.7 mi^2), and is surrounded by deep water reaching over 250 m (820 ft) in depth.

The earliest record of breeding adults on the island was in 1941, when Jewet (1942) documented 200 breeding adults with eggs and young present. Sowls et al. (1978) reported 550 breeding gulls on the island in 1973, and from 1972-1973, Patten (1974) documented approximately 1,000 breeding gulls on South Marble Island.

As part of an intensive glaucous-winged gull research project on South Marble Island, Zador and Piatt (1999) spent several weeks on the island observing the abundance and reproductive behavior of glaucous-winged gulls from May to June of 1999. The number of gulls observed on the island during the study remained high, as the highest count of breeding adult gulls was 829 gulls (Zador and Piatt 1999).

Arimitsu et al. (2007) documented a maximum count of 1,042 nesting adults on South Marble Island in 2005, which represents the highest count of adults on record for this island (Figure 50). The number of nesting adults was not recorded at this location again until 2012, when the GLBA glaucous-winged gull monitoring project began. The number of nesting adults at South Marble Island remained relatively high from 2012-2014, with an average of 823 nesting adults observed each year. The high count for observed adults each year ranged from 670 individuals (2012) to 922 individuals (2013) (Figure 51).

It is difficult to compare the long-term trends in adults for this island, as survey and timing differ between many of the studies, especially the early observations at the island which did not report survey methodologies. Comparisons are likely more accurate when made only between Zador and Piatt (1999), Arimitsu et al. (2007) and GLBA annual monitoring results, as these surveys used similar protocols. South Marble Island had the highest number of nesting adults observed each year of GLBA monitoring (Figure 51), and the average high count of nesting adults on South Marble Island between 1999 and 2014 was 868 individuals. Only one year (2012) had a large departure from this average (670 adults observed); however, that year had frequent cold temperatures and unusually high rainfall, and similar trends were observed at almost all priority islands in the park (Figure 51). Figure 55 represents the high counts of glaucous winged gull adults observed from 1941-2014, and years with comparable methodologies have been denoted with an asterisk.

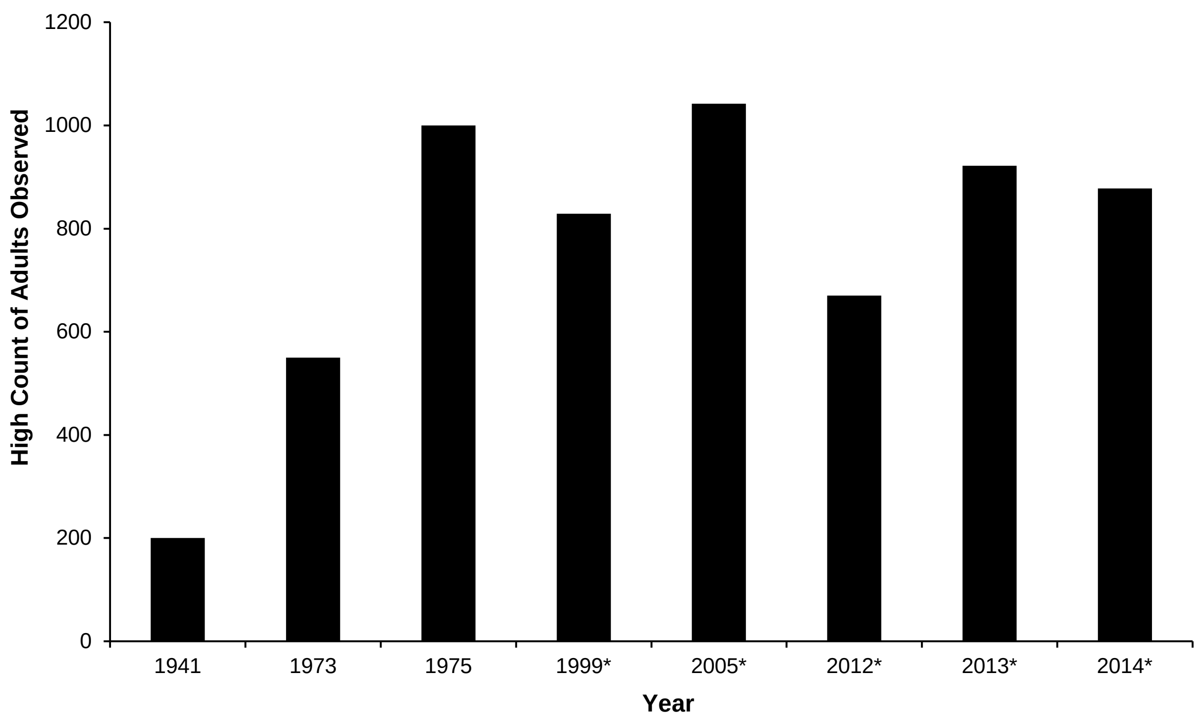

Figure 55. High count of glaucous-winged gull adults observed intermittently at South Marble Island from 1941-2014. Years denoted with an * indicate years where survey methodologies were similar and results are likely more comparable.

The spatial extent of ground based surveys (egg and nest counts), differs between these studies due to increasing numbers of Steller sea lions using the island and inhibiting access by researchers. Zador and Piatt (1999) documented 285 nests (many containing eggs) on South Marble Island during surveys on 18 and 19 June 1999. In 2005, Arimitsu et al. (2007) documented at least 200 nests containing eggs during a survey of the island (Figure 52). GLBA monitoring of the central portion of South Marble Island only began in 2012, when a high count of 128 nests was observed. The high count of observed nests declined in each year of the survey, with 107 nests observed in 2013, and 91 nests observed in 2014 (Figure 52).

Zador (2001) was the first study to document the number of glaucous-winged gull eggs at South Marble Island. However, this study investigated the effects of nest manipulation on egg replacement in the nesting colony, and from 1999-2000, several nests had either the first egg laid removed (1999 n=14, 2000 n=25) or the entire clutch removed (1999 n=17, 2000 n=24). For the purpose of this measure, only the numbers of eggs laid in the unmanipulated nests from Zador (2001) are summarized. In 1999, there were 151 unmanipulated nests, and the average number of eggs laid per nest was 3.05 ± 0.09. In 2000, the number of unmanipulated nests was 140, and the average number of eggs laid per nest was 3.74 ± 0.12 (Zador 2001). Between the two breeding seasons, 746 eggs were laid in unmanipulated nests, and the average number of eggs laid from 1999-2000 was 373 eggs.

Arimitsu et al. (2007) did not document the total number of eggs observed during a 2005 survey, but the authors did report an average clutch size of 2.4 eggs/nest. GLBA monitoring at South Marble Island began in 2012, when a high count of only four eggs was observed. As has been mentioned previously, this year was exceptionally wet and cold, which resulted in very low numbers of eggs being observed park-wide. Egg numbers rebounded on the island in 2013, when a high count of 183 eggs was observed. Egg numbers remained relatively high in 2014, as a high count of 157 eggs was reported (Figure 53). The number of eggs per nest at South Marble Island has been below average from 2012-2014, with the island having the lowest estimate of all priority sites in GLBA in both 2012 and 2014 (Figure 53). 2012 had an eggs per nest estimate of 0.031; this number increased to 1.710 in 2013 and 1.725 in 2014 (Figure 53).

Lone Island

The first survey to document the number of adults on Lone Island was Arimitsu et al. (2007), which documented a high count of 129 adults during a 2005 survey (Figure 50). The number of adults was not recorded at this island again until 2012, when the GLBA monitoring program observed a high count of 80 adults. The number of adults observed at Lone Island increased during each year of the GLBA monitoring, increasing to 192 adults in 2013, and 351 adults in 2014 (Figure 51).

One hundred fifteen nests were observed at Lone Island in 2005, with most nests being located in the grassy meadow at the center of the island (Arimitsu et al. 2007; Figure 52). In 2012, 134 nests were observed during GLBA monitoring of the island, this number increased slightly to 139 nests in 2013 (Figure 52). Hauled out harbor seals were present on the island for much of 2014, which prevented any nest or egg counts on the island that year.

Arimitsu et al. (2007) reported an average clutch size of 2.4 eggs per nest (n=115) at Lone Island in 2005. GLBA surveys of the island in 2012 resulted in a high count of 15 eggs (low production year as previously discussed). Egg numbers rebounded in 2013, and GLBA researchers documented a high count of 340 eggs (Figure 53). Because Lone Island did not have a ground survey in 2014, estimates of eggs per nest are only available for 2012 and 2013. In 2012, Lone Island had 0.112 eggs per nest. This number increased to 2.446 eggs per nest in 2013, which was the highest estimate obtained at any of the monitored sites that year (Figure 54).

Geikie Rock

One hundred twenty-three adult glaucous-winged gulls were counted on Geikie Rock in 2005 (Arimitsu et al. 2007; Figure 50). Seven years later, the number of adults observed on the island had more than doubled, as GLBA monitoring observed a high count of 264 adults in 2012 (Figure 51). Monitoring in 2013 resulted in a high count of 319 gulls on the island, and in 2014 the number declined to 160 adult gulls observed on the island (Figure 51). It should be noted that Geikie Rock appears to be a location of seasonal aggregations by non-breeding adult and juvenile gulls, and high counts may not reflect the number of breeding gulls using the island (Lewis, written communication, 24 April 2015).

The first record of the number of nests on Geikie Rock is from 2005, when Arimitsu et al. (2007) observed 48 nests (Figure 52). GLBA monitoring recorded 88 nests in 2012, 143 nests in 2013, and 131 nests in 2014 (Figure 52). In regards to egg production, Geikie Rock had an average of 2.52 eggs/nest in 2005. As was the case for all colonies in the park, egg production at Geikie Rock was low in 2012, with only 16 eggs observed. Monitoring in 2013 reported an increase in egg production to 330 eggs, and in 2014, the highest count of eggs was reported for the island (349 eggs; Figure 53). The number of eggs per nest at peak incubation at Geikie Rock in 2012 was below average, with only 0.182 eggs/nest. Both 2013 and 2014 resulted in above average years for the island, as the values reported were 2.308 and 2.664, respectively (Figure 54).

Flapjack Island

Flapjack Island had a high count of 200 individuals during a land survey in 2003 (Figure 50), although Arimitsu et al. (2007) noted that most of the glaucous-winged gulls observed were subadults. Annual GLBA monitoring has been unable to document the number of nesting adults on Flapjack Island. Surveys that count adults in the park are conducted from vessels, and the reefs that surround Flapjack Island make vessel surveys impossible (Lewis, email communication, 10 November 2014).

Arimitsu et al. (2007) documented 26 nests on Flapjack Island in 2003 (Figure 52). In 2012, GLBA monitoring recorded 114 nests on the island. In 2013 and 2014, Flapjack Island had the highest number of nests in GLBA (among the priority nesting islands). In 2013, researchers reported a high count of 174 nests, and in 2014 the island had a high count of nests of 144 nests (Figure 52).

The average number of eggs per clutch on Flapjack Island in 2003 was 1.68 (Arimitsu et al. 2007), which is a relatively low estimate as this species typically lays clutches of three eggs. Similar to the number of nests observed during GLBA monitoring, Flapjack Island had the highest number of eggs observed out of all the priority islands in the park in 2013 and 2014 (Figure 53). Fifty-seven eggs were observed in 2012 (second in number only to Boulder Island), while 2013 and 2014 had high counts of 353 and 398 eggs, respectively (Figure 53). Flapjack Island had 0.315 eggs per nest in 2012, and was one of only two sites in the park that had values that exceeded the park wide average that year (Figure 54). Estimates dipped slightly below average in 2013 (2.029 eggs/nest), but rebounded to an above average estimate again in 2014 (2.764 eggs/nest) (Figure 54).

Boulder Island

Boulder Island had a high count of 600 adults during a ground survey in 2003 (Arimitsu et al. 2007; Figure 50). This number represents the highest count of breeding adults on this island. GLBA monitoring in 2012 reported a high count of 148 adults, while high counts in 2013 and 2014 had high counts of 167 and 280 adults, respectively (Figure 51). It should be noted that GLBA monitoring of Boulder Island utilized vessel surveys, whereas Arimitsu et al. (2007) performed a ground survey. Also, similar to Geikie Rock, Boulder Island appears to be a location of seasonal aggregations by non-breeding adult and juvenile gulls, and high counts may not reflect the number of breeding gulls using the island (Lewis, email communication, 10 November 2014).

Forty-one glaucous-winged gull nests were observed on Boulder Island in 2003 (Figure 52). Somewhat surprisingly, during GLBA monitoring Boulder Island had its highest count of nests during 2012, a year in which most sites in the park reported very low nest counts. Ninety-two nests were observed in 2012, with nest numbers declining in 2013 and 2014 to 70 and 78 nests, respectively (Figure 52).

Egg production on Boulder Island has been variable. Arimitsu et al. (2007) reported an average clutch size of 2.58 eggs per nest. GLBA monitoring observed 83 eggs in 2012, followed by 165 eggs in 2013 and 183 eggs in 2014 (Figure 53). Boulder Island had eggs per nest values that exceeded park-wide averages in both 2012 (0.902 eggs/nest) and 2013 (2.357 eggs/nest). The estimate dropped slightly below average in 2014 to 2.346 eggs per nest (Figure 54).

Number of Harvested Eggs by Location

Glaucous-winged gull eggs were traditionally harvested in Glacier Bay by the Huna Tlingit tribe. However, the establishment of Glacier Bay National Monument in 1925, combined with regulations under the Migratory Bird Treaty Act of 1918 hindered the tribe’s ability to legally harvest the eggs, and by the early 1970s egg harvests had ceased in the park. Beginning in the late 1990s, the NPS began investigating allowing gull egg harvest in the park. Biological studies of the gull population took place in 1999 and 2000 (Zador and Piatt 1999, Zador 2001), and mathematical models were developed to analyze the potential effects gull egg harvests would have on the population (Zador and Piatt 2002, Zador et al. 2006). In addition to biological studies, the NPS completed an ethnographic study of the Huna Tlingit tribe (Hunn et al. 2002), with particular emphasis paid to historic egg collection practices in the Glacier Bay region.

In 2010, the NPS determined that the harvest of glaucous-winged gulls could be authorized in the park without creating a detrimental impact to the gull population (NPS 2010b). This decision has not been implemented in the park, and no data related to this measure exist at this time.

Several harvest constraints were outlined for future harvests in the park by NPS (2010b). Harvests will occur up to twice a year, with the first harvest beginning within 5 days of the onset of laying at one of 15 potential locations: South Marble Island, Lone Island, Geikie Rock, Boulder Island, Muir Inlet shoreline between Riggs and Muir Glaciers, Flapjack Island, Sebree Island, Tlingit Point Islet, Sturgess Island, Sealers Island, Graves Island (Outer Coast), Hugh Miller Islet, Margerie Glacier, Mt. Wright, Muir Inlet Cliffs (NPS 2010b). If a second harvest occurs, it must occur within 9 days of the first harvest and occur no later than 15 June.

There will be no limit on the total number of eggs harvested from a particular location. However, for nests that are harvested, collectors will be required to remove all eggs from the clutch in order to stimulate the laying of a replacement clutch. Nests with hatched chicks or eggs that are pipping or have star-shaped fractures are not to be harvested.

Harvesters will be required to document harvest statistics for each island that is visited during a season. An annual report will be completed by the Huna Indian Association (HIA) and delivered to GLBA at the conclusion of each year's harvest. Items to be included in this report will include:

- Date of site visits, harvest locations, and number of harvesters/site;
- Number of eggs taken from nests with one, two, three, and four eggs as well as number of nests with no eggs located at each site per visit;
- Number of pipped, star-fractured, or predated eggs and number of hatched chicks in nests located at each site per visit;
- Number of marine mammals hauled out at harvest location; number of animals leaving the haul out and entering the water before, during or immediately after harvest activities; behavioral changes including increased alertness or increased aggressive interactions at each site per visit;
- Other wildlife species present at each site per visit;
- Visitor interactions at each site per visit (NPS 2010b, p. 5).

<u>Threats and Stressor Factors</u>
While a naturally occurring process, plant succession in GLBA represents a threat to many glaucous-winged gull nesting colonies in the park. According to NPS (2010a, p. 10),

> *Agents and processes of disturbance (e.g., glaciers, earthquakes, floods, erosion, insect irruptions), along with those of ecosystem recovery (e.g., biological succession, landform evolution) are allowed to exist and proceed free of human influence. Fluctuations in animal and plant populations (driven by any natural cause) are allowed to occur and reach their own states of equilibrium.*

The ongoing vegetation succession and isostatic rebound in Glacier Bay continually creates and removes habitat that is suitable for nesting gulls. Gulls in the park have exhibited a preference for open meadows with low grassy cover near the shoreline. However, this habitat type is characteristic of an early successional stage in the park, and ecological processes gradually transition these areas to more densely forested areas with little open canopy. Relatively recently, North Marble Island had a sizeable nesting gull colony. However, successional processes transitioned the island's vegetation to a conifer forest with little open canopy or grassy areas. If succession continues at the same rate as has been observed at South Marble Island, it is likely that the forest on the island will expand into nesting gull colonies (Zador et al. 2006).Transitions like this are to be expected at many of the islands in the park and will eventually preclude gull nesting at historic locations (NPS 2010b).

Glaucous-winged gull eggs were collected by the Huna Tlingit in Glacier Bay during the 19th and early 20th centuries, although no harvests have occurred in the region since the early 1970s (Hunn et al. 2002, NPS 2010b). In 2010, the NPS determined that the harvest of glaucous-winged gulls could be allowed in the park without creating a detrimental impact to the gull population (NPS 2010b). Legislation passed in July 2014 that would legalize harvest in GLBA, but no regulations have yet been established and harvest has not occurred to date.

The effects that an egg harvest could have on the GLBA glaucous-winged gull population are relatively unknown. Zador et al. (2006) used a mathematical simulation to determine the effects of a controlled egg harvest at a glaucous-winged gull colony using data collected from South Marble Island in 1999 and 2000. The results of the simulation showed that hatching success following a controlled harvest of 20% nests in a colony would be slightly reduced (6%). However, a threat that was not simulated was the amount and frequency of disturbance created at nesting sites during harvest efforts; increases in disturbances at nesting sites could lead to an overall reduction in reproductive success in colonial nesters (Brown and Morris 1995, Sullivan et al. 2002, Hothem and Hatch 2004). Glaucous-winged gull egg harvests are going to occur in GLBA as soon as regulations are promulgated. Continued monitoring of harvest efforts and population trends at harvested islands will be needed to determine potential shifts in population abundance or nesting success.

Egg harvests are not the only threat to nesting gull colonies, however, as natural predation is common during the breeding period. Zador and Piatt (2002, p. 13) simulated the effects of predation and harvest on gull colonies in GLBA and noted that gulls "…respond more negatively to chronic exposure to predation than to episodic predation (i.e., harvest)." In GLBA, bald eagles (*Haliaetus leucocephalus*) appear to be the primary predator of glaucous-winged gull eggs and nestlings (Zador and Piatt 1999; Lewis, phone communication, 6 November 2014). Zador and Piatt (1999) observed eagles taking eggs from gull nests, and also found the remains of adult gulls near known perch trees. Depredated eggs are easily attributed to eagles, as eagles leave a characteristic large "V" puncture wound in the egg (Lewis, phone conversation, 6 November 2014). Other potential predators of glaucous-winged gulls in the park include the northwestern crow, the common raven (*Corvus corax*), river otters, and black and brown bears.

Data Needs/Gaps

Glaucous-winged gull monitoring in the park was sporadic prior to 2012, with most research dedicated to the nesting colony found on South Marble Island. With the potential of an egg harvest in the park in the near future, continued monitoring is needed in the park. Annual surveys early in the year to determine nesting initiation and incubation should continue, as should surveys later in the year to monitor the number of chicks and adults at colonies. The gull harvest LEIS ROD (NPS 2010b) also specifically mandates the NPS to carry out monitoring of gull colonies to inform and mitigate potential negative effects of the harvest. Yearly monitoring requirements include: 1) identify the onset of nesting; 2) determine breeding colony size; 3) determine number of eggs in nests during harvest; 4) determine number of eggs available for harvest or hatching; and 5) document other bird and marine mammal species present that may be impacted by harvest activities. Harvest sites are to be selected based on: size of colony; population parameters including productivity, population status, recent egg harvest, age of colony; and other species present. In addition, multi-year studies (e.g., 3 years) are highly recommended, subject to available funds, to identify potential causes of change in park gull population levels, and could include an assessment of egg laying phenology, predation pressure, and reproductive success.

The LEIS ROD does not specifically mandate monitoring gull nesting habitat, however it does indicate that the park should be looking for new nesting colonies as they emerge. Glacier Bay is a recently deglaciated fjord still undergoing vegetative succession and isostatic rebound. Gull nesting colony locations will undoubtedly change as plant communities mature into shrubs and eliminate nesting substrate while new islands emerge and become good nesting habitat. This decrease in abundance of breeding gulls corresponds with vegetative successional changes on the island transitioning from large open meadow areas in the 1970’s to the largely closed conifer forest in the early 2000’s. Due to the inevitable decrease in gull nesting habitat due to plant succession in the future, Zador (2006) recommends monitoring vegetation succession at gull colonies in Glacier Bay. Additionally, predictive models of usable gull nesting habitat would be helpful in determining the length of time nesting would be expected at each colony given its current successional state and trajectory.

Annual harvest summaries will be needed when harvest begins in the park. As outlined by NPS (2010b), these summary documents should include core statistics such as: harvest locations and date of harvest; number of eggs taken from nests with one, two, three, and four eggs; number of pipped, star-fractured, or predated eggs; and number of hatched chicks in nests located at each site per visit. Post-harvest nest surveys will be needed to determine the number of clutches that are successfully relaid. Additionally, egg size has been found to correlate positively with hatching weight, growth rates, and chick survival in many species of gulls and terns (Lundberg and Viasanen 1979, Bolton 1991). Comparing measurements of eggs from harvested and post harvested (presumably second clutch) nests would be helpful in determining effects of harvest on chick survival.

Overall Condition

Distribution

The distribution measure was assigned a *Significance Level* of 2 during project scoping. Glaucous-winged gulls in the park have occupied the same nesting islands for several decades in GLBA. Some historic nesting sites have been vacated (e.g., North Marble Island), but there does not appear to be any cause for concern in regards to the current distribution in the park. Sites with the highest quality nesting habitat are currently active, and islands that are located at least 1.6 km (1 mi) appear to be favored for nesting. Shifts in distribution in the park are largely driven by succession and habitat availability, and it is likely that nesting sites will continue to shift as new habitat becomes available and when historic nesting sites no longer support the necessary vegetation. The distribution measure was assigned a *Condition Level* of 1, indicating low concern.

Population Trends in Select Areas

The GLBA project team assigned the population trends in select areas measure a *Significance Level* of 3. This measure investigated trends in six areas of the park: Tlingit Point, South Marble Island, Lone Island, Geikie Rock, Flapjack Island, and Boulder Island. Trends at these islands were analyzed using data from 2003 and 2005 (Arimitsu et al. 2007) and from 2012-2014 (GLBA annual monitoring). Parameters used to determine trends included the number of nesting adults, the number of nests, the number of eggs, and the number of eggs per nest at peak incubation.

Determining trends at these select areas is problematic, as several islands do not have a reference condition for some/all of the parameters. Due to differences in methodology and timing, only data regarding the number of adults and number of eggs per nest were comparable between Arimitsu et al. (2007) and GLBA monitoring. For several of these priority areas, the values reported in this document will serve as a baseline or reference condition for future condition assessments. For South Marble Island, Zador and Piatt (1999) and Zador (2001) served as reference conditions.

Boulder Island and South Marble Island were the only islands that were consistently below the Arimitsu et al. (2007) reference conditions for number of nesting adults (note: Lone Island did not have adult surveys). Numbers increased from 2012-2014 at Boulder Island, but never approached the high count of 600 adults observed by Arimitsu et al. (2007) (Figure 50, Figure 51). South Marble Island had the highest number of adults in the park, and numbers increased from 2012-2013; in 2014 the colony had only a minor reduction in abundance (44 adults). While South Marble was below the Arimitsu et al. (2007) reference condition of 1,042 adults, nesting adult abundance from 2012-2014 was above the Zador (2001) reference condition of 842 adults. Tlingit Point had the lowest number of adults in the park (Figure 51), with numbers remaining consistently low (values ranged from 40-68 during GLBA surveys). There does not appear to be any major concern in GLBA at this time in regards to the number of nesting adults at the priority nesting areas. However, interpreting trends in the number of adults at a nesting colony is difficult due to the inherent natural variations as well as the number of data points in the time series. Factors such as the timing of a survey and influxes of non-breeding adults and subadults at a site can greatly influence abundance estimates in a given year. Sites such as Boulder Island and Geikie Rock are frequently visited by these non-breeding gulls, which increase the number of adults observed at those colonies, but may contribute to an inflated number of perceived 'nesting' adults.

Two nesting colonies in the park (South Marble Island and Tlingit Point) have had annual declines since 2012 in the number of nests observed (Figure 52). The number of nests at South Marble declined from 128 nests in 2012, to 91 nests in 2014. Tlingit Point had the lowest number of nests in the park each survey year, and the number of nests declined form 51 nests in 2012 to 22 nests in 2014 (Figure 52). While determining trends from only 3 years of data is not possible, park managers should keep a close watch on these colonies to determine if these recent declines are due to natural variations, or are indicative of a more serious issue.

There does not appear to be cause for concern regarding the number of eggs observed at these priority locations. 2012 represented an exceptionally cold and wet breeding season, and egg numbers were at historic lows at every site visited during the GLBA surveys. Numbers rebounded in 2013, and in most locations numbers continued to increase in 2014 (Figure 53). Despite having large numbers of adults, South Marble Island has produced only moderate numbers of eggs compared to other nesting sites that are surveyed (Figure 53), and this site was the only site to experience a decrease in the number of eggs in 2014.

The park-wide average for number of eggs per nest increased yearly (Figure 54). Historic lows were observed in 2012 (0.278 eggs/nest), and values increased to 2.440 eggs/nest in 2014; for reference, a typical clutch size for glaucous-winged gulls is three eggs per nest. Only one location in the park (Boulder Island) did not have peak eggs per nest numbers in 2014; however, Boulder Island's eggs per nest value was only slightly lower in 2014 compared to 2013 (2.357 eggs/nest in 2013 versus 2.346 eggs/nest in 2014). South Marble Island was below the park-wide average each year during GLBA surveys (Figure 54).

While it is not possible to ascertain trends from only 3 years of data, there does not currently appear to be major causes of concern regarding the glaucous-winged gull populations at the priority sites in the park. Comparisons to historic reference conditions did not result in substantial departures from what could be expected in number of nesting adults at each location. This measure was assigned a *Condition Level* of 1, indicating it is currently of low concern. However, particular attention should be given to this and other measures if a controlled harvest begins in the park. Values obtained from surveys before the harvest could likely serve as a baseline for comparison, and potential trends in population parameters could be investigated.

Number of Harvested Eggs by Location

The number of harvested eggs by location measure was given a *Significance Level* of 3 during project scoping. No glaucous-winged gull eggs have been legally harvested in the park since the early 1970s. Accordingly, there are no data for this measure at this time. However, should a controlled harvest begin in the park, the number of harvested eggs at each location will serve as a valuable piece of information to park managers. A *Condition Level* was not assigned to this measure at this time.

Weighted Condition Score (WCS)

The *Weighted Condition Score* for glaucous-winged gulls was determined to be 0.33, indicating that this component is currently of low concern. A trend arrow was not assigned to this measure, as recent data are only 3 years in duration, and there exists some degree of complexity in assigning a trend to several unique and distinct nesting colonies. This component represents a unique situation, in that eggs will likely be harvested in the near future. While the current condition of this component is of low concern, biologists and park managers will likely pay more attention to this species as harvests begin and trends emerge.

Glaucous-winged Gulls			
Measures	Significance Level	Condition Level	WCS = 0.33
Distribution	2	1	
Population Trends in Select Areas	3	1	
Number of Eggs Harvested by Location	3	n/a	

4.7.6. Sources of Expertise

- Tania Lewis, GLBA Wildlife Biologist.

4.7.7. Literature Cited

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4.8. Moose

4.8.1. Description

Moose are a relatively recent arrival to GLBA and did not colonize the area around GBP until the 1960s, after deglaciation (White et al. 2006, Scott 2010a). Northern populations in the Alsek River and Dry Bay areas were first seen in the 1920s and 1930s (Scott 2010b). Moose are generally found in the thickets, meadows and open forests in the lowlands of GLBA (Photo 20). They occupy boreal forests as well as areas of shrub/scrub near the rivers and lake riparian zones (Rausch et al. 2008). Moose are normally associated with northern forests and subarctic climates typical of southern Alaska. The species is typically a solitary mammal and rarely gathers in groups except during the mating season. Females weigh in excess of 363 kg (800 lbs), while males can weigh up to 725 kg (1,600 lbs) and exceed 1.83 m (6 ft) in height at the shoulder (Rausch et al. 2008). Antlers develop on males within the first year of life and grow larger each subsequent summer. Antlers typically develop in 3 to 5 months beginning in the spring, and are shed after the fall mating season. The average moose life span is 16 years, although 25-year-old individuals have been reported (Rausch et al. 2008).

Photo 20. A moose in GLBA (NPS photo).

Moose are herbivorous, feeding primarily on willow, aspen (*Populus* spp.), aquatic vegetation, and a variety of grasses (Rausch et al. 2008). Moose are most often associated with open low or mixed shrub vegetation classifications near riparian zones. Sexual maturity and breeding occur at about 28 months, with calves being born in the spring. Females typically produce one calf per year, or occasionally twins (Rausch et al. 2008). During the mating season, sparring occurs between bulls in order to secure mates. While injuries are common, they are rarely serious (Rausch et al. 2008). Adult moose are generally calm and subdued, although aggressive behavior is displayed by cows with calves when startled, angered, or when offspring are threatened, as well as by bulls during the breeding period (Rausch et al. 2008).

Natural predators of moose include wolves, black bears, and brown bears (Rausch et al. 2008, White et al. 2014). Predation can be a significant source of mortality among moose calves (Scott 2010a, b). Moose populations are protected from sport and subsistence hunting within the national park. Sport hunting occurs on lands south of the park, along the Gustavus forelands. However, subsistence and sport hunting are allowed within the national preserve (Figure 56, NPS 2015a).

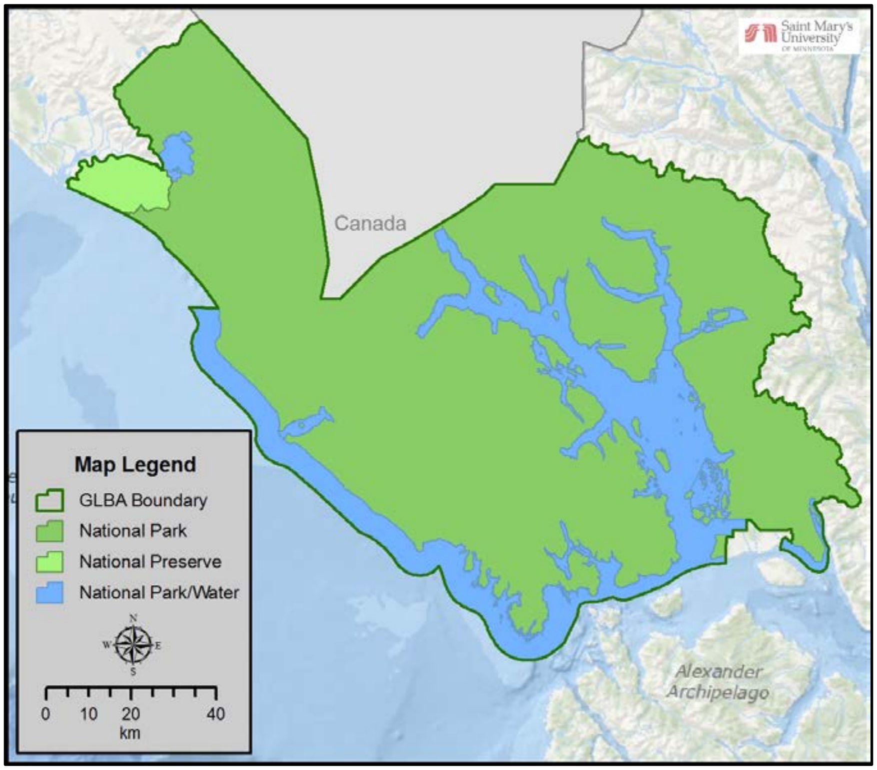

Figure 56. Location of the National Preserve portion of GLBA where hunting is allowed.

4.8.2. Measures

- Distribution (presence in a location)
- Abundance (number of moose per square mile)
- Population trends in select survey areas (Gustavus, Dry Bay [harvest], Adams Inlet, Taylor Bay)
- Age/sex ratios

4.8.3. Reference Conditions/Values

Reference conditions for the GLBA moose population are largely unknown due to limited information. The information presented in this assessment, particularly the 2012 NPS survey, could be used as a baseline for future assessments. With regard to sex ratios, no reference condition exists.

4.8.4. Data and Methods

The ADFG conducts aerial surveys for moose in selected areas, typically where moose hunting is common. Aerial survey results are presented by GMU. GLBA covers portions of three GMUs: 01C, 01D, and 05A (Figure 57). Within GMU 01C, survey reports focused on four distinct areas. Only one of these fell within or near GLBA boundaries: the Gustavus forelands (Scott 2010a). Survey areas within GMU 01D fell outside park boundaries. GMU 05A study areas included the Yakutat forelands, which extend into the national preserve area in the north. Information gathered includes population estimates, bull:cow ratios, and harvest data.

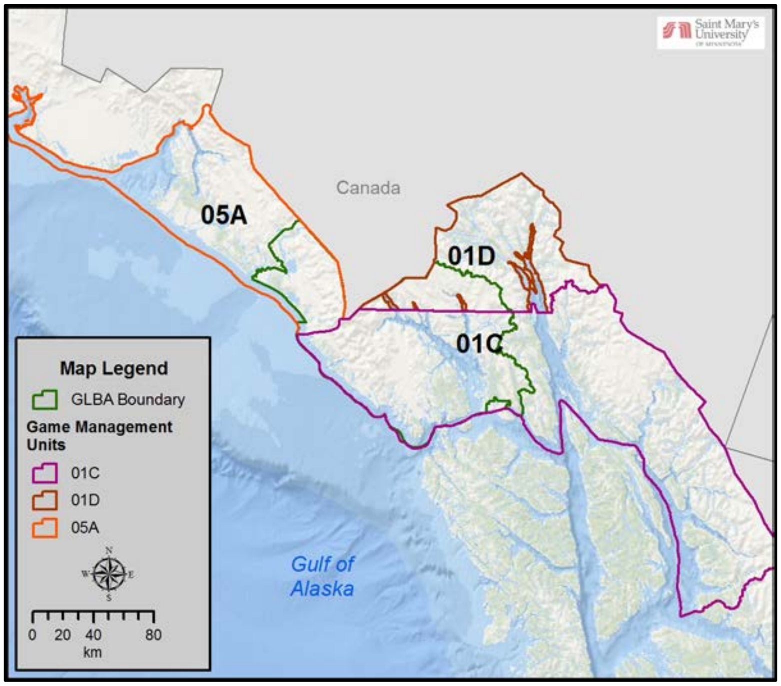

Figure 57. GMUs included within GLBA's boundaries.

White et al. (2006) studied the ecology of the Gustavus forelands moose population, with a focus on population responses to food/nutrient limitations. Data reported include moose range and density, forage selection, browse utilization, body condition, and reproductive success. White et al. (2014) studied moose in the same area to explore why some of the population remains in the forelands year round while much of the population migrates north in the summer. The research involved tracking radio-collared female moose from 2004-2010 to document migratory behavior. Body condition and calf survival data were collected to determine if migratory behavior provided nutritional or survival benefits. Aerial surveys were conducted annually each winter (2003-2010) to estimate population density (White et al. 2014).

The NPS conducted aerial moose surveys in GLBA during the winter of 2012/2013 (Lewis and White 2016). This survey focused on the south and central portions of the park, primarily around GBP, and did not extend to the outer coast or northern area of GLBA. The 2012 survey consisted of 15 flight hours over 3 days (two flights in December and one in March) (Lewis and White 2016). The purpose of these surveys was to gather preliminary data on the distribution and abundance of the moose population (Lewis and White 2016). Lewis and White (2016) included a comparison between historical data from Adams Inlet and the 2012 NPS survey.

4.8.5. Current Condition and Trend

<u>Distribution</u>

Moose occur at low elevations throughout the park. Areas around Glacier Bay were surveyed most recently in 2012/2013; moose sightings during these surveys are presented in Figure 58. It is likely that moose occur in other locations, especially along the coast, but these areas have not been surveyed in recent years.

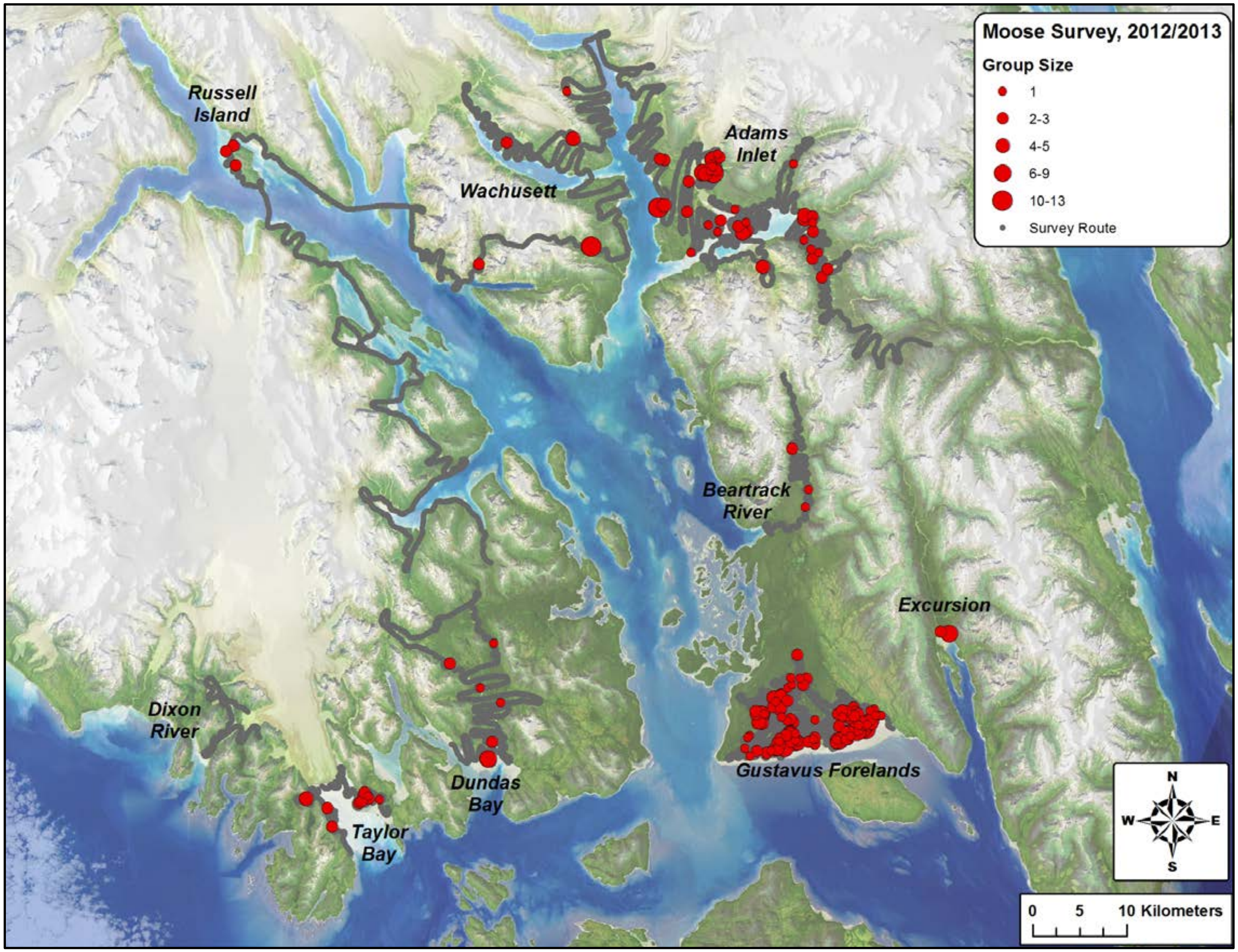

Figure 58. Moose sightings in GLBA during the 2012 aerial surveys (Lewis and White 2016). Note that the northern outer coast and Dry Bay (preserve) areas were not covered during these surveys (Lewis and White 2016).

White et al. (2006) mapped the winter and summer ranges of the Gustavus moose population based on telemetry tracking data from radio-collared moose in 2003-2004. Eight moose were tracked during the winter and 20 during the summer (White et al. 2006). Ranges are shown in Figure 59. Note that most of the winter range is outside park boundaries in an area where hunting is allowed.

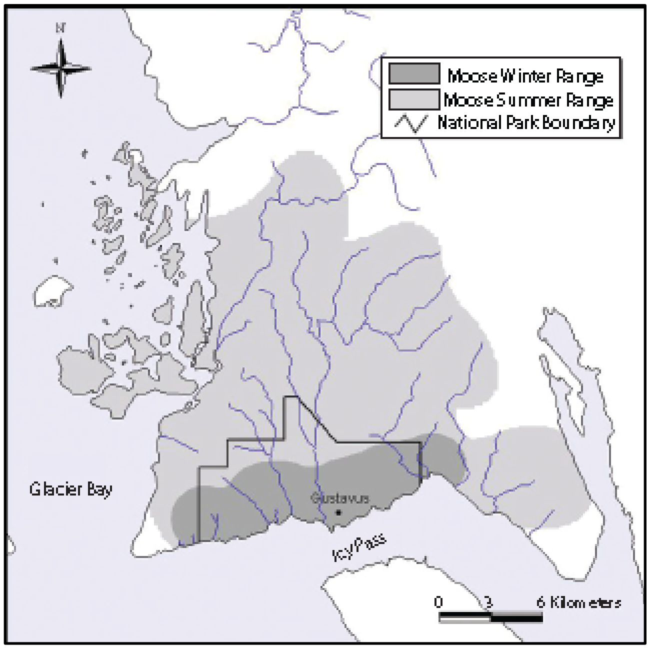

Figure 59. Winter and summer ranges of the Gustavus moose population (White et al. 2006).

White et al. (2014) further investigated the range and migration behavior of the Gustavus moose population. Radio-tracking of female moose showed that approximately 30% of the population remains in the Gustavus forelands area year-round and 48% migrate during the summer calving season, either north towards the Beardslee Islands or northeast to Excursion Ridge (White et al. 2014, Photo 21, Figure 60). The study found that calf survival rates were 2.6-2.9 time higher for migratory females than for year-round resident females. White et al. (2014) theorized that migration reduces the risk of calf loss to predation.

Photo 21. A radio-collared female moose in GLBA (NPS photo).

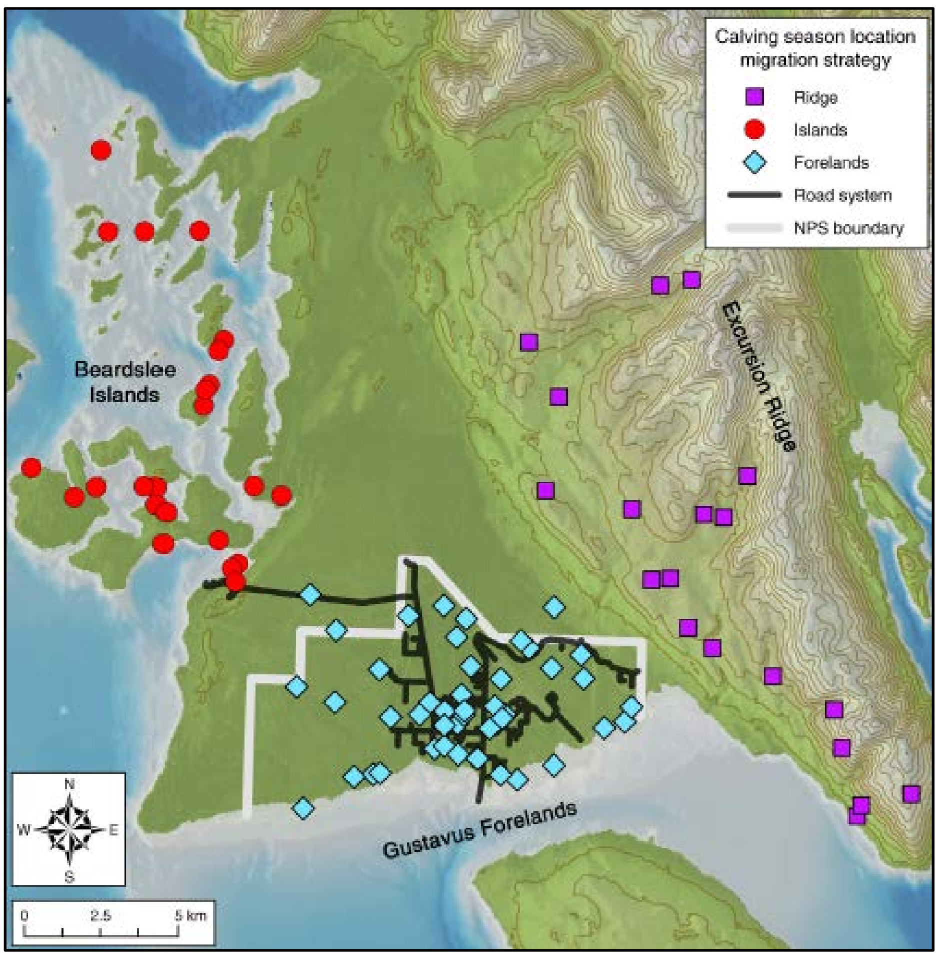

Figure 60. Locations where radio-collared female moose were first observed with calves during the 2004 calving season. Different symbols represent the three different migratory strategies of female moose (White et al. 2014).

Although not documented in recent surveys, moose are also known to occur in the Dry Bay area at the northern end of GLBA. During an aerial survey in the 1980s, Dinneford (1984) observed moose on the flats below Novatak Glacier and between the East and Alsek Rivers.

Abundance (Number of Moose per Square Kilometer)

Estimates of the entire GLBA moose population do not exist. The 2012 aerial survey, which covered a large portion of non-glaciated lower elevation lands in the park and the Gustavus forelands (see Figure 58) identified 466 individual moose (Lewis and White 2016). Since the ADFG typically estimates moose populations by GMU, full GMU population estimates would not reflect numbers within the park. However, within GMU 01C, the ADFG has surveyed the Gustavus forelands area and estimated that population size separately (Barten 2008, Scott 2010a). During 2005 and 2006 aerial surveys of the Gustavus area, 295 and 329 moose were observed, respectively (Barten 2008). Given that sightability is thought to be 70% (e.g., surveyors are able to see about 70% of moose present), the moose population was estimated to be above 400 animals (Barten 2008). This was down slightly from an estimate of 425 moose in 2004 (Barten 2006). Surveys of the same area counted 254 moose in 2007 and 273 moose in 2008 (Scott 2010a). Incorporating sightability, the moose population was estimated at 274-342 animals (Scott 2010a).

White et al. (2006) state that the Gustavus moose population expanded rapidly during the 1990s and early 2000s, reaching a winter density of 3.9 moose/km^2 by 2006. Anecdotal data suggest that until the early 1990s, this population was below 100 individuals (White et al. 2006). In a later report, White et al. (2014) estimated that the Gustavus winter range population density varied between 2.4 and 4.8 moose/ km^2 from 2004 to 2010. Compared to other coastal Alaska moose populations, this density is very high (White et al. 2006).

The Yakutat forelands portion of GMU 05A extends into (but is not limited to) the northern portion of GLBA around Dry Bay (Figure 57). As a result, the abundance of moose in the GLBA portion of GMU 05A alone cannot be isolated from survey data, but population trends in the area likely reflect conditions in the northern park and preserve. In the 2007 aerial survey of the Yakutat forelands, the ADFG observed the highest number of moose since the late 1960s (Scott 2010b). The number of moose observed often depended on the number of hours surveyed, suggesting that the best way to compare moose abundance data over time is to compare “moose per hour” numbers. Between 2000 and 2008, moose per hour during Yakutat forelands surveys ranged from 36 moose/hr to a peak of 76 moose/hr in 2007, with a mean of 51.3 moose/hr (Scott 2010b).

Population Trends in Select Survey Areas (Gustavus, Dry Bay, Adams Inlet, Taylor Bay)

The project team selected four areas in or near GLBA where moose population trends are of particular interest. These are: Gustavus, Adams Inlet, Taylor Bay, and Dry Bay. Three of these areas (excluding Dry Bay) were surveyed in 2012 (Figure 61). Historical survey data are also available for Adams Inlet (Lewis and White 2016).

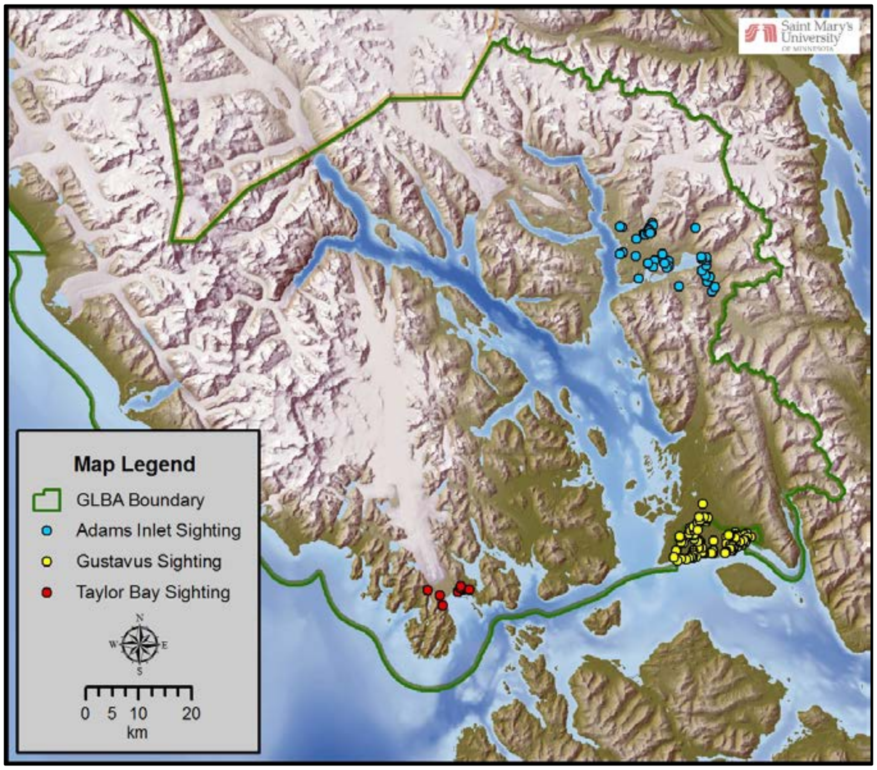

Figure 61. Moose sightings in three areas of interest during 2012 aerial surveys (NPS 2012).

Gustavus had the highest observed moose total with 272 individuals, over half of which were lone cows (Table 27; Lewis and White 2016). Adams Inlet had the second highest total with 112 moose; 77 could not be identified as bulls or cows. The lowest numbers were in Taylor Bay, with just 26 moose (Lewis and White 2016). No bulls were positively identified in Taylor Bay, although some of the "unknown" individuals could have been bulls that had already shed their antlers (Table 27). Note that Gustavus was surveyed on 8 December 2012, Adams Inlet on 18 December 2012, and Taylor Bay on 16 March 2013. Bulls drop their antlers in November/December, so after mid-December all single moose without antlers were considered unknown sex. For this reason, bull:cow ratios were only confirmed from Gustavus.

Table 27. 2012 aerial survey results for three areas of interest (Lewis and White 2016).

Location	Bulls	Lone Cows	Cows w/ 1 calf	Cows w/ 2 calves	Unknown	Total
Gustavus	33	160	38	1	1	**272**
Adams Inlet	25	0	5	0	77	**112**
Taylor Bay	0	0	4	1	15	**26**

Figure 62. Locations of Adams Inlet sub-areas (White and Lewis, *in prep*).

Lewis and White (2016) compare the 2012 data from Adams Inlet to several historical surveys of that area (Table 28). Several surveys further divided the Adams Inlet area into four sub-areas: Adams Island, and North, East, and South Adams (Figure 62). Moose numbers varied greatly between survey years, particularly with survey effort (e.g., survey time). In the 2012 survey, the highest proportion of moose were observed in the North Adams sub-area (Table 28).

Table 28. Numbers of moose observed during aerial surveys around Adams Inlet over time, 1984-2012 (Lewis and White 2016).

Year	Adams Island	N. Adams	E. Adams	S. Adams	Moose Per Hour	Total Moose	Source
1984	8	36	2	1	44	**47**	Vequist 1986
1985	39	31	25	0	68	**95**	Vequist 1986
1986	32	25	4	0	57	**67**	Vequist 1986
1994	–	–	–	–	75	**97**	Robus 1993
2000	–	–	–	–	68	**113**	Barten 2001
2001	–	–	–	–	11	**18**	Barten 2002
2010	0	27	3	0	60	**30**	Barten 2010
2012	12	72	23	5	43	**112**	Barten 2012
Mean					**53 (±13)**	**72.4**	–

Age/Sex Ratios

Information on GLBA moose age and sex ratios is very limited. White et al. (2006) documented a ratio of 55 calves:100 cows for the Gustavus moose population in 2003. In 2006, the ADFG estimated a much lower calf:cow ratio of 14:100 (Barten 2008). In 2008 and 2009 surveys of the Gustavus forelands, the ADFG estimated calf:cow ratios of 7:100 and 6:100, respectively (Scott 2010a). Based on numbers from the 2012 NPS aerial survey in three of the four selected survey areas (Table 27), the calf:cow ratio was approximately 20:100 (NPS 2012). Given the high proportion of “unknown” moose in 2012 surveys of Adams Inlet and Taylor Bay (Table 27), it is not appropriate to calculate age or sex ratios from those data. Some insight into conditions in the northern portion of GLBA can be gained from age/sex ratios for the Yakutat forelands portion of GMU 5A. In four surveys between 2002 and 2008, calf:cow ratio estimates ranged from 11:100 to 17:100 with a mean of 13.8 calves:100 cows (Scott 2010b).

Photo 22. A moose calf in GLBA (NPS photo).

Sex ratio information is also limited to surveys of the Gustavus forelands population and the Yakutat Forelands area. In 2006, the ADFG estimated the bull:cow ratio for the Gustavus forelands at 23:100 (Barten 2008). The 2008 survey found a ratio of 20 bulls:100 cows (Scott 2010a). In 2009, the ratio declined to 13 bulls:100 cows (Scott 2010a). The results of the 2012 NPS survey suggest a ratio of 17 bulls:100 cows on the Gustavus forelands (NPS 2012). In Yakutat forelands surveys (2002-2008), bull:cow ratio estimates ranged from 9:100 to 19:100 with a mean of 15.3 bulls:100 cows (Scott 2010b).

Threats and Stressor Factors

Threats to the GLBA moose population include habitat change, accelerated plant succession, harvest (limited to a few areas), predation, and disease. Habitat change and plant succession are natural processes within GLBA, as landscapes go through primary succession following deglaciation (Chapin et al. 1993). Vegetation in early to mid- successional stages (e.g., shrubs) provides critical forage for moose populations (Stephenson et al. 2006). Later successional stages, where trees are more dominant, support fewer moose browse species, but do provide valuable cover in the winter and during calving season (Stephenson et al. 2006). If the successional stages that provide forage were lost, the moose population would be negatively impacted. Climate change has the potential to influence plant succession and habitat at GLBA. The climate in southeast Alaska is projected to become warmer and drier over the next century (SNAP et al. 2009). Increased temperatures are likely to lengthen the growing season and impact soil water availability, which is critical for vegetation (SNAP et al. 2009). Climate change could also accelerate glacial recession, making more land available for succession, and shoreline erosion (e.g., loss of land area) (NPS 2015b).

Although hunting is not allowed within the national park boundaries, subsistence and sport hunting are permitted in the national preserve area around Dry Bay (Figure 56) and just outside the park around Gustavus on some of the GLBA moose population’s winter range (White et al. 2006, NPS 2015a). In the early 2000s, ADFG allowed increased hunting around Gustavus due to concerns over high moose densities and impacts on the landscape (Scott 2010a). During the bull moose hunts of 2005 and 2006, 47 and 37 bulls were harvested, respectively (Barten 2008). An antlerless hunt was also allowed in these years, yielding a harvest of 69 cows in 2005 and 12 cows in 2006 (success was low because of difficult conditions due to deep snow) (Barten 2008). Twenty-nine bulls were harvested in 2007, followed by a harvest of 15 bulls in 2008 (Scott 2010a). Only 15 antlerless permits were issued in 2008, yielding a harvest of 10 cows (Scott 2010a). By this time, the moose population around Gustavus had been reduced enough by antlerless hunts, harsh winters, and predation to return to a bull hunt only with a harvest quota of 15 bulls per year (Scott 2010a, 2012). Within the national preserve area, annual moose harvest has ranged from one to 10 individuals (bulls only) between 1995 and 2014 (ADFG 2015a). Mean harvest during this time period (excluding 2000-2001, where data are unavailable) was 4.8 bulls/year (ADFG 2015a).

In and around GLBA, moose are preyed upon by wolves and brown and black bears (White et al. 2014). Calves are targeted more frequently by these predators than adults. Until the past decade, there was little evidence of moose predation on the Gustavus population and calf recruitment was high (White et al. 2006). In the mid-2000s, calf numbers began to decline, likely due to increased predation (Barten 2008, Scott 2010a). This increase in predation impacts recruitment and could ultimately lead to a population decrease.

To date, diseases and parasites have not had much of an impact on southeastern Alaska moose populations. Alaska moose may be infected with parasites such as tapeworms, which form cysts in the animal’s organs and/or muscles, but these animals still appear healthy (Rausch 1959, ADFG 2015b). Chronic wasting disease (CWD), a fatal neurological disease of cervids, would be a serious threat to moose but it has not yet been detected in Alaska (ADFG 2015c). There is some concern that warming due to climate change could increase the threat from diseases and parasites. Warmer summers and shorter winters would favor the spread of parasites, such as ticks, and the diseases they carry (Sinnott 2013). Moose populations in the lower 48 states have experienced an alarming decline in recent years; while the exact cause is unknown, two of the leading suspects are increased parasite loads and heat stress, both linked to shorter and warmer winters (Robbins 2013, Sinnott 2013).

Data Needs/Gaps

The majority of information available for GLBA’s moose is from the Gustavus population. No distribution, abundance, population size, or population composition data could be found specific to the Dry Bay area. Abundance and population composition (age/sex ratio) data are only available for the Gustavus population, and age/sex ratio data are limited due to difficulties in identifying sex and age during aerial surveys (Scott 2010a, b). Data specific to the four selected areas of interest are limited to the 2012 NPS survey. A continuation of aerial surveys by the NPS would gather valuable data regarding moose distribution, abundance, population composition, and population trends in selected areas of interest.

Overall Condition

Distribution

The project team assigned this measure a *Significance Level* of 2. Moose are a relatively new arrival to most of the GLBA area, first seen in the 1960s (White et al. 2006, Scott 2010a). Recent surveys have documented moose primarily in coastal areas along Glacier Bay and towards the Gulf of Alaska (Lewis and White 2016). Moose are also known to occur in the Dry Bay area. Although the distribution of moose in GLBA has not been fully studied, there is no evidence of any cause for concern regarding distribution. As a result, this measure is assigned a *Condition Level* of 1, indicating low concern.

Abundance

The abundance measure was also assigned a *Significance Level* of 2. Abundance data are only available from the ADFG for the Gustavus moose population, and from a 2012 NPS survey of a portion of the park. Since information regarding moose abundance is limited for most of the park and preserve, a *Condition Level* was not assigned for this measure.

Population Trends in Select Survey Areas

This measure was assigned a *Significance Level* of 3. With the exception of the Gustavus population, little is known about the moose population trends in the selected survey areas (Adams Inlet, Taylor Bay, and Dry Bay). Multiple surveys of Adams Inlet show fluctuations in populations over time, although differences in sightability may be a factor (Table 28). Only 1 year of survey data are available for Taylor Bay (Lewis and White 2016) and no data specific to Dry Bay are available. The Gustavus population increased dramatically during the 1990s but seems to have stabilized in recent years (White et al. 2006, Scott 2010a). However, due to an overall lack of data for three of the four survey areas, a *Condition Level* could not be assigned for this measure.

Age/Sex Ratios

The project team assigned this measure a *Significance Level* of 2. Recent surveys suggest that the calf:cow ratio is declining in the Gustavus area. All documented sex ratios are below the ADFG management objective of 25 bulls:100 cows (Barten 2008, Scott 2010a, Lewis and White 2016). Since information regarding moose age/sex ratio is limited for most of the park and preserve, a *Condition Level* was not assigned for this measure.

Weighted Condition Score

A *Weighted Condition Score* is typically not assigned for components that have data gaps for more than half of the identified measures. However, GLBA staff indicate that the current condition of moose in the park is likely of low concern at this time. Accordingly, the Weighted Condition Score for moose was determined to be 0.33. A low confidence border was applied to the condition graphic, indicating that the current assessment of condition is based largely on professional judgement, and is not yet strongly supported by data related to the four selected measures.

Moose			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.33**
Distribution	2	1	
Abundance	2	n/a	
Population Trends in Select Areas	3	n/a	
Age/Sex Ratios	2	n/a	

4.8.6. Sources of Expertise

- Tania Lewis, GLBA Wildlife Biologist

4.8.7. Literature Cited

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4.9. Bears

4.9.1. Description

Bears are known to hold biological, cultural, and economic importance throughout southeast Alaska (Schoen and Gende 2006). Both brown bears (Photo 23) and black bears occur in GLBA. Adult black bears average 0.73 m (29 in) in height at the shoulder and 1.5 m (60 inches) in length (Johnson 2008). Adult males weigh approximately 81-91 kg (180-200 lbs) (Johnson 2008). Brown bears are typically larger than black bears, with males weighing up to 680 kg (1,500 lbs) in coastal areas and 227 kg (500 lbs) in inland areas (Eide and Miller 2008). Male brown bears are 30-50% larger than females on average, and both sexes have a prominent shoulder hump that is less pronounced in black bears (Eide and Miller 2008). In GLBA, brown bears are most common in recently deglaciated areas and along the outer coast, while black bears frequent forested areas, particularly in the lower bay (NPS 2013). Coastal areas are important foraging areas for both bear species, especially after den emergence in the spring (Partridge et al. 2009). A preliminary diet analysis found that unlike brown bears in other regions of North America, a majority of the diet of GLBA brown bears is plant material, with a significant contribution from marine organisms (e.g., salmon, marine invertebrates) and little terrestrial meat (e.g., moose or small mammals) (Mowat and Heard 2006).

Photo 23. Brown bear along the GLBA coast (NPS photo by Tania Lewis).

In 1939, when GLBA was still a National Monument, its boundary was greatly expanded to provide a brown bear sanctuary in response to public concerns over territorial game management policies

(NPS 2013). Bear management is currently a high priority at GLBA because visitors value bear-viewing opportunities, but bear-human interactions in the park are a major safety concern (NPS 2013). GLBA is known to support “glacier bears,” an extrememly rare blue-gray color phase of black bear, in addition to the more common black and cinnamon color phases; the chance to view a glacier bear may attract some visitors to the park (NPS 2013).

4.9.2. Measures

- Distribution of black bears
- Distribution of brown bears
- Number of black bears in the population
- Number of brown bears in the population
- Number of black bears harvested every year
- Number of brown bear harvested every year

4.9.3. Reference Conditions/Values

Reference conditions for GLBA’s bear populations are difficult to determine, given the dynamic nature of the landscape and shifting habitats. In addition, bear research in and around the park has been limited to habitat and activity assessments as well as population genetics analysis of brown bears. As a result, a reference condition cannot be determined for distribution or number of brown or black bears at this time. Regarding harvest numbers, the ADFG seeks to limit harvest to no more than 10% of the bear population (ADFG 2011). This goal can be used as a general reference condition, but since population sizes are uncertain within and around the park, actual allowable harvest numbers are unknown.

4.9.4. Data and Methods

The GLBA Bear Management Plan (NPS 2013, p. 2) “provides direction for the management of people, brown bears, and black bears for the purposes of preventing bear incidents and providing opportunities for bear viewing in Glacier Bay National Park and Preserve.” This document contains background information on bear-human interactions in the park, summarizes past and current research efforts, and outlines population threats and research needs (NPS 2013). The GLBA “bear team” also produces annual reports summarizing bear activity and incidents in areas of interest (e.g., Bartlett Cove and River, Gustavus, and Dry Bay) (Photo 24; Behnke et al. 2011, 2012, 2013).

Photo 24. A habituated subadult black bear in the Bartlett Cove maintenance yard; this bear was regularly encountered by the GLBA bear team in 2013 (NPS photo).

The ADFG publishes regular bear management reports, approximately every other year for brown bears, and every 3 years for black bears. These reports present harvest data and population information (if available) by GMU. GLBA contains portions of three GMUs: 1C, 1D, and 5A (see Figure 57). Reports relevant to GLBA are Scott (2009, 2011a) and Bethune (2011) for brown bears, and Scott (2011b, c) and Crupi (2011) for black bears.

During the 1980s, several field seasons were spent surveying bear activity in various areas of GBP where bear-human conflicts had occurred (e.g., Sharman and Kristensen 1982, Publicover 1985, Blackie 1989). The goal was to gain information on bear distribution, seasonal movements, and preferred habitats in order to better protect both humans and bears from negative interactions. Areas of interest included Sandy and Spokane Coves on the east shore (Lee 1985, Publicover 1985, Blackie 1989) and the upper west arm/Tarr Inlet (Warburton 1988, Wolfe 1989, Climo and Duncan 1991). Also in the 1980s, Bazilchuk and Paul (1986) surveyed the Gustavus-GLBA boundary area to identify areas with high black bear use.

In the summers of 2004 and 2005, the NPS and USGS studied bear use of shorelines in the Sandy Cove and Tarr Inlet closure areas to determine if continued closures of these areas were justified (Partridge et al. 2009). Several non-invasive survey methods were tested for monitoring bear activity (remote cameras, bear sign surveys, and hair sample collection). Researchers visited eight different study areas of management concern (Figure 63) bi-weekly between June and August of each year (Partridge et al. 2009).

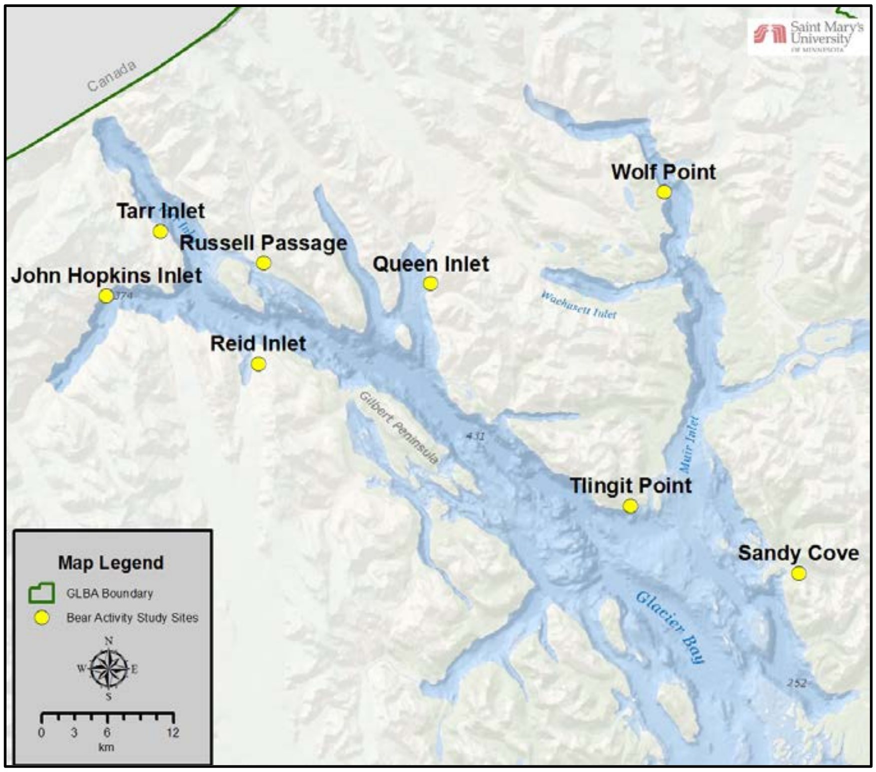

Figure 63. Partridge et al. (2009) bear activity study sites around Glacier Bay, 2004-2005.

Lewis (2012) used occupancy modeling based on 2009-2010 observations to estimate brown and black bear distribution along the shorelines of Glacier Bay and conducted genetic analyses of brown bear hair samples to provide insight into population structure in the park. Sampling sites were sections of shoreline with uplifted beach meadows at perennial stream mouths (Lewis 2012). Bears were monitored through a combination of direct observations, track identification, and analysis of hair samples. Hair samples were collected from bear "rub trees," which were fitted with pieces of barbed wire to increase the likelihood of obtaining hair samples (Photo 25; Lewis 2012).

Photo 25. A rub tree equipped with barbed wire in Queen Inlet, GLBA (USGS photo from Partridge et al. 2009).

Pinjuv (2013) used non-invasive techniques (rub trees and hair traps) to estimate the minimum black bear population on the Gustavus forelands (Figure 64). Hair samples were collected from the spring through the fall of 2011 and 2012. Genetic analysis of these samples can be used to estimate population densities, abundance, and trends through mark-recapture statistics. Pinjuv (2013) also researched harvest trends in the area to determine if recent harvest levels are sustainable.

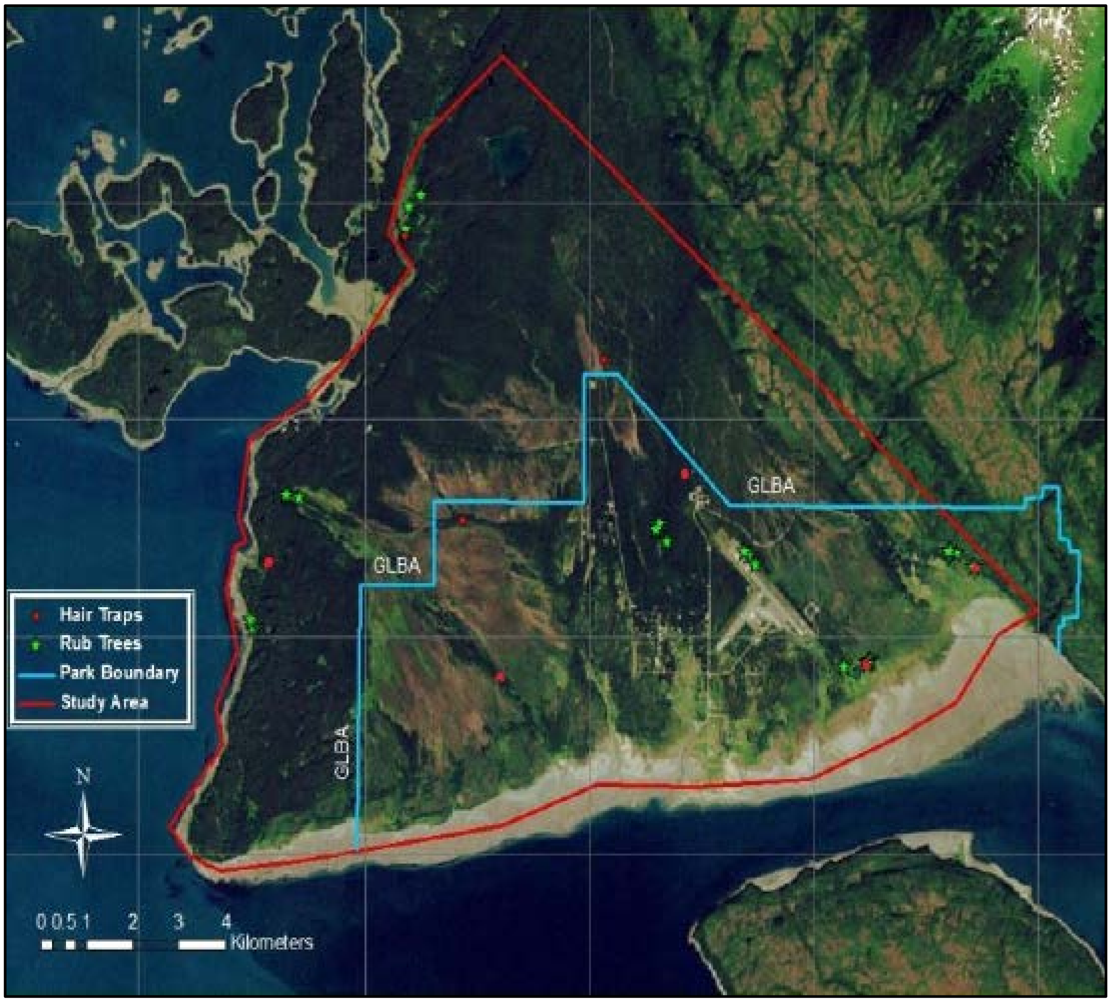

Figure 64. Pinjuv's (2013) study area in the Gustavus forelands. Sampling locations are indicated with red and green stars.

4.9.5. Current Condition and Trend

Distribution of Black Bears

The distribution of bears in GLBA is likely related to food resource availability in different vegetation and stream successional stages, and bear distribution can vary between years in response to food availability (Publicover 1985, Lewis 2012). Interaction between black and brown bears may influence species distribution, but recent research shows that the two species overlap within the park more than previously thought (Lewis 2012). Black bears are most common in the forested areas of lower Glacier Bay, and they also occur around Dry Bay (NPS 2013). Lewis (2012) found that black bear distribution was strongly linked to closed forest cover. Lewis (2012) classified study sites by successional stage; black bears were documented in 100% of young forest units (glaciated 150-260 years ago), 75% of old forest sites, and 44% of sites deglaciated 80-150 years ago, but *not* at sites deglaciated for less than 80 years. This relationship suggests that black bears could expand their range north as the landscape in northern Glacier Bay matures from early, open successional stages to closed scrub and forest (Lewis 2012). The locations of 2009-10 black bear sightings used in Lewis's (2012) analyses are shown in Figure 65.

Figure 65. Black bear sightings along Glacier Bay, 2009-10 (NPS 2013).

Individual surveys over time have identified areas of high black bear use within GLBA, which may be closed to camping for a portion of the year to prevent bear-human interactions. For example, high concentrations of black bears have been documented in Sandy and Spokane Cove (Figure 66) in the spring and early summer (Blackie 1989, NPS 2013). This is likely due to a natural geographic "funnel" of low elevation terrain with three creek systems and the high density of prime black bear forage plants (NPS 2013). In a survey of Spokane Cove, Publicover (1985) sighted most bears along streams, which serve as regular travel corridors, and noted that salmon were spawning in Spokane Creek. Similarly, Bazilchuk and Paul (1986) noted that areas of high black bear use along the park-Gustavus boundary had a combination of high quality forage, topography that channeled movement, and distance from human activity. Areas along creeks (e.g., Homesteader Creek) and coastal beaches (Gustavus beach, west and southwest of the town) tended to be high bear use areas (Bazilchuk and Paul 1986).

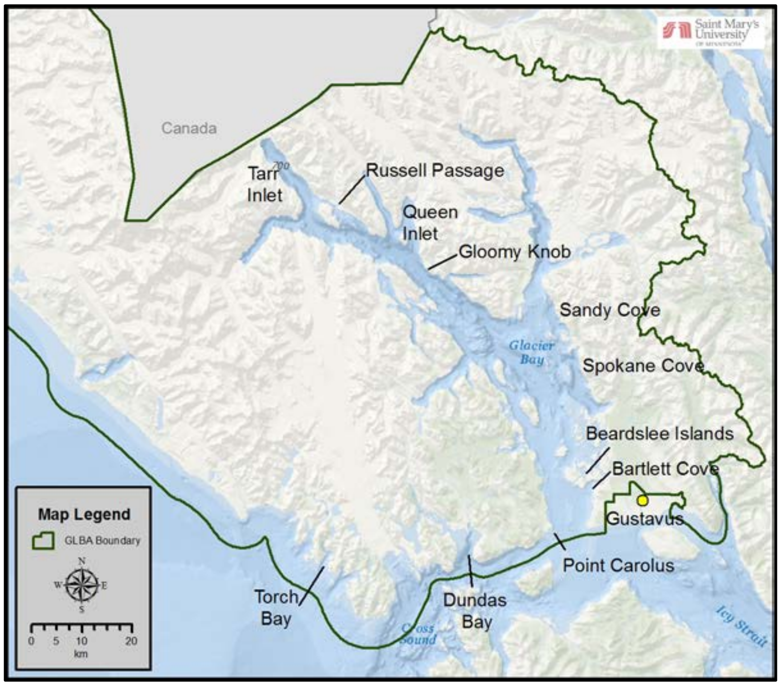

Figure 66. Locations in GLBA with high bear activity or increased frequency of sightings over the past few decades.

Black bears, most often subadults and females with cubs, do occur in Bartlett Cove (NPS 2013). High quality bear habitat exists in the Dry Bay area (the preserve portion of GLBA), but black bear sightings are generally rare, perhaps due to displacement by brown bears (NPS 2013).

<u>Distribution of Brown Bears</u>

As with black bears, brown bear distribution is likely related to food resource availability and may vary from year to year (Lewis 2012). Geographical features such as glaciers and fjords, which can limit dispersal, may also influence distribution (Warburton 1988, Lewis 2012). Brown bears generally dominate in recently deglaciated areas in upper Glacier Bay and along the outer coast up to Dry Bay (NPS 2013). The Dry Bay area in the northwest of GLBA supports high quality bear habitat, with multiple anadromous fish streams, a variety of vegetation, and occasional fish and marine mammals that wash up on the coastal shore (NPS 2013). In the past decade, brown bear frequency has appeared to increase in southern parts of the park, including the Dixon Harbor/Torch Bay, western Dundas Bay, Point Carolus, and lower Beardslee areas (Figure 66; NPS 2013). Brown bears had been reported in the Gustavus and Bartlett Cove areas during the 1920s and 1930s, but were absent from the area from the 1960s through the 1990s, perhaps due to extermination by settlers (Lewis 2012, NPS 2013). The first repeated brown bear sightings on the Gustavus forelands in many decades occurred in 2010, and sightings in the Gustavus and Bartlett Cove/Bartlett River areas have been increasing since that time (Lewis 2012, Behnke et al. 2013, NPS 2013). Factors contributing to changes in brown bear distribution over time include deglaciation providing access to new areas, vegetation and stream succession leading to new plant and prey sources, and immigration through newly opened travel corridors (NPS 2013). Moose population increases in south Glacier Bay likely expanded the prey base and scavenging opportunities for brown bears in this area (Lewis 2012).

In an investigation of shoreline bear distribution, Lewis (2012) found that brown bears occupied all of the selected stream mouth study areas, with highest use in recently deglaciated and old growth forest areas. The locations of 2009-10 brown bear sightings used in Lewis (2012)'s analyses are shown in Figure 67. Areas expected to be highly suitable for brown bears included Russell Passage, Sandy Cove, and Queen Inlet (Figure 66; Lewis 2012). Areas noted for high brown bear use by other observers include the upper west arm/Tarr Inlet (Warburton 1988, Climo and Duncan 1991) and the Vivid Lake outlet near Gloomy Knob, which supports one of the few salmon runs in the west arm of Glacier Bay (Behnke et al. 2011, 2012).

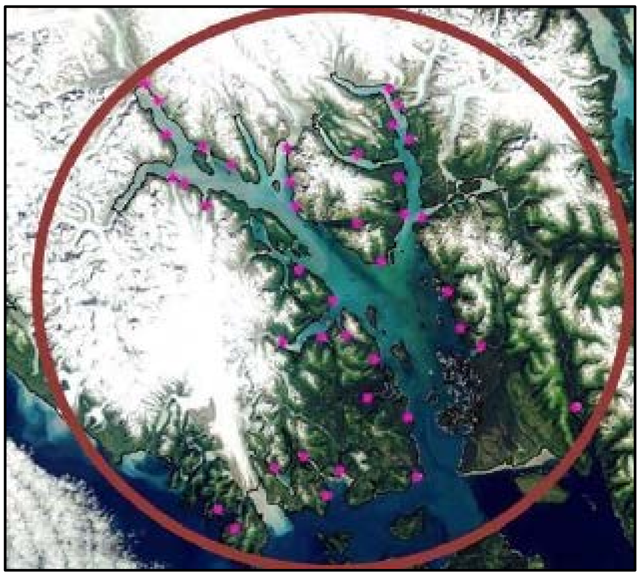

Figure 67. Brown bear sightings along Glacier Bay, 2009-10 (NPS 2013).

Number of Black Bears in the Population

Little is known about black bear population size in GLBA and surrounding areas (Scott 2011b, Lewis 2012). Aerial surveys are not practical due to the species' preference for forested habitats (Scott 2011b, c). Lewis (2012) speculated that bear numbers in GLBA are relatively low due to limited habitat availability, as most of the park is covered by ice, rock, or marine waters. Based on studies of similar terrain in Washington, the ADFG estimates that black bear densities in southeast Alaska are 1.3-1.5 bears/forested mi^2 (Scott 2011b, NPS 2013). Using a value of 0.57 bears/ forested km^2 (1.5 bears/forested mi^2) and an approximate closed forest area of 1,042 km^2 in GLBA (calculated by Lewis [2012] from Boggs et al. [2008]), the estimated population of black bears in GLBA's forests is 594 individuals (Lewis 2012).

Photo 26. Black bear cubs in GLBA (NPS photo).

Pinjuv (2013) conducted genetic analysis of black bear hair samples to estimate the minimum black bear population on the Gustavus forelands. Genetic analysis identified 33 different black bears within the study area. Based on the Huggins linear logistical model, the total population in the study area was estimated at 54.5 ± 10.3 individuals (95% C.I. = 41.6-84.8) (Pinjuv 2013).

Number of Brown Bears in the Population

Similarly, the brown bear population size in GLBA and the surrounding areas has been understudied and little is known about total population size estimates (Bethune 2011, Scott 2011a, Lewis 2012). Population numbers are thought to be relatively low in the area due to limited habitat availability (Lewis 2012). Brown bears throughout North America typically have lower densities and larger home ranges than black bears (Lewis 2012), suggesting that the GLBA brown bear population is likely smaller than the black bear population. Based on anecdotal reports and harvest data, the ADFG believes the brown bear population in GMU 1 (most of GLBA) is stable or possibly increasing (Scott 2009, Bethune 2011). Biologists estimate that the population in GMU 5A, approximately half of which consists of the northern part of the park and preserve, is around 522 bears (Scott 2011a). This is based on density estimates of 0.31 bears/ km^2 (0.5 bears/mi^2) (from Admirality and Chichagof Islands) and the area of available habitat within the GMU (Scott 2011a). GMU 5A includes the Yakutat Forelands, which are thought to have the highest brown bear densities in mainland southeast Alaska (Schoen and Gende 2006, NPS 2013).

Photo 27. Young brown bear foraging along the GLBA coast (NPS photo).

Hildebrand et al. (1999) found that salmon availability is one of the most important factors influencing coastal Alaskan bear population parameters, including body mass, litter size, and population density. Since deglaciation is relatively recent throughout much of GLBA, salmon resources have not yet developed as extensively in GLBA as in other parts of coastal Alaska (NPS 2013). Mowat and Heard (2006) estimated that just 31% of GLBA brown bears' diets come from marine sources (e.g., salmon). Using this number and equations from Hilderbrand et al. (1999), NPS (2013) estimated a brown bear population density for GLBA of 0.2 bears/km^2. This may be an overestimate since sources other than salmon (e.g., shellfish, barnacles) likely contribute to the marine portion of brown bear diet (NPS 2013).

Brown bear population numbers have occasionally been studied in selected areas of GLBA. Using genetic analysis of hair samples from eight sample sites in upper Glacier Bay, Partridge et al. (2009) identified 34 individual brown bears. The sample site at Russell Passage (in the west arm) had the greatest number of bears with 13 individuals (Partridge et al. 2009). Nine bears were identified at Queen Inlet and six at Reid Inlet, also in the west arm (Partridge et al. 2009). A previous study in

Tarr Inlet (upper west arm) in the late 1980s estimated 21 bears (including cubs) occurring on the east side of the Inlet (Warburton 1988).

Number of Black Bears Harvested Every Year

Hunting is not allowed within the national park boundaries; however, sport and subsistence hunting are allowed in the national preserve area around Dry Bay, around the town of Gustavus, and east of the park on U.S. Forest Service land (NPS 2013). In the state of Alaska, the ADFG typically tracks and reports harvest data by GMU. Although black bears have been hunted around GLBA for many decades, harvest information was not collected until the early 1970s (Scott 2011b). In GMUs 1C and 1D, which include Gustavus and areas east and northeast of the park, black bear harvest has increased since record-keeping began but stabilized in recent years (Crupi 2011, Scott 2011b). In GMU 5a, which includes the GLBA national preserve, harvest increased during the 1980s, decreased in the 1990s, and fluctuated throughout the 2000s (Scott 2011c). Since GMU boundaries include much larger areas than the region around GLBA, harvest statistics for the entire GMU may not be reflective of actual harvest in the vicinity of GLBA. However, this data is included in NPS (2013).

The NPS (2013) and Pinjuv (2013) were able to obtain black bear harvest data specific to Dry Bay (1971-2010) and Gustavus (1990-2011), respectively. In Dry Bay, annual black bear harvest has ranged from zero in several years to a maximum of eight in 1976 (Figure 68; NPS 2013). Mean annual harvest over the period of record is 1.9 bears per year, with a slightly lower mean of 1.4 bears per year from 2001-2010, suggesting a decreasing harvest (NPS 2013).

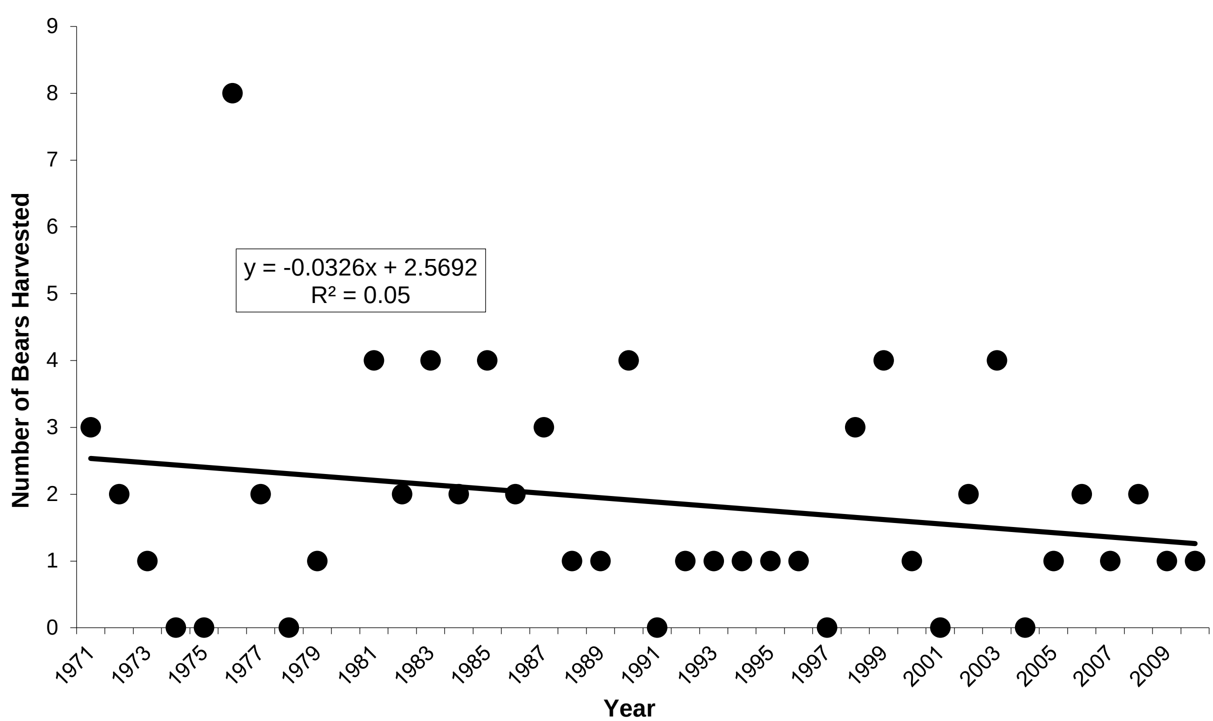

Figure 68. Annual black bear harvest in Dry Bay, GLBA National Preserve, from 1971-2010 (NPS 2013, from ADFG unpublished data).

Around Gustavus, black bear harvest has ranged from zero in three different years to 12 in 2002 (Figure 69; Pinjuv 2013). The annual mean harvest from 1990-2011 was 2.8 bears per year, and from 2002-2011, mean annual harvest was 3.6 bears per year, partly due to high harvests in 2002 (12 bears) and 2011 (9 bears) (Pinjuv 2013, using ADFG data).

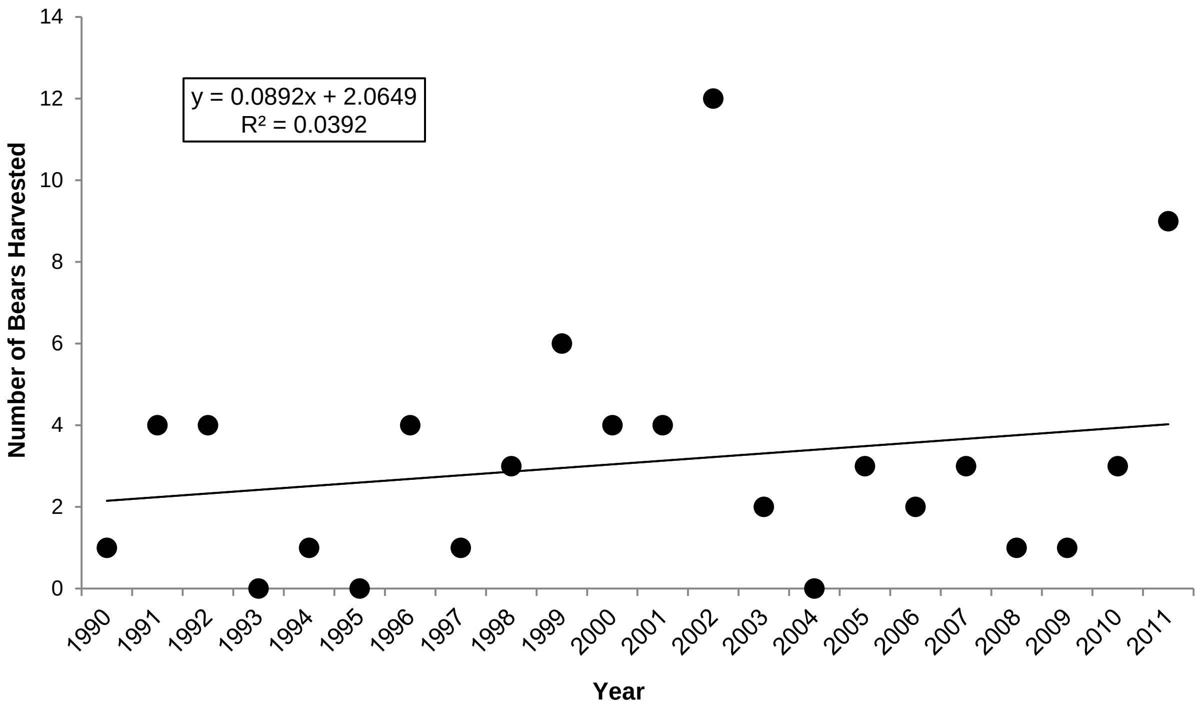

Figure 69. Annual black bear harvest around Gustavus (south of GLBA), 1990-2011 (Pinjuv 2013, from ADFG data).

<u>Number of Brown Bears Harvested Every Year</u>

Brown bear harvest is also tracked by GMU and, as with black bears, harvest statistics for the entire GMU may not be reflective of actual harvest in the vicinity of GLBA. For example, because brown bears did not occur in the Gustavus area (part of GMU 1C) until recent years, harvest data from GMU 1C is not particularly relevant to GLBA. However, harvest data by GMUs that include GLBA are included in NPS (2013). According to NPS (2013), 2012 was the first year a brown bear was legally harvested by a sport hunter in the Gustavus area. NPS (2013) was able to obtain brown bear harvest data specific to Dry Bay (1960-2010) from the ADFG (Figure 70). Mean annual harvest has ranged from one bear to 14 bears (in 2008). The mean annual harvest for the entire period of record is 5.8 bears; mean annual harvest from 2001-2010 is higher at 8.3 bears, indicating increased harvest in the past 10 years (NPS 2013, from ADFG data).

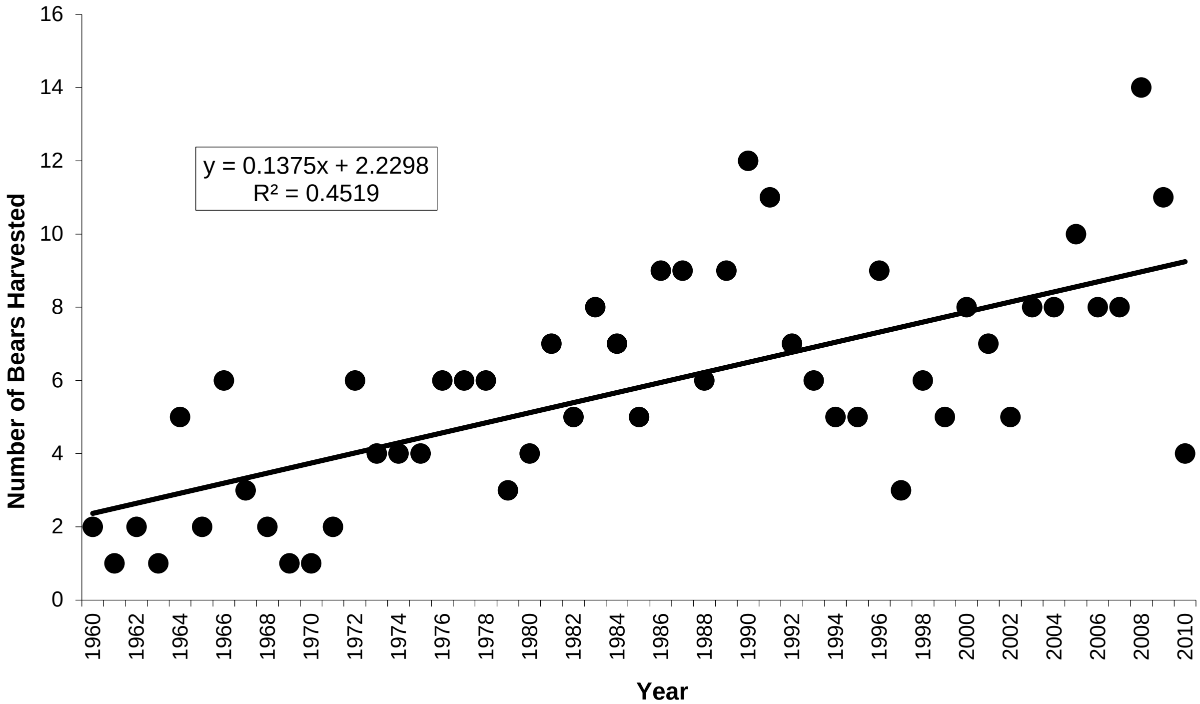

Figure 70. Annual brown bear harvest in Dry Bay, GLBA National Preserve, from 1960-2010 (NPS 2013, from ADFG unpublished data).

Threats and Stressor Factors

Threats to the GLBA bear population include conflicts with humans, harvest (sport, subsistence, and defense of life or property [DLP]), interspecific competition (brown vs. black), and plant succession. Records of bear-human conflict in the GLBA extend back for about a century (NPS 2013). Negative impacts to humans from these conflicts include property damage, injury, and loss of life (Smith et al. 2003). Known fatalities occurred in 1976 and 1980, when lone kayakers camping in the park were killed by a brown bear in the east arm and a black bear at Sandy Cove, respectively (NPS 2013). These and other conflicts have led to camping closures along two large shore areas within the park (Figure 71), for the protection of both visitors and bears (NPS 2013). Other areas where temporary closures have occurred due to bear-human interactions include Gloomy Knob/Vivid Lake outlet and the Bartlett River Trail, which both contain streams that support salmon (Behnke et al. 2012, 2013, NPS 2013). Angler use of the Bartlett River area has increased since the mid-1990s, and bears have obtained fish or other food from these anglers (Murdoch and Soiseth 2007, NPS 2013). Black bear sightings are nearly twice as frequent as brown bear sightings within the park, but the proportion of bear-human conflicts is split nearly evenly between the two species (Smith et al. 2007). No one has been injured by a bear in GLBA since 1980 (NPS 2013).

Figure 71. Camping closure areas in GLBA (Reproduced from NPS 2013).

Human activity in areas of high bear use can impact bears by disrupting their natural activity patterns and/or displacing them from preferred habitats; humans can also contribute to bear injuries and deaths (Smith et al. 2003). Recreational activities, including bear viewing and sea kayaking, are becoming increasingly popular in GLBA. These activities often focus along the bay's narrow beaches, which are also seasonally important areas for bears (Smith et al. 2007, NPS 2013). The potential for bear-human interactions is likely highest at beach areas, including backcountry campsites (Smith et al. 2007). Lewis (2010) found that vessels approaching within 100 m (328 ft) of a bear significantly increased the frequency of stress behaviors exhibited by the bear. A majority of bears approached within this distance fled short distances and some completely left the beach (Lewis 2010, NPS 2013).

According to NPS (2013), preventative management is the most important step in minimizing bear-human conflict. When individual bears are able to access human food and/or trash, they often become "food conditioned" and are more likely to visit areas used by humans (Herrero and Fleck 1989, NPS 2013). Smith et al. (2007) determined that human foods were involved in nearly half of GLBA bear-human conflicts. In 1991, GLBA began requiring the use of bear-proof food storage methods in the backcountry, such as Bear Resistant Food Canisters (BRFC) (NPS 2013). GLBA loans BRFCs to backcountry visitors at no cost (NPS 2015). Despite these improvements, bear-human incidents increased during the late 1990s as backcountry visitation also increased (NPS 2013). In response to this increase, the park constructed a database of bear sightings and incidents, including records dating back to 1932. In 2006, records from this original database were transferred into a new system called the Bear-Human Information Management System (BHIMS). Figure 72 shows the number of bear-human incidents in GLBA from 1959 through 2011, along with backcountry visitor use since 1997 (NPS 2013). Bear-human incidents peaked in 1991, just before bear-proof food storage was required.

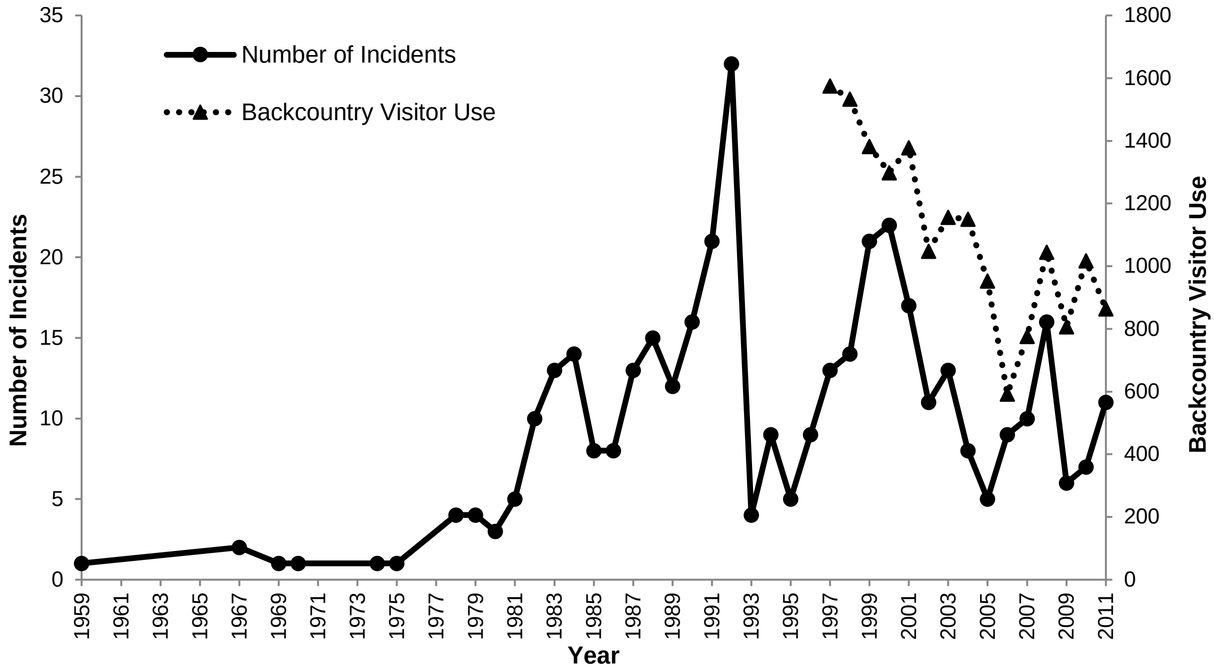

Figure 72. Number of bear-human incidents in GLBA, 1959-2011, and backcountry visitor use, 1997-2011 (NPS 2013).

Bear-human conflicts are common around Dry Bay in the preserve portion of GLBA. The area supports three fishing and hunting lodges as well as a small group of commercial fisherman that live in cabins during the summer (NPS 2013). The relatively high number of people participating in outdoor activities in the area makes bear-human conflict more likely; bears have caused extensive property damage to cabins, boats, equipment, and fishing nets (NPS 2013). Dry Bay's problematic bears have occasionally disappeared in the past, and the NPS and ADFG suspect that residents regularly shoot problem bears and do not report the kills (NPS 2013).

Although hunting is not allowed within the park portion of GLBA, bears can be harvested within the preserve, on private in-holdings, and on lands adjacent to the park (NPS 2013). Harvest data are discussed above (see Figures 68-70). Resource managers are concerned that bear harvest may increase in the Gustavus area, due to increased access through the Alaska ferry system, which recently began service to the community (NPS 2013). The increase in the local brown bear population may also attract more hunters (NPS 2013). A general guideline used by the ADFG in setting harvest objectives is to limit bear harvest to 10% of the population; setting limits is a challenge in areas where bear population size is unknown, such as around GLBA (NPS 2013). Based on Pinjuv's (2013) population estimate of 54 black bears in the Gustavus area, the mean annual harvest of 3.68 bears between 1990 and 2011 meets the 10% objective. However, harvest in five out of those 21 years exceeded the 10% limit (Pinjuv 2013).

Bears are also killed in DLP situations and occasionally due to vehicle collisions. In the case of DLP kills, bear hides and skulls must be turned over to the ADFG. Food-conditioned bears or bears investigating garbage and other human attractants in Gustavus are often (NPS 2013). Some of these nuisance kills could be classified as DLP, but shooters often choose to legally harvest the bears so they can keep the hides (NPS 2013). NPS (2013) notes that several black bears were killed in 2002 after getting into resident's outdoor refrigerators and freezers, but only two out of 14 bears killed were registered as DLPs. Total reported bear kills in the Gustavus area (harvest, DLP, and roadkill) from 1990 through 2011 are shown in Figure 73. Similarly, only one DLP kill has been reported within the GLBA preserve despite many anecdotal reports of nuisance bears killed illegally (NPS 2013). It is likely that bear kill data are lower than actual harvest numbers due to underreporting (NPS 2013, Pinjuv 2013). With increased access and visitation to Gustavus, bear-human interactions are also likely to increase and could lead to higher DLP kill numbers (Pinjuv 2013).

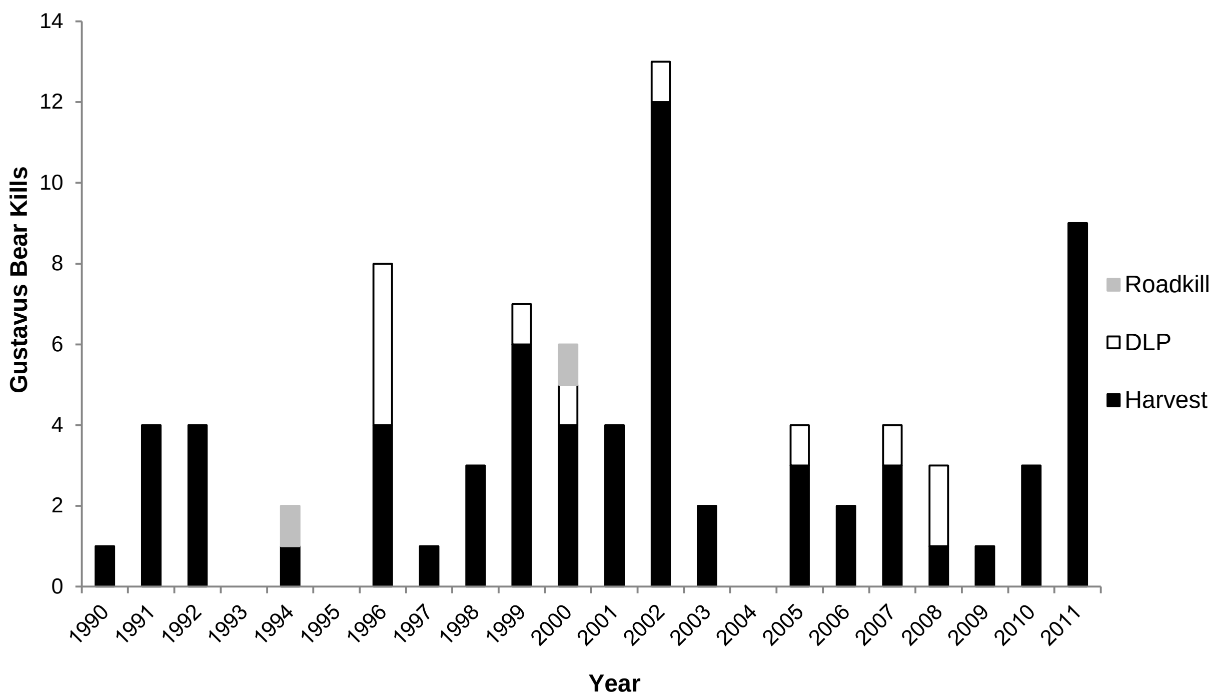

Figure 73. Black bears killed in Gustavus through sport/subsistence harvest, DLP kills, or on roads (ADFG data from Pinjuv 2013).

Since brown and black bear ranges overlap in much of GLBA, there is a potential for competition between the two species to impact the populations. Due to their greater size and sometimes aggressive behavior, brown bears encroaching upon areas previously dominated by black bears could impact the smaller species (Pinjuv 2013). To date, there is no evidence that interspecific competition is limiting the distribution of brown or black bears at GLBA (Lewis 2012). Mattson et al. (2005) found that brown bears have an advantage over black in areas with concentrations of high quality foods at predictable times (e.g., salmon runs). Black bears can dominate in areas with more dispersed and less predictable food resources (e.g., vegetation), due to their smaller size and, therefore, lower food/forage requirements (Mattson et al. 2005, NPS 2013).

As mentioned previously, food availability likely influences bear distribution and population parameters in southeast Alaska (Publicover 1985, Lewis 2012). Since a large portion of bear diet in GLBA comes from vegetation (Mowat and Heard 2006), plant succession can impact the park's bear population. For example, Lewis (2012) noted that brown bear use of recently deglaciated (50-150 years) areas was high, perhaps due to the presence of important plant food sources such as alpine sweetvetch (*Hedysarum alpinum*), locoweed (*Oxytropis campestris*), and russet buffaloberry (*Shepherdia canadensis*). As plant succession progresses, these species will be replaced by others, creating habitat that may not be as suitable for brown bears.

Data Needs/Gaps

Little information is available on the black and brown bear populations of GLBA (NPS 2013, Pinjuv 2013). In particular, the distribution, home range size, population size, and movements of the two species in the park are unknown (Lewis 2012). Black bears are also the least studied big game animal in Southeast Alaska (Schoen and Peacock 2006, NPS 2013). Since the majority of GLBA is intact wilderness with only small areas of human development and hunting limited to the preserve, there has been little need to conduct invasive or disruptive research (e.g., capturing, radio-collaring) for management purposes (Publicover 1985, NPS 2013). If human use increases greatly in GLBA or if human-caused bear mortality on adjacent land becomes significantly higher, such study methods may be warranted (NPS 2013).

Recent developments in noninvasive sampling techniques, such as genetic analysis of hair samples, could be continued to gain insight into bear abundance and distribution, particularly in areas of management concern (e.g., Gustavus and the preserve) (Lewis 2012). Additional areas of research interest include the impacts of brown bears' southern expansion on black bear populations; baseline data collection on glacier bear distribution, abundance, and genetics; and further analysis of bear-human interactions over time to determine if current management strategies are effectively minimizing conflicts (NPS 2013, Pinjuv 2013).

Overall Condition

Distribution of Black Bears

The project team assigned this measure a *Significance Level* of 2. The full distribution of black bears within GLBA is not known. It is likely related to food availability and can vary between years (Publicover 1985, Lewis 2012). Black bear distribution along the bay has been associated with forest cover; the species is rarely seen in open, recently deglaciated areas (Lewis 2012). As a result, black bears could expand their range north as the landscape in northern Glacier Bay matures from early, open successional stages to closed scrub and forest (Lewis 2012). There is some concern that the recent southward expansion of brown bears, particularly in the Gustavus area, could impact black bear distribution. Currently, this measure is of no concern (*Condition Level* = 0).

Distribution of Brown Bears

This measure was also assigned a *Significance Level* of 2. As with black bears, brown bear distribution is related to food availability and can vary between years (Lewis 2012). Over the past decade, brown bear frequency has increased in southern portions of the park (NPS 2013). In 2010, a brown bear was sighted in the Gustavus area for the first time in many decades; sightings in the Gustavus and Bartlett Cove/Bartlett River areas have been increasing since that time (Lewis 2012, Behnke et al. 2013, NPS 2013). Due to this apparent range expansion, brown bear distribution is currently of no concern (*Condition Level* = 0).

Number of Black Bears in the Population

The number of black bears measure was assigned a *Significance Level* of 3. Little is known about black bear population size in GLBA. Numbers are likely relatively low due to limited habitat availability (Lewis 2012). Because population data are very limited, a *Condition Level* could not be assigned for this measure.

Number of Brown Bears in the Population

A *Significance Level* of 3 was also assigned for this measure. Similar to black bears, little is known about brown bear population size, but numbers are likely low due to limited habitat (Lewis 2012). A *Condition Level* could not be assigned for this measure due to limited data availability.

Number of Black Bears Harvested Every Year

The project team assigned this measure a *Significance Level* of 3. Black bear harvests in the Dry Bay area are low and have decreased slightly over time (NPS 2013). Mean annual harvest has increased slightly in the Gustavus area, averaging 3.6 bears annually from 2002-2011 with a maximum of 12 bears in 2002 (Pinjuv 2013). There is some concern that harvest statistics may be lower than actual kill numbers due to underreporting, particularly from nuisance bear kills (NPS 2013, Pinjuv 2013). There is also concern that harvest in the Gustavus area may increase, as accessibility has recently increased due to Alaska ferry service (Pinjuv 2013). However, given that hunting is only allowed in the preserve and on adjacent lands (i.e., the entire park is essentially a “refuge”), harvest numbers are of low concern at this time (*Condition Level* = 1).

Number of Brown Bears Harvested Every Year

This measure also received a *Significance Level* of 3. Brown bear harvests in the Dry Bay area have increased over time, reaching a mean annual harvest of 8.3 bears from 2001-2010 (NPS 2013, from ADFG data). Brown bears did not occur around Gustavus until recently and none were harvested in the area until 2012 when a single bear was taken (NPS 2013). As with black bears, there is some concern that brown bear harvest may be underreported, such as in Dry Bay where problem bears sometimes “disappear” (NPS 2013). Given that brown bears are protected from hunting throughout the park, this measure is currently of low concern (*Condition Level* = 1).

Weighted Condition Score

The *Weighted Condition Score* for GLBA’s bears is 0.20, indicating good condition. While the park-wide black and brown bear populations have not been extensively studied, there is no evidence that would indicate high management concern for either of the species. Given the limited data available, a trend in condition could not be determined.

Bears			
Measures	Significance Level	Condition Level	WCS = 0.20
Black Bear Distribution	2	1	
Brown Bear Distribution	2	0	
Black Bear Numbers	3	n/a	
Brown Bear Numbers	3	n/a	
Black Bears Harvested	3	1	
Brown Bears Harvested	3	1	

4.9.6. Sources of Expertise

- Tania Lewis, GLBA Wildlife Biologist

4.9.7. Literature Cited

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4.10. Mountain Goat

4.10.1. Description

The mountain goat (Photo 28) is a large member of the Family Bovidae that is common in mountainous regions in the lower United States as well as the coastal regions of Southeastern Alaska (Johnson 2008). Although mountain goats prefer mountainous regions, they can be observed at a wide range of elevations and habitats (sea level to alpine) (Vequist 1983). Mountain goats may be mistaken for Dall sheep (*Ovis dalli*); however, Dall sheep do not usually inhabit the coastal regions of Alaska (Johnson 2008).

Photo 28. Mountain goat standing on rugged terrain in GLBA (NPS photo).

Mountain goats are adapted for both cold climates and rugged terrain, and grow a long winter coat in mid-October that lasts through June. After the winter coat is shed, a shorter and smoother coat is grown for summer months (Johnson 2008). Mountain goat hooves are also uniquely shaped to traverse steep, rugged terrain (Johnson 2008).

While mountain goats appear similar for both genders, male goats on average are about 40% larger than females; the average adult female weighs 82 kg (180 lbs) while males weigh 118 kg (260 lbs) (Johnson 2008). Both male and female mountain goats have horns that are short (203 to 254 mm [8 in to 10 in]) and curve backwards (Rideout and Hoffman 1975). However, females have thinner horns with a sharper curve toward the tip (Johnson 2008).

Mountain goat populations are protected from hunting within the national park. However, subsistence and sport hunting is legal within the national preserve (ADFG 2012). Most mountain goats in GLBA GMUs are taken by local subsistence hunters (ADFG 2012). The ADFG establishes individualized management objectives for mountain goats in each GMU and in many cases for each subunit. ADFG (2012) defines GMUs across Alaska in an effort to assess regional game populations and define hunting regulations. Portions of three individual GMUs extend over GLBA: units 01C, 01D, and 05A (Figure 74).

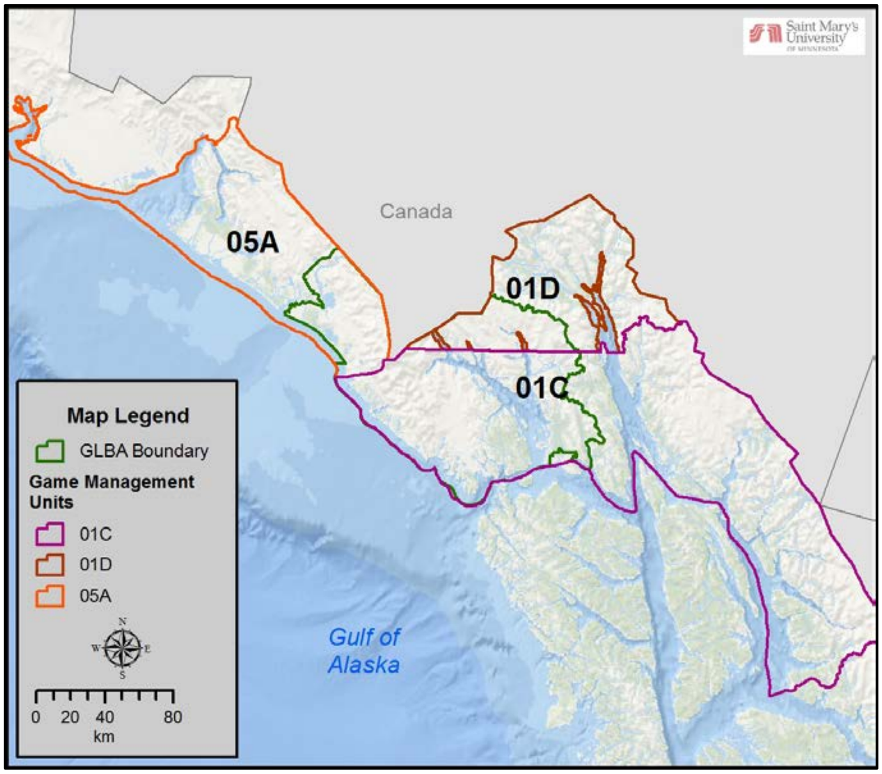

Figure 74. Portions of three individual GMUs extend over GLBA, including units 01C, 01D, and 05A.

4.10.2. Measures

- Distribution
- Abundance
- Population changes in select (surveyed) areas
- Age/Sex ratio

4.10.3. Reference Conditions/Values

A reference condition has not been assigned to the mountain goat component. A lack of historic data related to this component makes it difficult to determine a point in time to compare to the current population. The current condition of this resource will be determined by the best professional judgment of the identified park experts.

NPS (2012) conducted aerial surveys of mountain goats in GLBA in 2012. These surveys covered a larger more complete portion of the park than surveys in the past. These data may serve as a baseline reference for future mountain goat management in GLBA.

4.10.4. Data and Methods

Adams and Vequist (1986) conducted aerial surveys in a small portion of the park from 8 July - 14 July 1985. The study area was divided into 11 count units (Figure 75), which were separated by rivers, glaciers, or mountains. Cliffs and open habitats were searched; however, forested areas, and tall shrub habitats were not due to lack of visibility. Aerial passes were flown by helicopter 152 m (500 ft) above tree, shrub lines, shorelines, and glacial margins. It should be noted that survey tracks were not documented, making it difficult to accurately assess the total area surveyed. Total number of goats, position, number of kids, number of yearlings, number of adults, habitat type, slope, elevation, and total survey time were recorded. Observations were not readjusted to account for sightability.

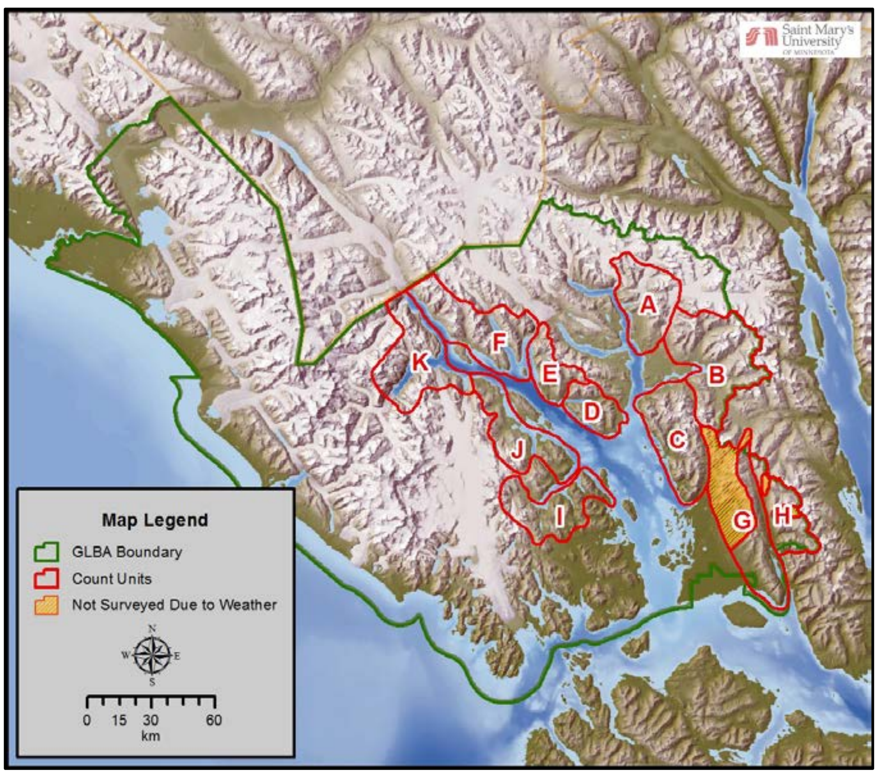

Figure 75. Count units used in July mountain goat surveys in GLBA in 1985 (Adams and Vequist 1986).

ADFG management reports (ADFG 2002, 2004, 2006, 2008, 2010, 2012) provide information on defined population management objectives. These reports establish mountain goat population and composition objectives, as well as monitor temporal trends in population size and composition. ADFG conducts annual aerial surveys in late summer or early fall. The main objective for all three GLBA GMUs is to maintain guideline harvests (not exceeding 6 points per 100 goats observed during aerial surveys). It should be noted that males count as one point and females count as two points. Units 01C and 05 also have an objective to maintain goat densities at 30 goats per hour during fall surveys (as a minimum).

NPS (2012) conducted aerial surveys of mountain goats in the park in 2012 using a Supercub airplane. GPS units were used to record mountain goat observations. Survey tracks, observation points, and survey polygons were recorded to better document the survey route, observations, and area surveyed, respectively. Observers documented the total number of goats, position, composition of groups, habitat type, slope, elevation, and total survey time. Observations have yet to be readjusted to account for sightability.

4.10.5. Current Condition and Trend

<u>Distribution</u>

In 1985, Adams and Vequist (1986) documented mountain goats in nine of the 11 count units in GLBA. Goat observations appeared to be widely distributed throughout the eastern portion of the park with the exception of count unit G. However, it should be noted that a majority of count unit G and a portion of count unit H were not surveyed due to weather (fog). Count unit I was the only unit west of the Bay with mountain goat observations. Count units J and K were also located on the southwest side of the bay in the West Arm of Glacier Bay (Figure 76). Figure 76 displays mountain goat sightings in GLBA in 1985 and 2012.

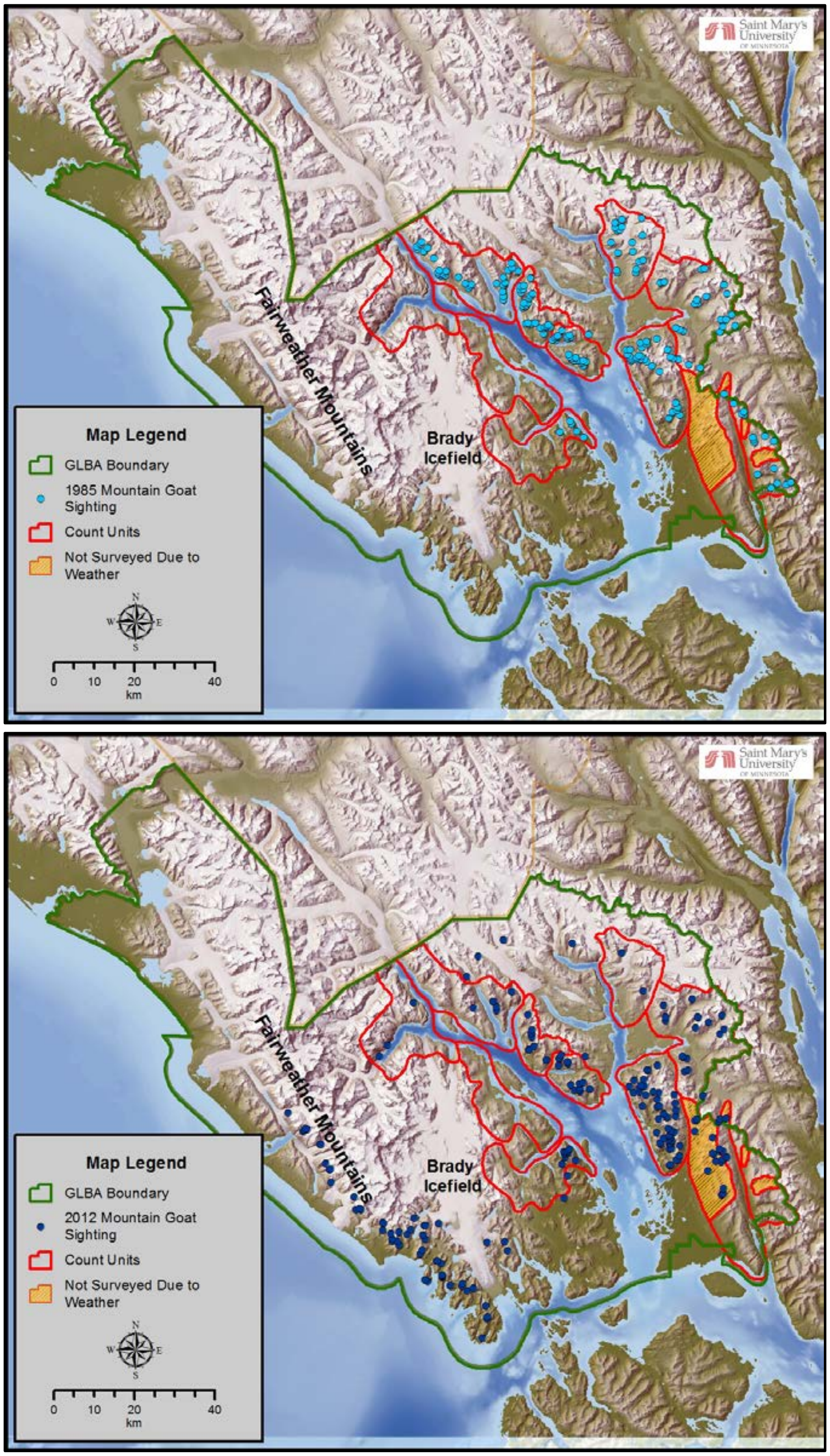

Figure 76. Mountain goat sightings in GLBA in 1985 (top) and 2012 (bottom) (Adams and Vequist 1986, NPS 2012).

NPS (2012) recorded mountain goat distribution in GLBA in 2012; all of the Adams and Vequist (1986) count units were resurveyed except for unit H. Mountain goats were observed in similar areas as the 1985 survey. No goat observations were recorded in count unit J in 1985 or in 2012Goats were sighted further up the East Arm, in count unit K, and along the Fairweather Mountains (from Crillon-Lituya Glacier to Table Mountain). Goats were also sighted inland between Brady Icefield and Dundas River (Figure 76). This survey documented a wider distribution for mountain goats than the previous survey, due to the fact that it covered more area in the western portion of the park.

<u>Abundance</u>

Adams and Vequist (1986) recorded a higher abundance of goats in count units D and E in 1985 than in any other units, with densities over 0.8 goats/km^2 (2 goats/mi^2). Most of the count units had abundances over 0.3 goats/km^2, with the exception of count units G and I (Figure 77, Figure 78). However, abundance in count unit G is likely inaccurate due to a large portion of the unit not being surveyed because of unfavorable weather.

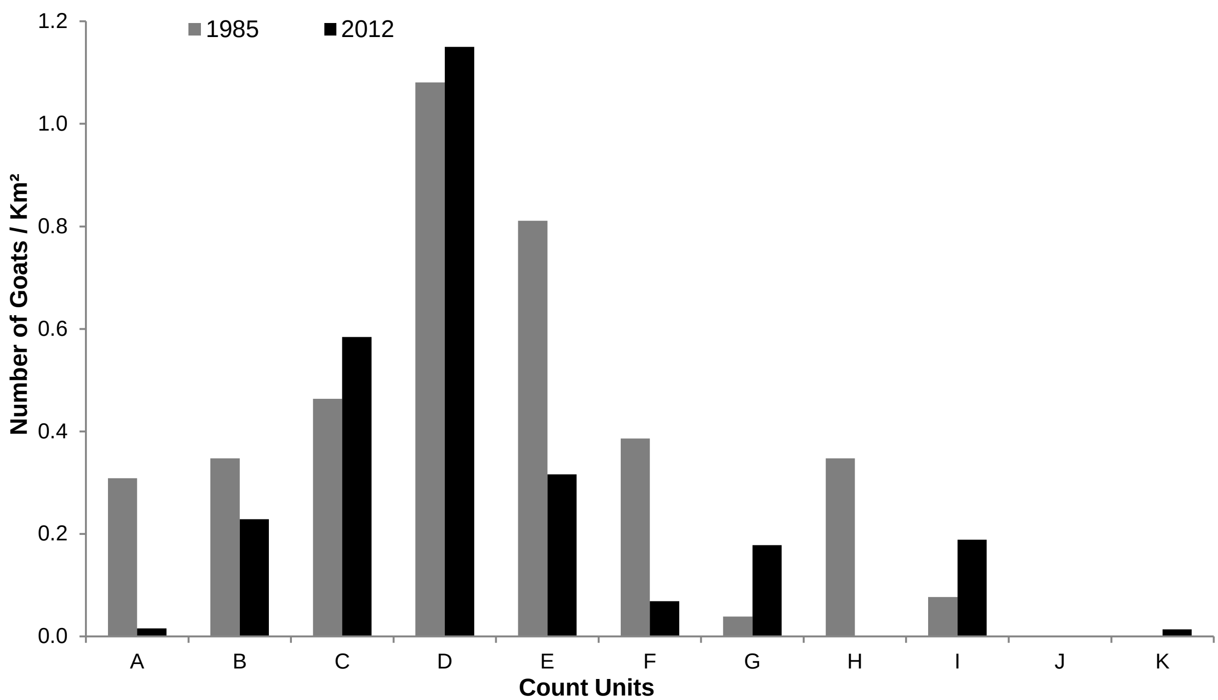

Figure 77. Mountain goat abundance (number of goats/km^2) by count unit in GLBA in 1985 (gray) and 2012 (black) (Adams and Vequist 1986, NPS 2012). Note that much of Unit G was not surveyed in 1985 due to unfavorable weather and Unit H was not surveyed in 2012.

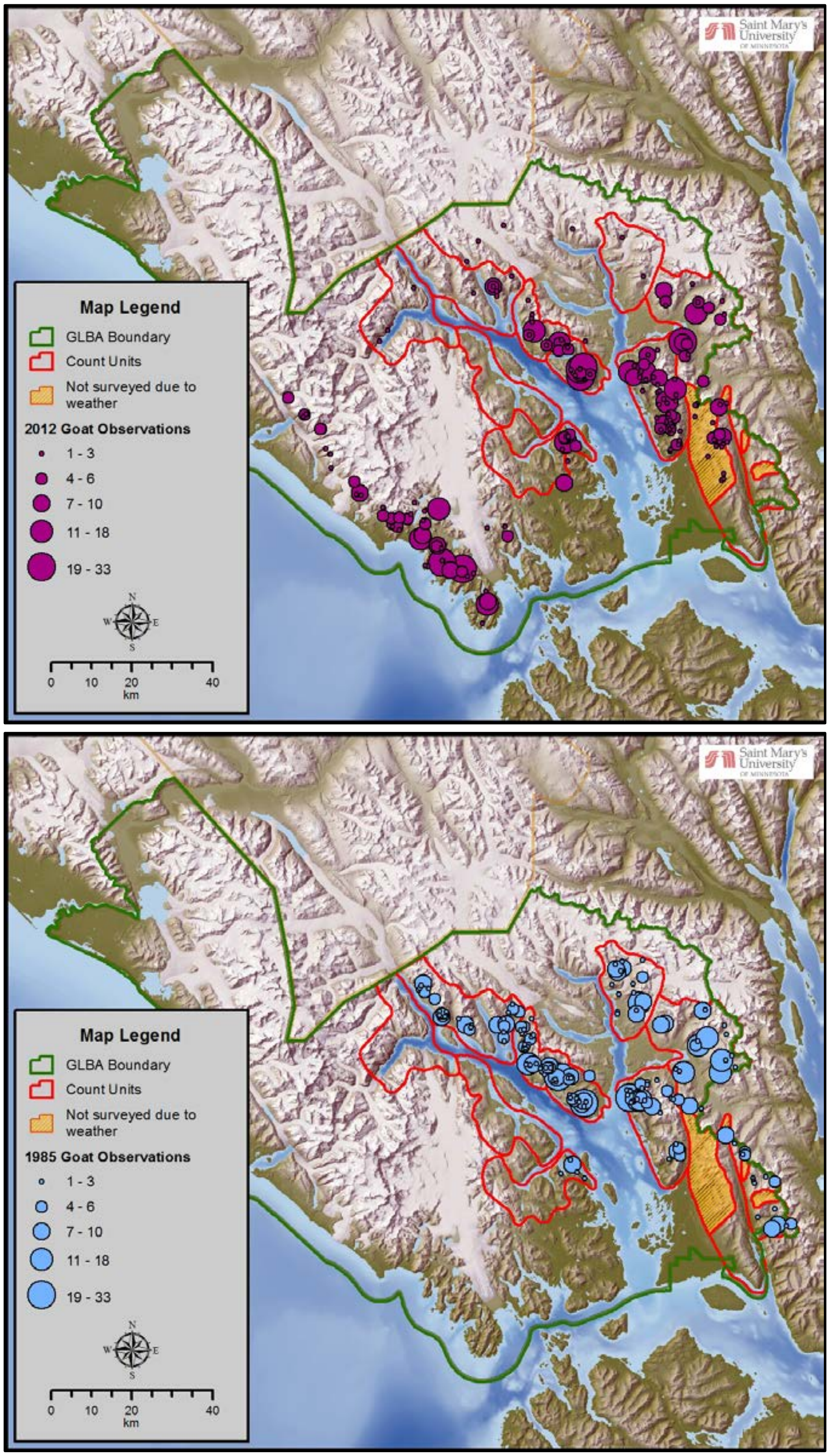

Figure 78. Mountain goat observations in GLBA in 1985 (top) and 2012 (bottom) (Adams and Vequist 1986, Lewis and White 2015).

In 2012, abundance values ranged from 0 goats/km^2 to 1.08 goats/km^2 in the 1985 count units (NPS 2012). While this survey did not focus only on the count units utilized previously, SMUMN GSS analysts calculated abundances for these units so that some comparison could be made between surveys (for units that were surveyed in both years). As with the Adams and Vequist (1986) survey, the 2012 surveys completed by NPS (2012) also recorded the highest abundance of goats in count unit D (Figure 77). Goat abundance decreased in count unit E as well as count units A, B, and F. Abundance increased in count units C, D, G, I, and K. No observations were recorded in count unit J in either survey (Figure 78). Comparisons could not be made for units G, H, or the southwest portion of the park, as these areas were only covered by one of the surveys.

Population Changes in Select (Surveyed) Areas

GLBA park staff selected four survey areas in the park to assess mountain goat population changes: Tlingit Point, Marble Mountain, Mount Wright, and Endicott (harvest - RG015) (Figure 79). These areas were chosen by NPS and ADFG as areas of interest that could be feasibly monitored in the future. The first three areas lie within park boundaries while Endicott borders the park to the southeast.

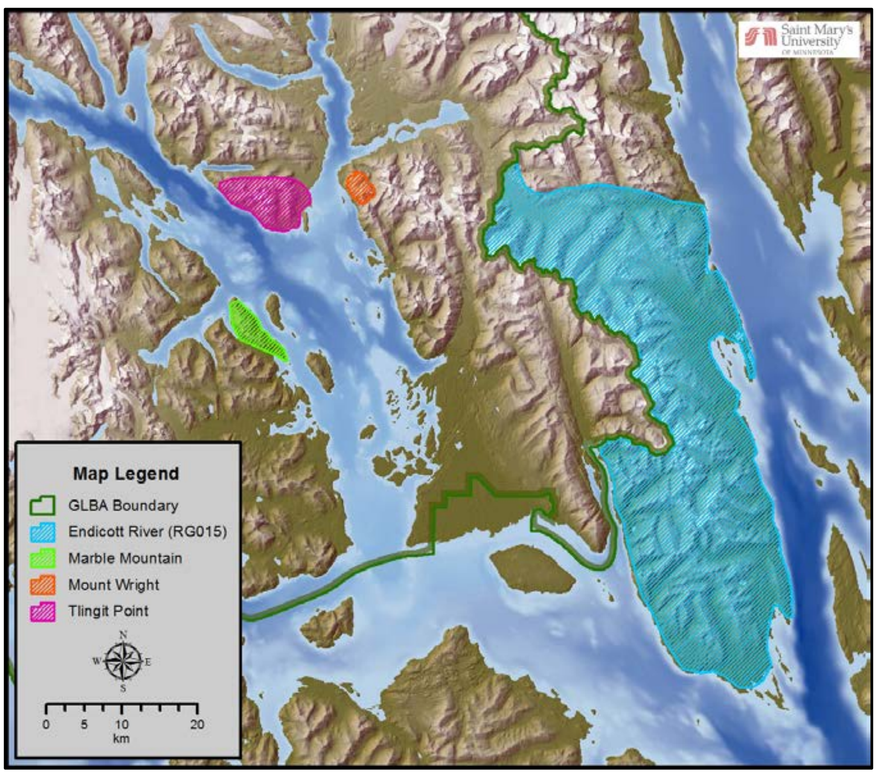

Figure 79. Four select survey areas in or bordering GLBA.

Adams and Vequist (1986) documented 173 goat observation groups in GLBA in 1985. Mount Wright, Marble Mountain, and Tlingit Point were located in count units C, I, and D, respectively. A majority of the observations in count unit C were on or near Mount Wright (Table 29). Over half of the goats observed in count unit D were at Tlingit Point. Of the few observations recorded in count unit I, a majority of the observations (13 goats) were recorded on Marble Mountain.

Table 29. Population changes in mountain goats in GLBA at Mount Wright, Marble Mountain, and Tlingit Point during surveys in 1985 and 2012 (Adams and Vequist 1986, NPS 2012). Surveys in 1985 were conducted by helicopter, whereas surveys in 2012 were conducted by airplane, which may affect differences in sightability between surveys.

Year	Mount Wright	Marble Mountain	Tlingit Point	Total Survey Area
1985	138	13	70	764
2012	168	40	87	841

NPS (2012) documented 201 goat observation groups in these specific survey areas of GLBA in 2012. The number of goats observed increased in three of the four selected survey areas between 1985 and 2012. Similar to the 1985 study, a larger number of goats were observed on or near Mount Wright in count unit C. In count unit D, most goats were observed on Tlingit Point. Goat observations at Marble Mountain increased from 13 goats in 1985 to 40 goats in 2012 (Table 29). Results from this survey should be considered minimum population counts, as sightability models have not been applied to account for the likelihood of spotting or missing individual animals (Lewis, written communication, 8 July 2015).

The Endicott survey area consists of the area just east of the park in the Chilkat Range, from the Endicott River south to Point Couverdon (Barten 2002, Figure 79). For hunting purposes, the ADFG classifies this area as RG015 (Barten 2008). It has not been included in NPS mountain goat surveys, since it lies just outside park boundaries, but has occasionally been surveyed by the ADFG (Table 30; Barten 2008, Scott 2012). The most recent survey in 2006 was limited due to fog in a portion of the survey area (Barten 2008). As a result, conclusions cannot be drawn about current population changes in this area.

Table 30. Mountain goat aerial survey data for the Endicott harvest area (RG015) east of GLBA (Barten 2008, Scott 2012).

Year	Location	# Adults	# Kids	Goats/Hour	Total
1996	East Chilkat Range	215	78	52	**293**
2002	Chilkat Range, Endicott River to Couverdon	152	26	85	**178**
2006	Chilkat Range, Endicott River to Couverdon	203	33	n/a*	**236**

*Data not available for this calculation; survey was only partial due to fog in 20% of survey area (Barten 2008).

<u>Age/Sex Ratio</u>

Adams and Vequist (1986) recorded an average age ratio of 37 kids: 26 yearlings: 100 adults throughout the survey area. In a previous study by Bailey and Johnson (1977), native and long established transplanted goat herds in southeast Alaska had an average age ratio of 34 kids: 17 yearlings: 100 adults. The highest age ratios recorded during the 1985 survey occurred in count unit B (50 kids: 39 yearlings: 100 adults). Count units A and D were also above average with 43 kids: 23 yearlings: 100 adults, and 41 kids: 30 yearlings: 100 adults, respectively. Age ratio in count unit C (35 kids: 20 yearlings; 100 adults) was below the 1986 survey average, but was above historic averages documented by Bailey and Johnson (1977) (34 kids: 17 yearlings: 100 adults). Winter kid survival was not calculated in the 1985 survey. Kid winter survival could be established if surveys were conducted biannually in the park, during fall months and then in late winter. Figure 80 displays the age ratio of mountain goats in 11 count units in GLBA in 1985. It should be noted that no ratios were calculated for units G, J, and K in 1985 or for Units H and J in 2012 due to the lack of observations.

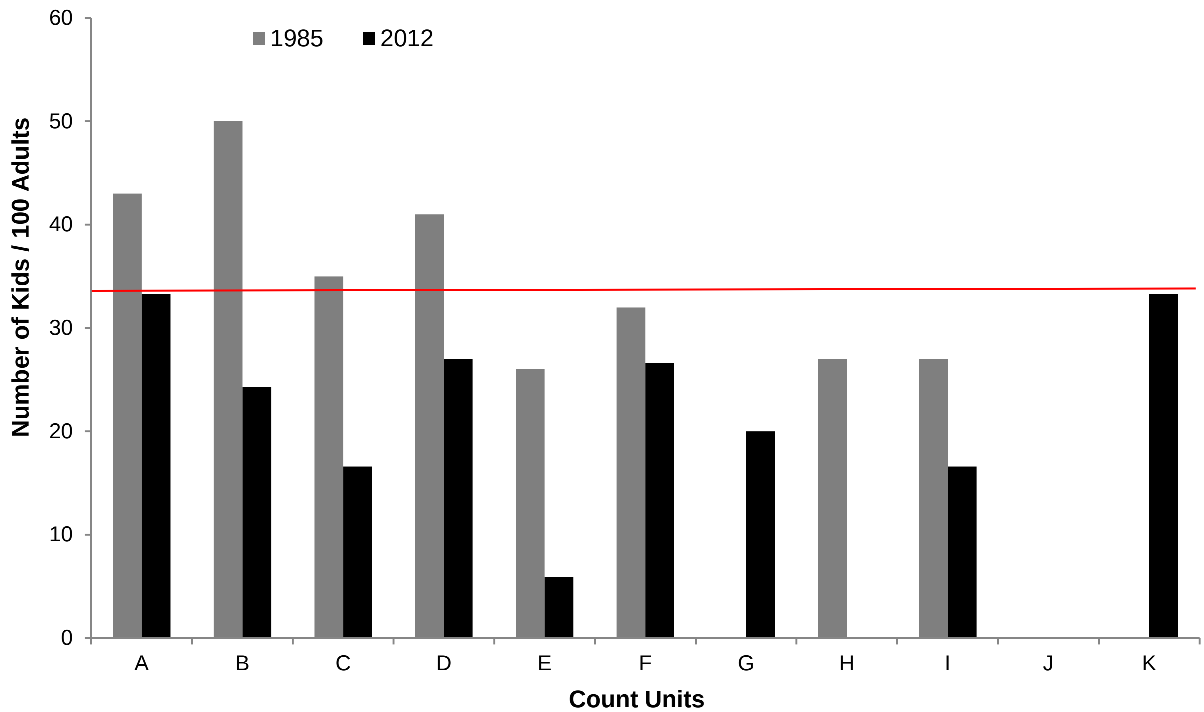

Figure 80. Age ratio of mountain goats in 11 count units from the 1985 survey compared to age ratios from a 2012 survey (Adams and Vequist 1986, NPS 2012). The red line represents the average age ratio (34 kids:100 adults) of native and long-established transported goat herds in Southeast Alaska (Bailey and Johnson 1977).

NPS (2012) recorded age ratios during the 2012 GLBA survey. Age ratios were lower in the 2012 survey than in the 1985 survey. Age ratios ranged from approximately 6 kids: 100 adults (count unit E) to approximately 33 kids: 100 adults (count units A, K) with an overall mean of 21 kids: 100 adults. Goats were not observed in count units J and H, so age ratios were not calculated for these units. It should be noted that winter kid survival was not calculated in this survey. Figure 80 displays age ratios of mountain goats located in the 11 count units established by Adams and Vequist (1986), in GLBA in 2012. No age ratios were calculated for observation points that fell outside of 1985 count units. It should also be noted that sex ratios were not documented during this survey.

Threats and Stressor Factors

There are several threats to the mountain goats in the park. GLBA staff identified four stressors that have occurred or are presently occurring in the park: predation, harvest, rising tree lines, and climate change.

Harvest may be a threat to mountain goats that move outside park boundaries. Harvest of females in Alaska is legal but discouraged because goat populations have declined due to high rates of female harvest (Johnson 2008); this may be because goats have a relatively low reproductive rate (Johnson 2008). ADFG (2012) documented mountain goat harvest within Alaska GMUs (Figure 81; Table 31). The number of females harvested over the years has decreased in units 01C and 05, but seems to have increased in unit 01D. According to ADFG (2012), the total number of goats harvested by GMU appears to be relatively consistent (Figure 81). Harvest will not be a major threat to mountain goats in GLBA as long as the female harvest numbers on surrounding lands remain low.

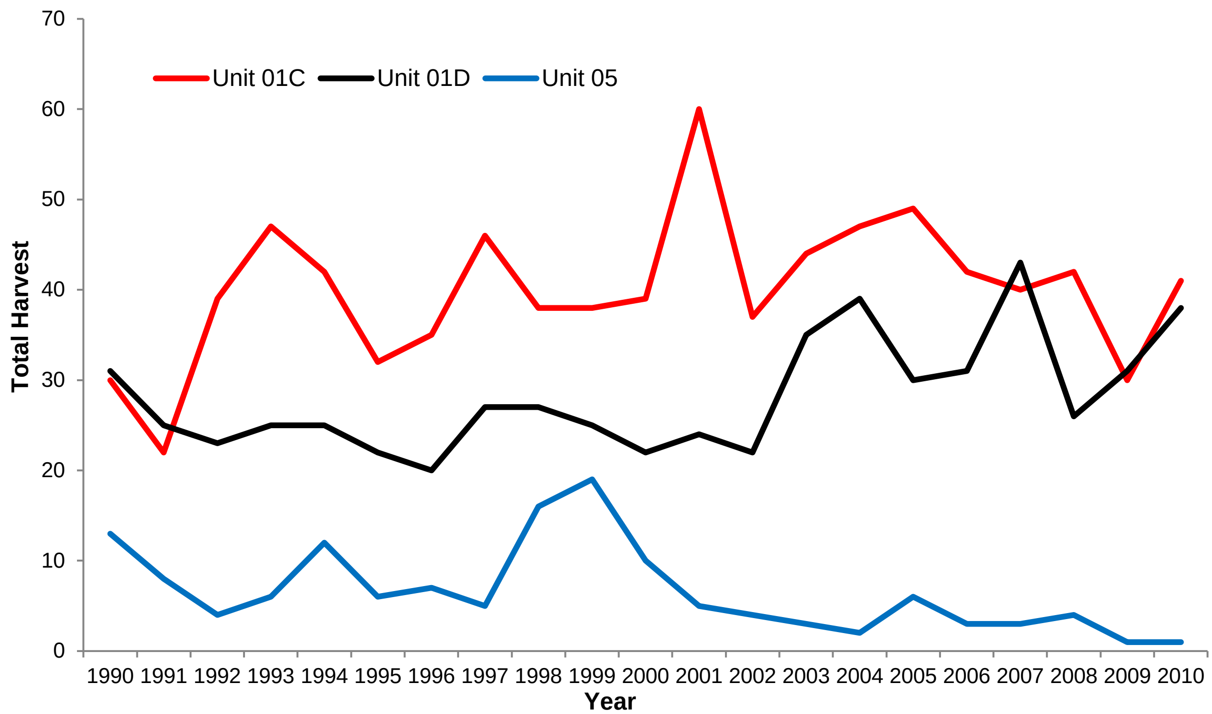

Figure 81. Annual goat harvest during regulatory years (2001 to 2010) in GMU 01C, 01D, and 05 (ADFG 2012).

Table 31. Annual goat harvest during regulatory years (2001 to 2010) in GMU 01C, 01D, and 05 (Recreated from ADFG 2012). It should be noted that an "unknown" column was left out of this table and all totals are correct.

Year	Unit 01C			Unit 01D			Unit 05		
	Males	Females	Total	Males	Females	Total	Males	Females	Total
1990	19	10	**30**	18	12	**31**	11	2	**13**
1991	14	8	**22**	18	5	**25**	4	4	**8**
1992	27	12	**39**	9	11	**23**	2	2	**4**
1993	35	12	**47**	15	8	**25**	4	2	**6**
1994	36	6	**42**	12	12	**25**	6	6	**12**
1995	25	7	**32**	14	8	**22**	4	2	**6**
1996	24	8	**35**	12	8	**20**	5	2	**7**
1997	30	14	**46**	15	12	**27**	3	2	**5**
1998	30	6	**38**	20	6	**27**	9	6	**16**
1999	28	10	**38**	10	15	**25**	10	6	**19**
2000	35	3	**39**	13	9	**22**	7	2	**10**
2001	51	8	**60**	17	7	**24**	5	0	**5**
2002	34	3	**37**	15	6	**22**	3	1	**4**
2003	40	4	**44**	27	7	**35**	2	1	**3**
2004	40	7	**47**	32	6	**39**	1	1	**2**
2005	39	10	**49**	20	10	**30**	6	0	**6**
2006	35	7	**42**	20	11	**31**	3	0	**3**
2007	36	4	**40**	33	10	**43**	2	1	**3**
2008	37	4	**42**	16	10	**26**	4	0	**4**
2009	28	2	**30**	21	10	**31**	0	1	**1**
2010	36	5	**41**	24	14	**38**	1	0	**1**

Predation is another threat to the mountain goats in southeast Alaska. Although there are several predators that prey on mountain goats, wolves are the primary predator in this area (Fox and Streveler 1986, Fox et al. 1989). Mountain goats were the most frequently observed prey item in wolf scats collected between 1973 and 1977, with approximately 62% of the wolf scat examined in GLBA and surrounding areas contained mountain goats remains (Fox and Streveler 1986). Other predators of mountain goats include black bear, brown bear, wolverines (*Gulo gulo*), and coyotes (Fox et al. 1989). Association with steep, rugged and broken terrain is one advantage mountain goats have against large predators (Fox and Streveler 1986).

Climate change is a threat to mountain goats in the park, specifically in the form of increased snowfall in the winter months which may put stress on mountain goats and cause additional winter kills. Kids may be more vulnerable to heavy snowfall than yearlings or adults. Studies in the Lynn Canal area (located east of GLBA) have documented significant declines (45%) in mountain goat survival over a period of time marked by severe winters (ADFG 2014). According to SNAP et al.

(2009), winter precipitation is projected to increase by 7% in GLBA. If more snow and ice were to occur as a result of the precipitation increase, goat survival may decline as a result of reduced health and lack of available food. However, with the increasing temperature it may be more likely that precipitation would fall in the form of rain. Hotter summers may negatively impact mountain goats as well. Over-winter survival of adult mountain goats in coastal Alaska is substantially higher following cool summers as opposed to warmer summers (White et al. 2011). This may occur because green-up is protracted during cooler summers, resulting in higher nutritional quality of forage on summer ranges. This allows for higher rates of over-summer fat deposition, energy stores that are critical for over-winter survival (White et al. 2011).

Rising tree lines, also related to climate change and rising temperatures, may threaten the park's mountain goat population. According to Adams and Vequist (1986), mountain goats frequent lower elevations in the winter months. Alpine meadows and rocky slopes, located above the tree line, are preferable summer mountain goat habitat. Changes in temperature and time of snow melt (early snowmelt) may allow trees to become established in higher elevations, particularly in alpine meadows (Lloyd et al. 2002). This may result in the decrease in open habitat, over-crowding, and competition for limited resources.

Data Needs/Gaps

There are some data regarding the distribution, abundance, population trends, and age ratio of mountain goats in GLBA; however, there are large gaps in the timing of the available data. Adams and Vequist (1986) and NPS (2012) document distribution, abundance, and population composition in 1985 and 2012, respectively, a gap in survey years of 27 years. The length of time between surveys, the difference in surveyed area, and the limited observations in a time series makes assessing trends unachievable. The composition data are lacking consistency in sex ratio, as many of the goat observations are classified as "unclassified adult" instead of identifying individuals as male or female. Ground surveys would provide a better assessment of age/sex ratios in the future. Winter kid survival data are also needed. Kid winter survival could be established if surveys were conducted biannually in the park, during fall months and then in late winter. ADFG (2012) provided age ratios for the GMUs from the 1980s to the present, but the data are not specific to GLBA and there were many gaps over the years among GMUs. Consistent monitoring efforts are needed to better assess the condition of this component and to aid park staff in managing the GLBA mountain goat population.

Overall Condition

Distribution

The project team defined the *Significance Level* for distribution as a 2. Adams and Vequist (1986) documented distribution of mountain goats in 1985. Goats were observed in nine of 11 count units located along the East and West Arm and the Bay. NPS (2012) documented distribution of mountain goats in 2012 (Figure 76), with goats being observed in similar areas as the 1985 survey, and additional observations occurring along the Fairweather Mountains near the coast. As a result, this measure was assigned a *Condition Level* of 0, or of no concern.

Abundance

The project team defined the *Significance Level* for abundance (sightings per year) as a 2. Adams and Vequist (1986) documented abundance of mountain goats in 1985. Goats were higher in abundance in count units D and E than the other nine units (Figures 77 and 78). NPS (2012) data were used to calculate abundance for existing count units to compare to those from 1985. Abundance decreased in four count units (A, B, E, F), but increased in an additional four units (C, D, I, K). Several observations were not included in these calculations because they were not located in or near existing count units. This makes it difficult to assess abundance throughout the park. Given that the abundance data and maps do not show major differences between 1985 and 2012, this measure was assigned a *Condition Level* of 1, or low concern.

Population Changes in Select (Surveyed) Areas

The project team defined the *Significance Level* for population changes in select (surveyed) areas as a 3. GLBA park staff selected Tlingit Point, Endicott, Marble Mountain, and Mount Wright as the selected survey areas. Adams and Vequist (1986) and NPS (2012) provided mountain goat sightings data in three of the selected areas in GLBA (excluding Endicott) for 1985 and 2012, respectively. The number of sightings increased in all three locations from 1985-2012. There were 138 goat sightings on Mount Wright in 1985, and by 2012 sightings increased to 168 (Table 29 and Figure 76). A similar increase in number of sightings occurred at Tlingit Point, where there were 70 sightings in 1985 and 87 sightings in 2012. A larger increase in the number of sightings occurred at Marble Mountain with 13 sightings in 1985 and 40 sightings in 2012. Conclusions could not be drawn regarding population changes in the Endicott area, which is surveyed occasionally by the ADFG, due to a lack of recent data. As a result, this measure was assigned a *Condition Level* of 0, or of no concern.

Age/Sex Ratio (Kid Winter Survival)

The project team defined the *Significance Level* for age/sex ratio as a 2. Adams and Vequist (1986) documented age ratios of mountain goats in 1985. The average age ratio for the entire 1985 survey area was higher than the typical age ratio of native and established herds in southeast Alaska (as defined by Bailey and Johnson 1977). NPS (2012) data allowed for age ratios to be calculated by count unit. Age ratios decreased (i.e., fewer kids per adult) in eight count units by 2012 (Figure 80). Age ratios increased in count units G and K in 2012. The age ratio increase in count unit G most likely occurred because the unit was not fully surveyed in 1985 (due to weather complications) like it was in 2012. The numbers of female and male goats were recorded in1985; however, the number of unclassified adults was also recorded and often outnumbered the male and female counts. No sex ratio data were included in the NPS (2012) data, so an accurate assessment of sex ratio could not be completed. Because data were not available for half of this measure (sex ratios), this measure was not assigned a *Condition Level*.

<u>Weighted Condition Score</u>

The *Weighted Condition Score* for mountain goats in GLBA is 0.10, indicating that the component is in good condition. Data show an expanded documentation of distribution and population changes in select survey areas between 1985 and 2012; however, the expanded documentation is the result of a more comprehensive study of the park. As a result, and because of the limited time series of observations, a trend could not be assigned.

Mountain Goats			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.10**
Distribution	2	0	
Abundance	2	1	
Population Changes in Select Survey Areas	3	0	
Age/sex Ratio	2	n/a	

4.10.6. Sources of Expertise

- Tania Lewis, GLBA Wildlife Biologist
- Kevin White, ADFG Wildlife Biologist

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4.11. Harbor Seals

4.11.1. Description

Harbor seals, a member of the Phocidae family, are the most widely distributed pinniped species in the northern hemisphere (Photo 29). They inhabit temperate and subarctic coastal waters of the North Pacific, North Atlantic, and continuous seas (Scheffer 1958, Shaughnessy and Fay 1977). In the North Pacific Ocean, harbor seals range over approximately 16,000 km (9,942 mi) of coastline from Mexico to Japan, covering a variety of environmental conditions. Within Alaska, harbor seals occupy a geographically extensive range from approximately 172ºE to 130ºW (over 3,500 km east to west) and 61ºN to 51ºN (over 1,000 km north to south). Harbor seals frequently "haul out" of the water to rest, molt, and engage in activities such as pupping and nursing. Haul out habitats used by harbor seals include small islands, sandy beaches, rocky reefs, and glacial ice from tidewater glaciers. Possible reasons for the use of different haul out habitats by harbor seals include proximity to prey resources, reduced risk of predation, reduced wave exposure, and local bathymetry (e.g., water depths) (Brown and Mate 1983, Nordstrom 2002, Montgomery et al. 2007, Grigg et al. 2012). Although harbor seals generally occur in near shore coastal waters, they may occasionally range 100 km or more from shore (Lowry et al. 2001, Womble and Gende 2013b).

Photo 29. Harbor seal on an iceberg in GLBA (NPS photo).

Historically, Alaska's harbor seals have been managed as three stocks: the Bering Sea, Gulf of Alaska, and Southeast Alaska. In 2010, the NMFS and their partners, the Alaska Native Harbor Seal Commission, revised the stock structure and identified twelve separate harbor seal stocks, based largely on genetics (Allen and Angliss 2012). The twelve stocks of harbor seals identified in Alaska are: 1) the Aleutian Islands, 2) the Pribilof Islands, 3) Bristol Bay, 4) North Kodiak, 5) South Kodiak, 6) Prince William Sound, 7) Cook Inlet/Shelikof, 8) Glacier Bay/Icy Strait, 9) Lynn Canal/Stephens, 10) Sitka/Chatham, 11) Dixon/Cape Decision, and 12) Clarence Strait (Allen and Angliss 2012).

Some of the largest harbor seal aggregations in the world and an estimated 15% of Alaska's harbor seal population are found seasonally at glacial ice sites (Bengtson et al. 2007, Jansen et al. 2015). However, population declines at two of the primary glacial ice sites, at Glacier Bay and at Aialik Bay in south-central Alaska, have raised concerns regarding the viability of seals inhabiting tidewater glacial fjord environments (Mathews and Pendleton 2006, Womble et al. 2010, Hoover-Miller et al. 2011). At Glacier Bay, harbor seals utilize a number of terrestrial sites (e.g., reefs, outwash plains, islets) in addition to glacial ice sites. Currently, these sites are distributed throughout the park, from the Beardslee Islands in the lower parts of Glacier Bay to Johns Hopkins Inlet in the upper West Arm of the bay (Figure 82; Mathews and Pendleton 2006, Womble et al. 2010).

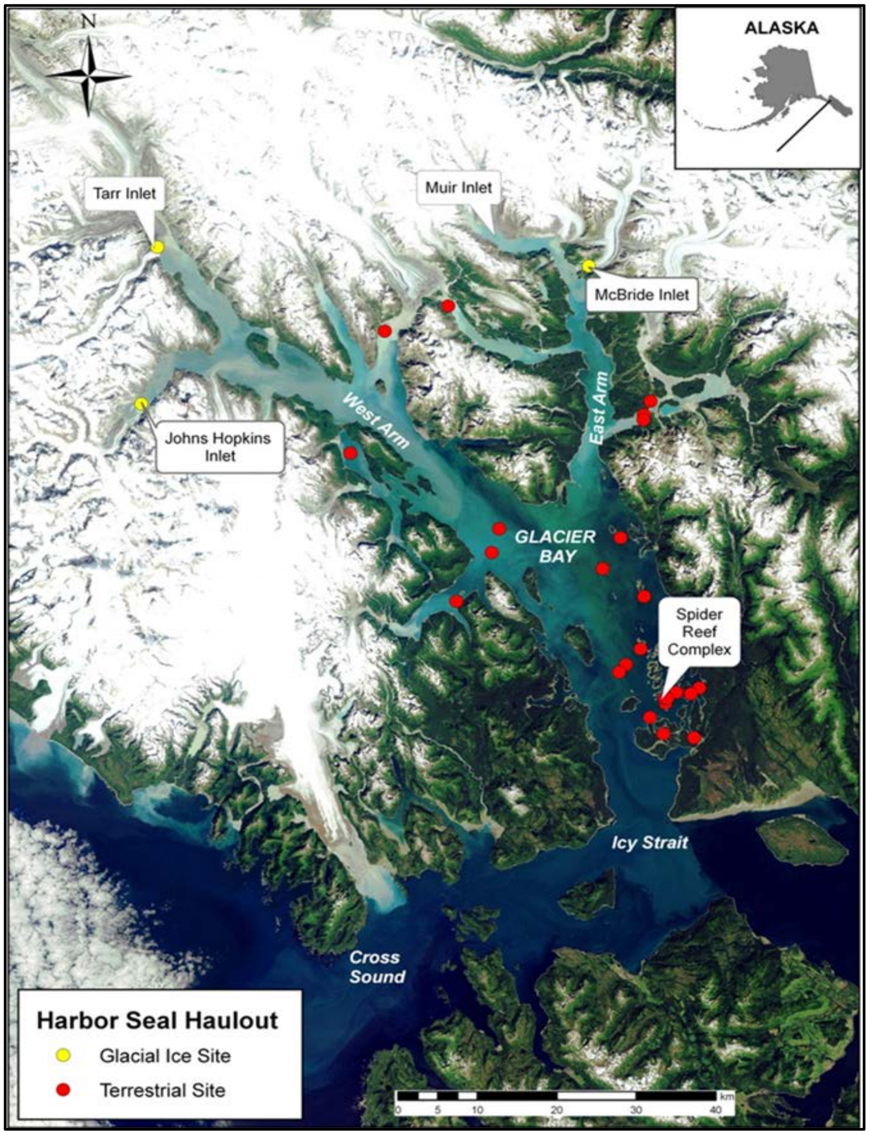

Figure 82. Locations of harbor seal haul out sites in Glacier Bay (Womble et al. 2010).

Harbor seals have also been documented at several haul out sites on the outer coast, including Lituya Bay, Graves Rocks, and Cape Spencer (Jettmar 1984). Large numbers of seals tend to aggregate in small areas at terrestrial sites, providing advantages for monitoring; aerial surveys can be conducted at each site and a single photo can often encompass the aggregate group of seals (Jamie Womble, NPS Wildlife Biologist, written communication, November 2015). In contrast, at the primary glacial ice site in Johns Hopkins Inlet, ice bergs are numerous and can be distributed over large areas (~22.5 km^2), necessitating aerial photographic sampling.

4.11.2. Measures

- Annual estimate of population trend at glacial ice sites during the pupping and molting periods
- Annual estimate of population trend at terrestrial sites during the pupping and molting periods
- Proportion of pups during the June pupping period

4.11.3. Reference Conditions/Values

The reference condition for the two population estimate measures (glacial ice and terrestrial sites) is a stable or increasing 10-year-average population trend. For pup proportion, the reference condition is also a stable or increasing value (most estimates in recent decades from GLBA glacial ice sites range between 30-38%).

4.11.4. Data and Methods

Population monitoring of harbor seals has a long history in Glacier Bay, spanning from the 1970s to the present (Streveler 1979, Calambokidis et al. 1987, Mathews and Pendleton 2006, Womble et al. 2010), and representing one of only a few sites in Alaska where such long-term monitoring efforts for harbor seals exist. The current objectives of harbor seal monitoring in GLBA are to:

- generate annual population trend and abundance estimates for non-pups and pups at terrestrial sites (including small glacial ice sites at Tarr and McBride Inlets) and at the glacial ice site in Johns Hopkins Inlet during the June pupping period and the molting period in late July and August;
- provide a spatially-explicit data structure that provides a permanent record of harbor seal distribution;
- provide data on spatial distribution and numbers of seals to aid in park management decisions that may impact seals (Womble and Gende 2013a).

Formal population monitoring of harbor seals in Glacier Bay was initiated in the mid-1970s, in response to concerns regarding the effects of human impacts on harbor seals (Streveler 1975, 1978, 1979). Streveler (1975, 1979) conducted surveys of harbor seals throughout Glacier Bay using a variety of methods. Land-based counts were conducted in Muir Inlet (1973-1978) and Johns Hopkins Inlet (1975-1978), while fixed-wing aerial surveys were attempted in Muir Inlet (1972, 1973) and at terrestrial sites in GLBA (1972). Finally, the winter and summer distribution of seals was compared using boat-based counts during several winter (1972, 1974, 1977) and summer (1975-1977) seasons (Streveler 1979).

From 1982-1984, Calambokidis et al. (1987) conducted harbor seal population monitoring in Glacier Bay. Survey efforts were focused primarily in Muir Inlet with additional surveys conducted in Johns Hopkins Inlet and at a subset of terrestrial haul out sites in lower and middle Glacier Bay (Calambokidis et al. 1987). Surveys were conducted during the summer months from land-based sites overlooking Muir and Johns Hopkins Inlets.

Harbor seal monitoring at glacial and terrestrial habitats throughout Glacier Bay was reinitiated in 1992 (Mathews and Pendleton 1997). From 1992-2002, harbor seals at terrestrial sites and at one glacial ice site in McBride Inlet were surveyed during the August molting season using aerial photos from fixed-wing aircraft. Harbor seals at the primary glacial ice site in Johns Hopkins Inlet were counted during the June pupping and August molting periods by two ground-based observers using high-powered binoculars mounted on a tripod (Mathews and Pendleton 2006). Survey effort during the molting period ranged from 3-21 days between the 9th and 23rd of August. Mathews and Pendleton (2006) estimated trends using models that included covariates for environmental factors that may influence the haul out behavior of seals including time of day, day of year, tidal height, time to high tide.

Several other ground and air-based surveys for harbor seals were conducted between 1992 and 2002. An extensive aerial survey was conducted from Icy Strait to Icy Bay in August 1996 and included harbor seal haul out sites along the outer coast of Glacier Bay, the Yakutat Forelands, Yakutat Bay, and Icy Bay (Mathews and Womble 1997). Counts of harbor seals were conducted in conjunction with disturbance-related studies in McBride Inlet in 1998 (Lewis and Mathews 2000) and at Spider Reef Complex (1991-1992, 1997-1999) (Mathews and Driscoll 2001). There were also efforts to develop aerial-based photogrammetric methods for enumerating harbor seals in Johns Hopkins Inlet in 1997 and 2001-2002, in collaboration with NOAA-Southwest Fisheries Science Center and the National Marine Mammal Laboratory (NMML) (Mathews et al. 1997, Bengtson et al. 2007).

Beginning in 2004, the NPS (with additional support from the Alaska Fisheries Science Center and ADFG) began conducting harbor seal population monitoring in GLBA (Womble et al. 2010). From 2004-2006, surveys were only conducted at terrestrial harbor seal haul out sites (and two glacial ice sites in McBride and Tarr Inlets) during August using fixed-wing aircraft and following the methodology employed by Mathews and Pendleton (2006). Starting in 2007, harbor seals were surveyed throughout Glacier Bay at both glacial ice and terrestrial sites (Womble et al. 2010, Womble and Gende 2013; Photo 30). In collaboration with the NMML, a new aerial survey method was implemented in Johns Hopkins Inlet in 2007, designed specifically for estimating the abundance and spatial distribution of seals in glacial fjords using high-resolution aerial digital imagery (Ver Hoef and Jansen 2007). During 2007 and 2008, harbor seal surveys were also conducted in Johns Hopkins Inlet using ground-based observers and following the methodology of Mathews and Pendleton (2006) to establish correction factors between the two different survey methods (aerial vs. ground) (Womble et al. 2010).

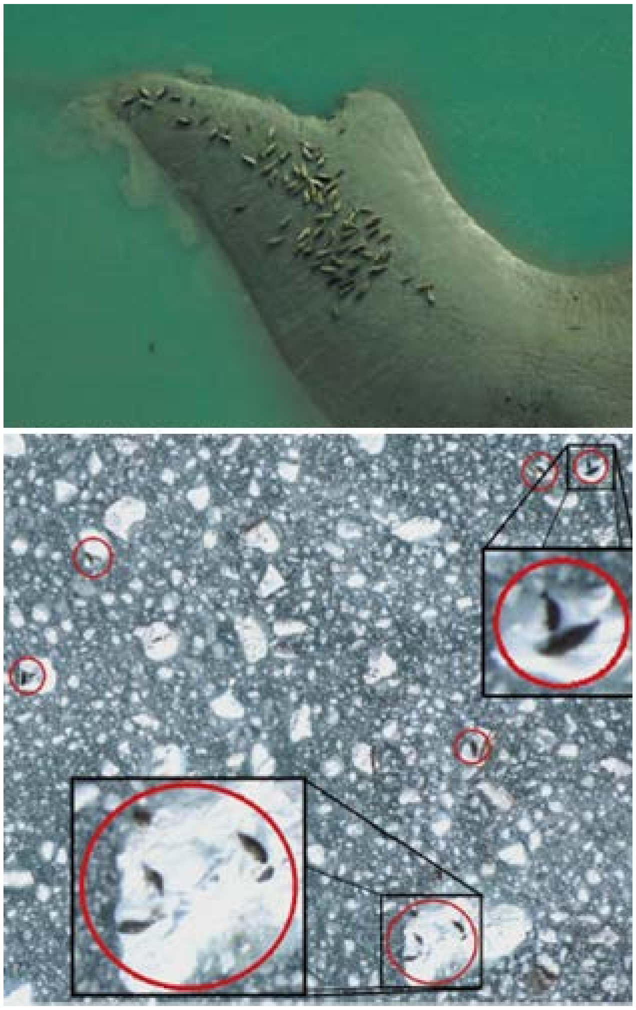

Photo 30. Digital images from 2012 aerial surveys of a terrestrial site (Adams Inlet) and the glacial ice site in Johns Hopkins Inlet (NPS photos by Jamie Womble).

Harbor seal declines documented in the late 1990s and early 2000s led to new research efforts initiated in 2004 and aimed at trying to further understand the basic ecology of seals and identify possible factors that may have contributed to the declines (Womble, written communication, November 2015). Research on harbor seals from 2004-2009 focused on assessing: 1) baseline health and disease status of harbor seals (Hueffer et al. 2011, 2013); 2) the effects of vessel disturbance (Young et al. 2014); 3) harbor seal diet and activity budgets (Herreman et al. 2009, Blundell et al. 2011); 4) harbor seals genetics (Herreman et al. 2009); 5) the effect of Steller sea lion predation on harbor seals (Mathews and Adkison 2010); 6) the post-breeding season migrations of harbor seals (Womble and Gende 2013b); and 7) the foraging ecology and diving behavior of harbor seals in relation to prey availability in glacial ice and terrestrial sites (Womble et al. 2014).

Womble and Gende (2013b) used satellite-linked transmitters to track the post-breeding season movements of female harbor seals, to provide insight into the amount of time seals spend inside and outside the protected area of Glacier Bay. Thirty-seven female seals were captured and fitted with transmitters during September of 2007 and 2008 (Photo 31).

Photo 31. Harbor seal in Johns Hopkins Inlet with a satellite-linked transmitter attached (NPS photo by Jamie Womble).

4.11.5. Current Condition and Trend

Annual Estimate of Population Trend at Glacial Ice Sites during the Pupping and Molting Periods
The majority of harbor seals in Glacier Bay occur at the primary glacial ice sites in Johns Hopkins Inlet (Mathews and Pendleton 2006, Womble et al. 2010). These sites are thought to be particularly important during pupping season, as they may provide a refuge from predation for young seals and a stable platform for resting and nursing that is not subject to tidal inundation (Calambokidis et al. 1987, Mathews and Pendleton 2006, Blundell et al. 2011). Seals also use glacial ice as a resting habitat in Tarr Inlet and McBride Inlet. Historically, harbor seals used glacial ice haul out sites in Muir Inlet as well, but the retreat and grounding of Muir Glacier resulted in the cessation of iceberg calving and seals largely abandoned upper Muir Inlet (Womble et al. 2010). From 1975-1978, June seal populations in Johns Hopkins Inlet increased at an estimated 30.7% per year (95% C.I. 24.3-37%) (Mathews and Pendleton 1997, from Streveler 1979 data). When available data through 1996 are incorporated, the longer-term trend (1975-1996) for June was a lower increase of 3.9% per year (95% C.I. 2.4-5.3%) (Mathews and Pendleton 1997). August seal counts in Johns Hopkins Inlet from 1983-1996 increased at 2.6% (95% C.I. 1.2-4.1%). It should be noted that early surveys (pre-1992) were conducted using different methods than more recent surveys and, as a result, trend estimates are not directly comparable (Womble, written communication, November 2015).

Population declines in Johns Hopkins Inlet were first documented by Mathews and Pendleton (2006) (Table 32). In 1992, 6,200 harbor seals had been counted at glacial and terrestrial haul out sites in

Glacier Bay; in 2002, only 2,250 seals were documented at the same sites (Mathews and Pendleton 2006). June non-pup counts declined by an average of 6.6% per year from 1992-1999, and August counts by 9.6% per year from 1992-2002. The number of pups during June counts remained stable from 1994-1999 (Mathews and Pendleton 2006). These declines continued through the 2000s. When Womble et al. (2010) added data through 2008 to Mathews and Pendleton (2006)'s numbers, the population trend for Johns Hopkins Inlet during June counts indicated a decrease of 7.7% per year (1994-2008), and August populations declined at 8.2% per year (1992-2008) (Table 32). June pup counts showed a declining trend for the first time, at a rate of 5.5% per year from 1994-2008 (Womble et al. 2010). Based on preliminary estimates from aerial surveys of Johns Hopkins Inlet, the population appears to have stabilized between 2007 and 2010 (Womble and Gende 2013a).

Table 32. Annual population trend estimates (with 95% confidence intervals) for the Johns Hopkins Inlet glacial haul out site over various time periods in June (pupping) and August (molting) (Mathews and Pendleton 2006, Womble et al. 2010). Only time periods during which survey methods were comparable are included here.

	June surveys		August surveys	
Location	**1992-1999**	**1994-2008**	**1992-2002**	**1992-2008**
Johns Hopkins Inlet	-6.6% (-8.5, -4.7)	-7.7% (-8.4, -7.1)	-9.6% (-10.3, -8.8)	-8.2% (-8.5, -7.8)

<u>Annual Estimate of Population Trend at Terrestrial Sites during the Pupping and Molting Periods</u>

While harbor seals have utilized terrestrial haul out sites throughout Glacier Bay, the largest aggregations currently occur in the Spider Reef Complex, at Flapjack Island, Boulder Island, and in Adams Inlet (Womble and Gende 2013a). Population trends for harbor seals at terrestrial sites were first reported by Mathews and Pendleton (1997, 2006). Estimates showed a declining trend, with an average annual rate of -14.5% for the period 1992-2001, which is equivalent to a decline of -75% over 10 years (Table 33; Mathews and Pendleton 2006). These rates of decline were nearly the highest ever reported in Alaska (Mathews and Pendleton 2006). When data through 2008 were incorporated, the rate of population decline was a slightly lower -12.4% per year (1992-2008; Womble et al. 2010). From 2007-2011, counts at terrestrial sites appeared to have stabilized, with a positive trend over the 5-year period of 3.5% per year (Table 33; Womble and Gende 2013a). The trend for 2009-2013 increased further to 13.25% (Womble et al. 2015). When incorporated with previous data, the long-term trend (1992-2013) is still negative, at a declining rate of -6.9% per year (Womble et al. 2015). Recent data suggest that the decline at terrestrial sites is lessening, and there have been increasing numbers of harbor seals at Spider Reef and Flapjack Island in recent years (Womble and Gende 2013a).

Table 33. Annual population trend estimates (with 95% confidence intervals) in the number of harbor seals counted at terrestrial sites during the molting period in late July and August (Mathews and Pendleton 2006, Womble et al. 2010, Womble et al. 2015).

Location	1992-2002	1992-2008	1992-2013	2007-2013	2009-2013
Terrestrial sites	-14.5% (-17.1, -11.9)	-12.4% (-13.7, -11.1)	-6.9% (-7.7, -6.1)	4.4% (0.9, 7.9)	13.3% (4.7, 21.8)

Proportion of Pups during the June Pupping Period

From 1994-1999, Mathews and Pendleton (2006) observed mid-June pup proportions in Johns Hopkins Inlet ranging from 34-36%. Womble et al. (2010) reported that pup *proportions* remained stable in Johns Hopkins Inlet through 2008 at 34-37%, despite a 5.5% decline in pup *numbers* during that time. The proportion of pups calculated from new aerial survey methods in Johns Hopkins Inlet averaged 31% from 2007-2010, with proportions ranging from 21-37% (Womble and Gende 2013a). These results are preliminary, as analytical tools for estimating the abundance of seals at glacial sites are still in development and may not be comparable to previous land-based surveys (Womble and Gende 2013a).

Threats and Stressor Factors

The harbor seals in Glacier Bay are one of the most protected marine mammal populations in the world (Womble and Gende 2013b), yet the population has still declined. Potential anthropogenic threats and natural stressors that may influence harbor seals include legal and illegal harvest, human disturbance, interactions with commercial fisheries, pathogens and pollutants, predation, interspecies competition, and climate change.

Human Activities

Subsistence harvest of harbor seals by Alaska Natives is authorized under the Marine Mammal Protection Act of 1972; however, subsistence harvest of harbor seals has not been permitted in the park portion of GLBA since 1973 (Catton 1995). Harbor seals are an important cultural and subsistence resource for Alaska Natives, particularly in southeastern Alaska, and harvest has occurred for many generations (de Laguna 1972, Emmons 1991). Harvested seals are used for meat, oil, skins, and handicrafts and are an important item for trading and cultural exchange (Wolfe et al. 2009). There are typically two distinct seasonal peaks for subsistence harvest of seals, one during spring and another in fall/early winter (de Laguna 1972, Wolfe et al. 2009). These peaks coincide with seasons when seals tend to travel beyond the boundaries of GLBA (Womble and Gende 2013b). Four native communities within the documented range of GLBA harbor seals (Womble and Gende 2013b) took a total of 195 harbor seals in 2008 (Figure 83, Table 34; Wolfe et al. 2009). However, current harvest levels are much lower than in the early 1990s, when the estimated number of seals taken by the Hoonah community alone was over 350 annually (Mathews and Pendleton 2006). It is currently unknown whether or not subsistence harvest may have population-level effects on harbor seals in Glacier Bay (Womble and Gende 2013b).

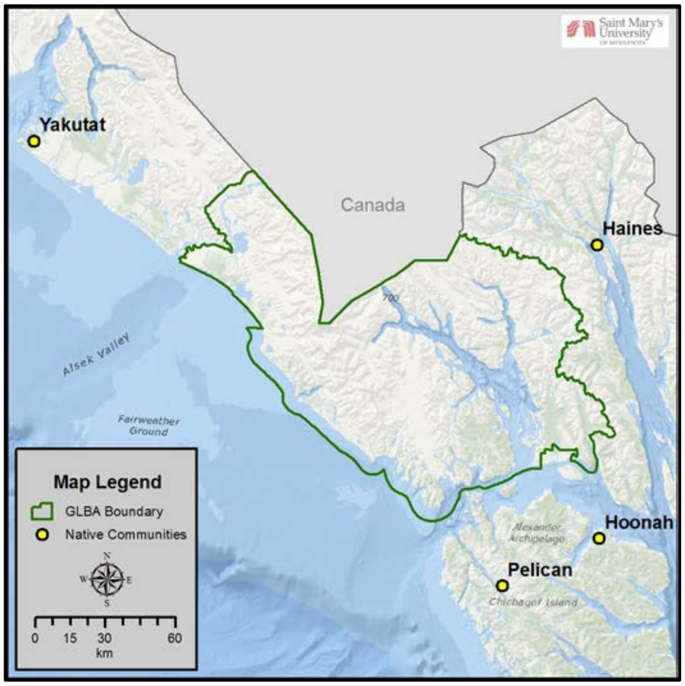

Figure 83. Native communities in the vicinity of GLBA with reported subsistence harvests of harbor seals.

Table 34. Estimated subsistence harbor seal take by four Alaska Native communities in the GLBA region, 2008 (Wolfe et al. 2009; locations shown in Figure 83).

Community	Harvest[A]	Struck and Lost[B]	Total Take[C] (95% C.I.)
Hoonah	25.3	6.3	31.6 (30.0, 39.0)
Pelican	10.0	0	10.0 (8.2, 11.8)
Haines	15.6	2.6	18.2 (14.0, 35.3)
Yakutat	115.2	20.0	135.1 (115.0, 170.8)

[A] Harvest numbers represent animals successfully collected by the community.

[B] Struck and Lost numbers represent animals that were not successfully collected by the community.

[C] Total Take = Harvest + Struck and Lost.

Human activities (e.g., fishing, boating, wildlife viewing) are a threat to the GLBA seal population, both within the park and outside park boundaries. Harbor seals may interact with commercial and subsistence fisheries and may become injured or entangled in fishing gear. Commercial fishing is limited within GLBA, so the risk of conflict is low in the park, but many seals travel extensively outside the park during the non-breeding season (Womble and Gende 2013b). Commercial and subsistence gillnet fisheries for salmon occur in several areas in Southeast Alaska, including Lynn Canal, Yakutat Bay, along the coast of the Yakutat Forelands in the eastern Gulf of Alaska, and in the Taku Inlet-Stephens Passage area (Womble, written communication, November 2015). Many of these areas are also used by harbor seals from Glacier Bay during the post-breeding season (Womble and Gende 2013b); however, the extent to which harbor seals interact with gillnet fisheries in southeastern Alaska is largely unknown. Between 2007 and 2011, nine cases of human-related harbor seal injury or mortality were reported in Southeast Alaska, including gunshots, a ship strike, and fishing gear entanglement/ingestion (Allen et al. 2014).

Vessel traffic in and outside the park may influence harbor seal behavior, potentially influencing the condition of individual seals (Lewis and Mathews 2000, Young et al. 2014). Vessel disturbance can cause hauled-out seals to flush into the water, which may result in increased energy expenditure or increase the risk of mother/pup separation (Lewis and Mathews 2000, Young et al. 2014). The park has established several closure areas and approach distance restrictions to prevent vessel and human-related disturbance of seals (Figure 84). For example, Johns Hopkins Inlet is closed to all vessels from 1 May-30 June and to cruise ships until after 31 August (36 CFR 13.1178) (Young et al. 2014). Vessels are also required to maintain a distance of 0.25 nautical miles from hauled-out seals “except when safe navigation requires otherwise” (36 CFR 13.1178) (Young et al. 2014). However, a recent study suggests that compliance with these regulations may be low. In Johns Hopkins Inlet during the summers of 2007 and 2008, only 16% of observed tour vessels and 33% of cruise ships maintained the minimum distance of 0.25 nautical miles (Young et al. 2014). During 2007-2008 observations, 86% of cruise ships, 79% of tour vessels, and 47% of kayak groups entering Johns Hopkins Inlet flushed at least one seal into the water (Young et al. 2014).

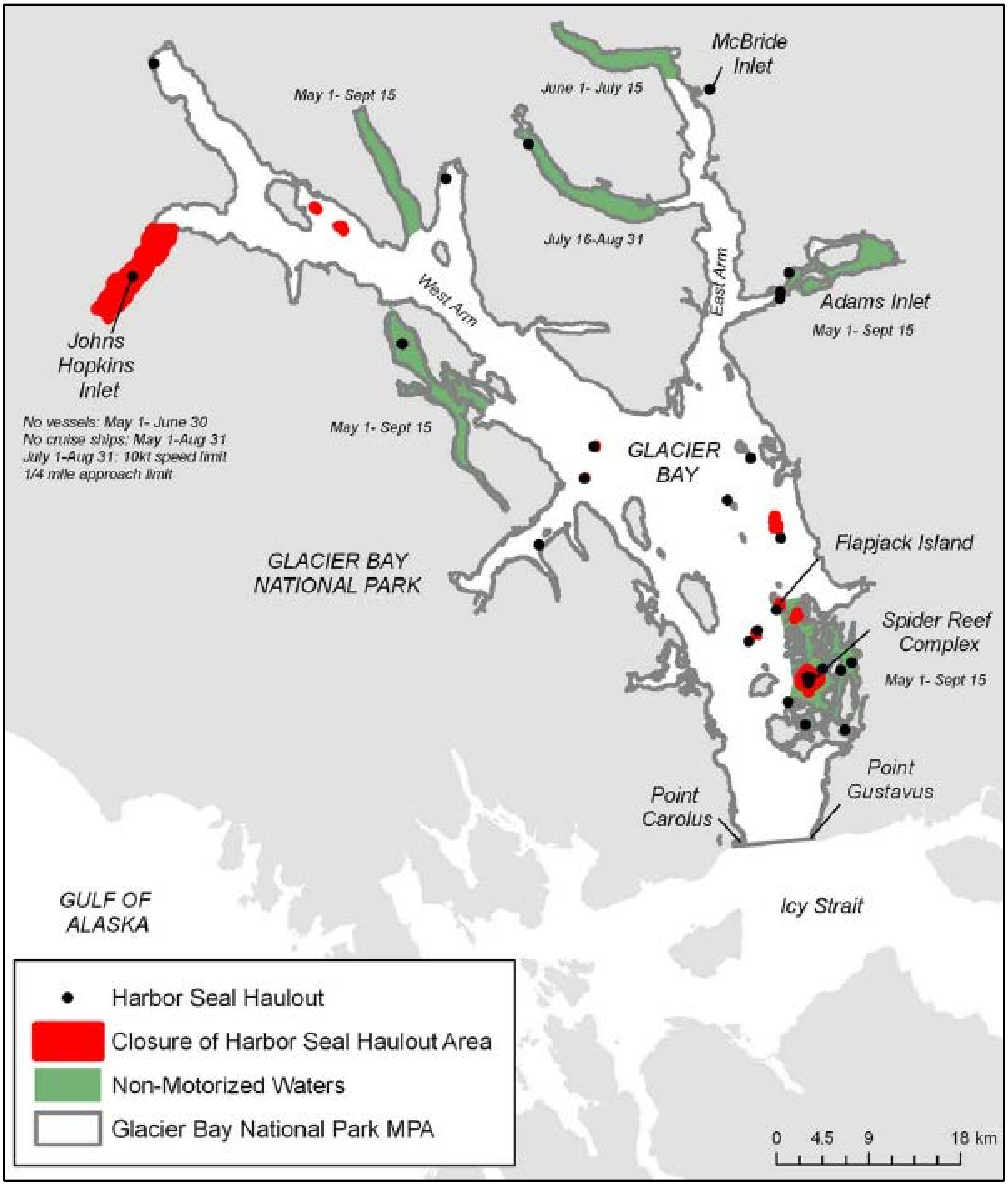

Figure 84. Harbor seal haul out sites and associated closure areas (with dates), as well as additional restricted areas (e.g., non-motorized waters) (Womble and Gende 2013b).

Health Status

Several pollutants and pathogens threaten the health of harbor seals throughout Alaska, including in the Glacier Bay region. The GLBA harbor seal population has likely been exposed to several pathogens, including *Brucella* and *Leptospira* bacteria, which may affect the health of the population but are likely not a major cause of the observed population decline (Hueffer et al. 2011, 2013). Phocine distemper virus (PDV), which has caused epidemics among North Atlantic harbor seals, has been detected in sea otters in the North Pacific and could threaten the GLBA population (Goldstein et al. 2009). Serologic tests show that GLBA harbor seals have not been previously exposed to distemper viruses and could be vulnerable to severe mortality if a virus such as PDV is introduced (Goldstein et al. 2009, Hueffer et al. 2011). Although harbor seals in GLBA have not been tested for exposure to pollutants, organochlorines such as PCBs have been documented in the northern Gulf of the Alaska seal population (Wang et al. 2007) and in Glacier Bay mussels (Tallmon 2012). In pinnipeds, elevated organochlorine levels can adversely affect the reproductive and immune systems (Reijnders 1986, de Swart et al. 1996, Wang et al. 2007).

Changes in the Availability of Glacier Ice for Seal Habitat

The loss of glacial ice, through both recession and thinning, is a potential threat to harbor seals' glacial haul out sites. At these glacial sites, seals rely on icebergs from tidewater glaciers for calving, molting, resting, and protection from predators (Calambokidis et al. 1987, Mathews and Pendleton 2006, Womble et al. 2010). Glacier Bay's tidewater glaciers have been receding for at least several decades (Womble et al. 2010, Loso et al. 2014). Muir Glacier, for example, retreated over 7 km (4.3 mi) between 1973 and 1986; the glacier "grounded" and stopped producing icebergs in 1993 (Hall et al. 1995, Womble et al. 2010). Muir Inlet supported approximately 1,300 harbor seals during the 1970s (Streveler 1979) but was abandoned by seals after the glacier grounded (Mathews and Pendleton 1997, Womble et al. 2010). In the late 1970s and early 1980s, 12 glaciers were calving icebergs into Glacier Bay (Molnia 2007, Womble et al. 2010). By 2008, only five glaciers were actively calving into the bay (Womble et al. 2010). Alaska's tidewater glaciers are predicted to continue thinning and/or receding, largely due to global climate change (Larsen et al. 2007, Womble et al. 2010). The loss of glacial haul out sites in GLBA may cause harbor seals to use terrestrial sites for breeding (increasing exposure to land-based predators) or to seek glacial sites in other areas outside of Glacier Bay (Womble et al. 2010, Allen et al. 2011).

Natural Stressors/Mortality Sources

Orcas and Steller sea lions are known to prey upon harbor seals in Glacier Bay (Calambokidis et al. 1987, Matkin et al. 2007, Womble and Conlon 2010). Harbor seals were the prey in 40% of documented transient killer whale "kill incidents" in GLBA from 1986-2003 (Matkin et al. 2007). Killer whales are more common in the lower and central parts of the bay and have not yet been sighted in areas as far north as seal glacial haulout sites (Gabriele et al. 2011). While seals are not a substantial component of Steller sea lion diet in the area, individuals may target young seals due to their small size and naïve behavior (Womble and Conlon 2010). Pacific sleeper sharks (*Somniosus pacificus*) may also prey upon harbor seals; seal tissue was found in the stomach of a sleeper shark caught in Muir Inlet (Taggart et al. 2005). The risk of predation may drive seals to forage in areas with lower-quality prey items, which could influence their health and body condition (Herreman et al. 2009).

Interspecific competition is also a natural process that may be contributing to the population's decline (Mathews and Pendleton 2006). In Glacier Bay, harbor seal foraging habitat and prey species are similar to those of Steller sea lions and humpback whales (Mathews and Pendleton 2006, Womble et al. 2010). Shared prey items include small fish such as Pacific capelin, walleye pollock (*Theragra chalcogramma*), Pacific sand lance, and Pacific herring (Womble et al. 2010). Steller sea lion and humpback whale numbers have increased significantly in Glacier Bay in recent decades (Gabriele et al. 2011, Mathews et al. 2011, Saracco et al. 2013), potentially creating more competition for harbor seals.

Data Needs/Gaps

The drivers behind the Glacier Bay harbor seal decline are not fully understood, although some causes have been largely ruled out (e.g., harvest, commercial fisheries conflicts). Research into GLBA seal reproductive rates, survival rates, and the extent to which emigration may occur would provide a better understanding of harbor seal population ecology in Glacier Bay.

Overall Condition

Annual Estimate of Population Trend at Glacial Ice Sites

The project team assigned this measure a *Significance Level* of 3. Since 1992, harbor seals have shown a declining population trend in Johns Hopkins Inlet, the primary glacial haulout site in GLBA (Mathews and Pendleton 2006, Womble et al. 2010). Recent surveys suggest that the population trend may be stabilizing (Womble and Gende 2013a), but this is still a major concern for park managers. Therefore, this measure is assigned a *Condition Level* of 3.

Annual Estimate of Population Trend at Terrestrial Sites

This measure was also assigned a *Significance Level* of 3. Population trends at Glacier Bay terrestrial haulout sites have also been declining since 1992, but at an even higher rate than glacial sites (Mathews and Pendleton 2006, Womble et al. 2010). In recent years, the trend appears to have stabilized and even increased, resulting in a lessening of the long-term rate of decline (Womble and Gende 2013a, Womble et al. 2015). However, the long-term decline and the uncertainty behind its cause is still of moderate concern (*Condition Level* = 2).

Proportion of Pups during the June Pupping Period

A *Significance Level* of 2 was assigned to this measure. Pup proportions have only been reported once for Glacier Bay terrestrial sites (10% from 1982-1984; Calambokidis et al. 1987). Pup proportions at glacial ice sites in GLBA are relatively high (34-40%) and have remained fairly stable since harbor seal surveys began. As a result, this measure is of low concern (*Condition Level* = 1).

Weighted Condition Score

The *Weighted Condition Score* for GLBA's harbor seals is 0.71, meaning the resource is of significant concern. Since population estimates appear to have stabilized in recent years and pup proportions are stable, the current trend is considered unchanging/stable.

Harbor Seals			
Measures	Significance Level	Condition Level	WCS = 0.71
Population Trend at Ice Sites	3	3	
Population Trend at Terrestrial Sites	3	2	
Pup Proportion	2	1	

4.11.6. Sources of Expertise

- Jamie Womble, NPS Wildlife Biologist

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4.12. Humpback Whales

4.12.1. Description

Humpback whales are a large endangered baleen whale species characterized by their long pectoral fins (Clapham and Mead 1999; Photo 32). Adults can weigh up to 36,000 kg (35 tons) and have mean lengths from 13-15 m (43-49 ft) (Chittleborough 1965, Mikhalev 1997; as cited in Fleming and Jackson 2011). Similar to other baleen whale species, female humpbacks are larger than males (Clapham and Mead 1999, NOAA OPR 2014). Male humpback whales reach lengths up to 14.3 m (~46 ft), while females have been known to be as long as 15.5 m (~50 ft) (Chittleborough 1965). Humpback whales are mainly dark grey or black; however, there may be white coloration in some areas (NOAA OPR 2014).

Photo 32. Breaching humpback whale (NPS photo).

Humpback whales are a migratory species whose habitat preferences change depending on the activity and time of year. During the summer months, the whales feed in cold coastal waters at higher latitudes. Warm shallow waters (e.g., reef systems, islands, coastal shores) closer to the Equator are preferred for calving grounds in the winter (NOAA OPR 2014). Humpback whales travel great distances every year between their summer and winter grounds. During the summer months, Alaska's whales inhabit the Gulf of Alaska, and Southeast Alaska humpbacks tend to migrate to the Hawaiian Islands during the winter months to their calving grounds (Baker et al. 1986). Whales travel approximately 10,000 km (6,214 mi) (round trip) each year.

Mating activities occur in the subtropical wintering grounds. Humpback whales are considered polygynous breeders, meaning males may mate with more than one female. With most polygynous species, the male does little to protect and nurture the offspring, leaving the female to provide most, if not all, of the parental support. The gestation period is approximately 11 months (NOAA OPR

2014). Historically, it was believed that humpback whales reach sexual maturity at five years of age (Clapham and Mead 1999). However, in more recent studies, female whales in southeastern Alaska have been at least 8 years old before their first calving with a mean age at first calving of 11.8 years (Gabriele et al. 2007). Breeding females generally give birth every 2 years; however, some North Pacific females have given birth annually for two or three consecutive summers (Straley et al. 1994).

Humpback whales consume large amounts of food during the summer months and live off of fat reserves during the winter months (Krieger 1988). Humpbacks were first reported in GLBA in 1899 as part of the Harriman Expedition, but were not commonly reported in the park until the 1950s (NPS ranger logs, NPS 2014). Individually identified whales have been documented using GLBA as a feeding ground since the 1970s (Gabriele, written communication, 7 August 2014). Individuals have strong maternally-directed fidelity to feeding grounds (Baker et al. 2013). According to Krieger (1988), humpback whales feed on schools of fish, such as capelin, walleye pollock, and Pacific herring as well as several genera of euphausiids (e.g., *Euphausia*, *Thysanoessa*, and *Meganyctiphanes*). Whale distribution in GLBA changes dramatically depending on the distribution of available schools of prey (Krieger 1988, Neilson et al. 2014).

Humpback whales have been identified as a Vital Sign within SEAN. The NPS has been monitoring humpback whales in the park and surrounding waters since 1985 (Neilson et al. 2014). There are four distinct population segments (DPS) in the North Pacific, which are designated by breeding ground locations. The four segments are Hawaii, Mexico, Central America, and West Pacific/Asia (Fleming and Jackson 2011). Whales from the first two DPS may summer in Alaskan waters. According to NOAA OPR (2014), the North Pacific stock populations once declined to an estimated 1,400 whales due to commercial whaling; however, in recent years the population has been estimated at 20,000 whales. Humpback whales were listed under the Endangered Species Act and classified by the IUCN as endangered in 1970 and 1986, respectively (Clapham and Mead 1999, IUCN 2014, USFWS 2014). In 1990, humpback whales were moved to the Vulnerable Category on the IUCN Red List of Threatened Species and then downgraded to “Least Concern” in 2008, due to their increasing population trend (IUCN 2014). The NMFS has proposed removing the endangered classification for the species as a whole, and instead classifying two population segments (Northwest Africa and Arabian Sea DPS) as endangered and two as threatened (Western North Pacific and Central America DPS) (NOAA 2015).

4.12.2. Measures

- Distribution
- Population estimate
- Crude birth rate

4.12.3. Reference Conditions/Values

The reference condition for population estimates is the rate of population increase from 1985-2012, which was 4.4%. Crude birth rates are naturally highly variable; therefore, the reference condition for crude birth rate is the median (8.7) and interquartile range (6.7-12.1) for the period 1984-2014 (Gabriele, written communication, May 2015). Crude birth rates are determined by dividing the number of births in a year by the population estimate, then taking the quotient and multiplying by 100. The 1985 distribution of humpback whales will serve as the reference condition for distribution.

4.12.4. Data and Methods

Baker (1985) conducted a study on humpback whales in GLBA and surrounding waters in 1985.This study was the beginning of a continuous monitoring program to monitor the humpback whale population in GLBA and surrounding waters. The report focused on Glacier Bay and Icy Strait. Thirty-one vessel surveys were performed from June to August 1985. A total of 234 survey hours were completed, with each vessel survey lasting approximately 7.5 hours on average. Photo identification and prey assessment were also documented. Population dynamics parameters that were assessed included abundance, seasonal influx, local movement and residency, regional movement, distribution, prey assessment and feeding behavior, and social behavior.

NPS staff continued the ongoing monitoring program by conducted monitoring studies from 1985 to the present (Neilson et al. 2013, 2014, 2015). These studies are the most current efforts in the ongoing monitoring program, which was initiated by Baker (1985) in 1984. These reports summarize over 30 years of data. Methods used in this survey were similar if not the same as the methods used in the Baker (1985) report. Population dynamics such as distribution, whale counts, and crude birth rates were also included in this report.

Straley et al. (2009) conducted a mark and recapture survey that estimated the abundance of humpback whales in southeastern Alaska between 1994 and 2000. The study was conducted to determine the best mark-recapture model for estimating abundance. Minimum counts, Peterson estimates, Darroch stratified estimates, and Hilborn open, multi-strata approach were analyzed for accuracy.

Hendrix et al. (2012) conducted a Bayesian estimation study of humpback whale population abundance and movement patterns in Southeast Alaska. Whale survival, abundance, and rate of increase were calculated using a mechanistic movement model. Figure 85 displays the study area (Southeast Alaska), which includes GLBA.

Figure 85. Study area (Southeast Alaska) used in Bayesian estimation study (Hendrix et al. 2012).

Saracco et al. (2013) conducted a study on population dynamics and demography of humpback whales in GLBA and Icy Strait. Humpback whale data from 1985-2009 were acquired and used in a capture-recapture model to calculate population size, growth rate, survival, and temporary emigration rate.

4.12.5. Current Condition and Trend

Distribution

Baker (1985) documented whale distribution in GLBA from June through August in 1985. In June, whales were observed from the Muir Inlet to Sebree Island to Garforth Island, and south to Point George. In July, whales were more commonly observed in the West Arm, more specifically between Lone Island and Blue Mouse Cove. Whale distribution shifted from the lower and mid-bay in August; sightings were more common in Flapjack Island, Willoughby Island, Young Island and Lester Island (Baker 1985). Figure 86 displays the whale observations recorded in GLBA in 1985.

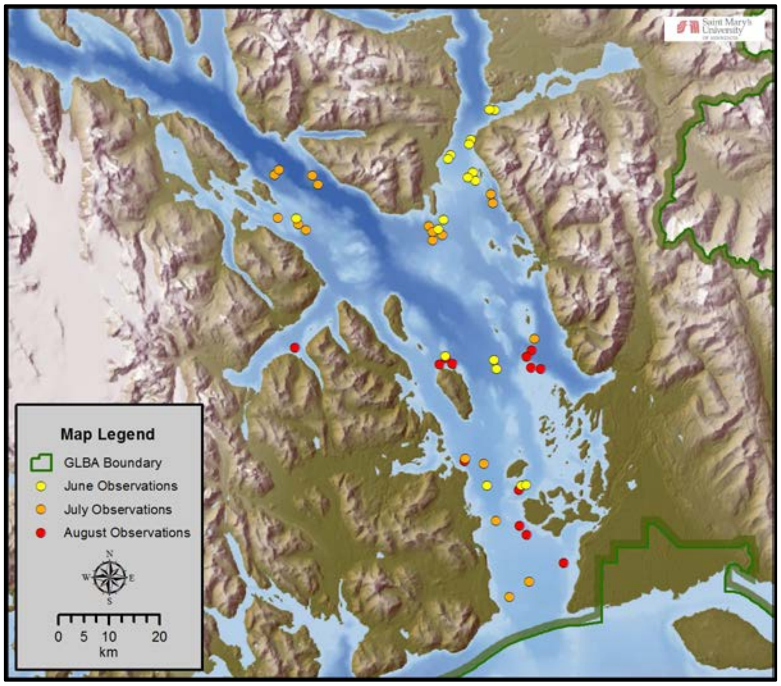

Figure 86. Whale observations recorded in GLBA in 1985 (Baker 1985).

Neilson et al. (2013) documented whale sightings in GLBA in 2012. Some sightings occurred in the West Arm and Muir Inlet, but a majority of whale sightings occur from the entrance of the bay to the main channel. Neilson et al. (2013) states that whales have been known to use the mid-channel as well as nearshore areas. Neilson et al. (2014) documented whale sightings in GLBA in 2013. Whale observations in 2013 were more widely distributed than in 2012. More whale sighting were observed near the entrance of the bay and main channel as well as near Bartlett Cove, Willoughby Island, Sandy Cove, Sturgess Island, and in Geikie Inlet. Whales were also sighted as far north as Adams Inlet and Russell Island. Figure 87 displays the yearly distribution of humpback whales recorded from 1993 to 2001, 2002 to 2007, and 2008 to 2013.

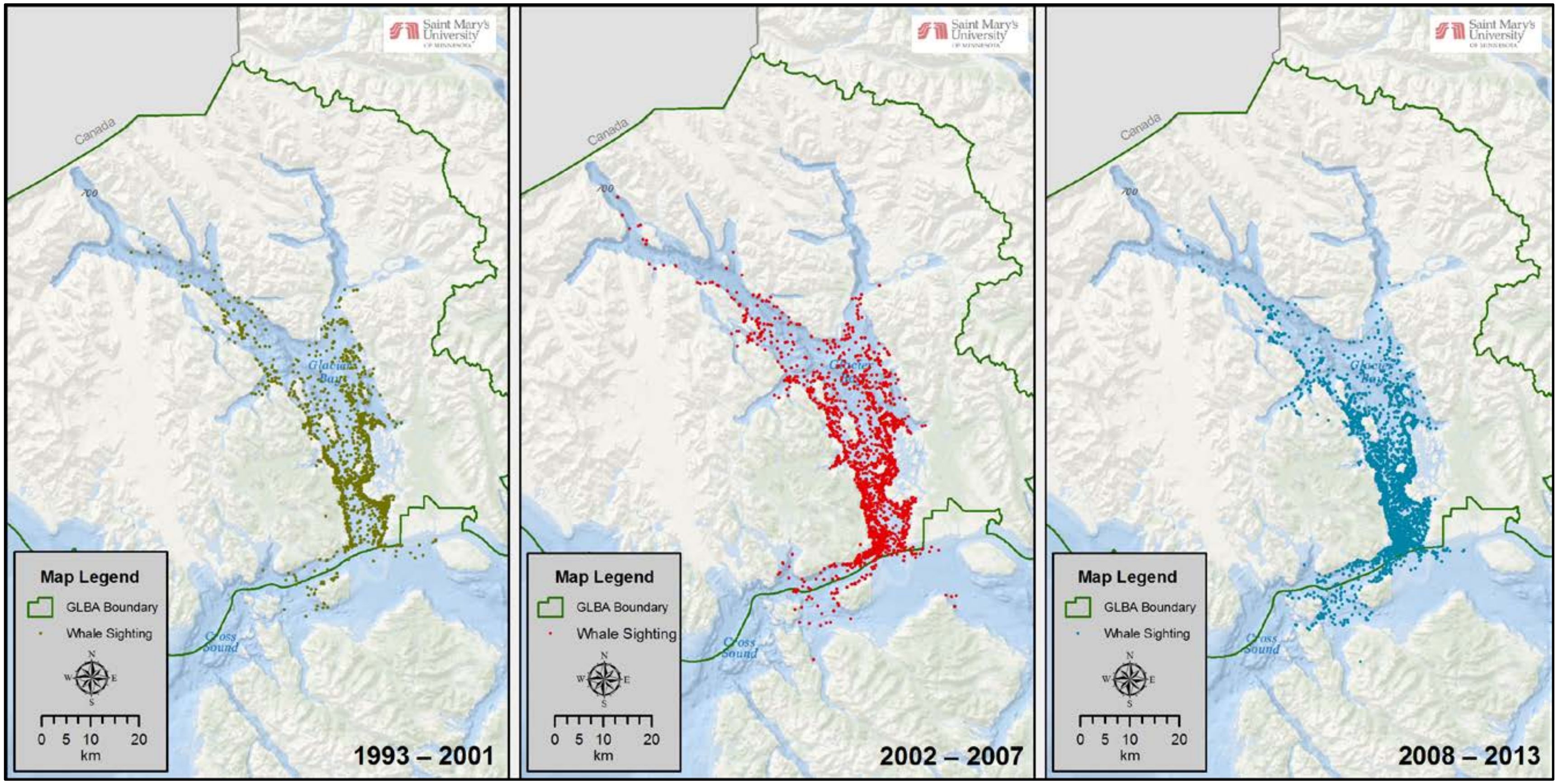

Figure 87. Humpback whale sightings in GLBA from 1993 to 2001 (left), 2002 to 2007 (middle), and 2008 to 2013 (right) (NPS 2013).

Population Estimate

Whale population estimates vary depending on the estimation method used. Straley et al. (2009) utilized and compared several of these methods. The study documented a minimum count of 842 whales in Southeast Alaska between 1994 and 2000 (Table 35). It was concluded that the best population estimate for 2000 was obtained using the Hilborn approach model; the estimated population for humpback whales in Southeast Alaska using this method was 961 whales.

Table 35. Humpback whale population estimates from various studies in southeastern Alaska (Straley 1994, Straley et al. 2009, Hendrix et al. 2012, Saracco et al. 2013).*

Area/Survey	Year	Estimate
Glacier Bay/Icy Strait		
Hendrix et al. 2012	1995	75
	2008	550
Saracco et al. 2013	1986	50
	2009	181
Southeast Alaska		
Straley 1994	1986	393
Straley et al. 2009	2000	961
Hendrix et al. 2012	2008	1,585

* Variation between survey estimates in similar years may be due to differences in the models used to calculate population estimates (e.g., mechanistic movement model used by Hendrix et al. [2012] vs. closed robust design used by Saracco et al. [2013]).

Hendrix et al. (2012) documented population abundance estimates in GLBA and Icy Strait between 1995 and 2008. Population estimates fluctuated slightly but increased throughout the study period. In 1995, the population estimate for GLBA and Icy Strait was approximately 75 whales. By 2008, the authors estimated that the population had grown to approximately 550 whales. The median abundance in 2008 for humpback whales throughout Southeast Alaska was 1,585 whales (Hendrix et al. 2012). A median growth rate of 5.84% was recorded for the humpback whale population in Southeast Alaska between 1995 and 2008.

Saracco et al. (2013) calculated population estimates in GLBA and Icy Strait between 1985 and 2009. Population estimates have fluctuated over the years, but seem to be increasing and range from approximately 49 whales (1986) to 181 whales (2009). An annual rate of increase of 4.4% was documented for the whale population in GLBA between 1985 and 2009; however, this study documented an increase of approximately 7.7% between 2002 and 2009.

Neilson et al. (2013, 2014, 2015) documented total whale counts in GLBA between 1985 and 2013. The overall trend in whale counts has increased over the 30-year monitoring period, but has fluctuated. The whale counts between 2010 and 2013 are the highest records throughout the monitoring period (Neilson et al. 2015). The highest whale count occurred in 2013 with a total of 160 whales observed. Observations decreased slightly in 2014 to 98 whales (Neilson et al. 2015). Figure 88 displays the total whale count in GLBA between 1985 and 2014.

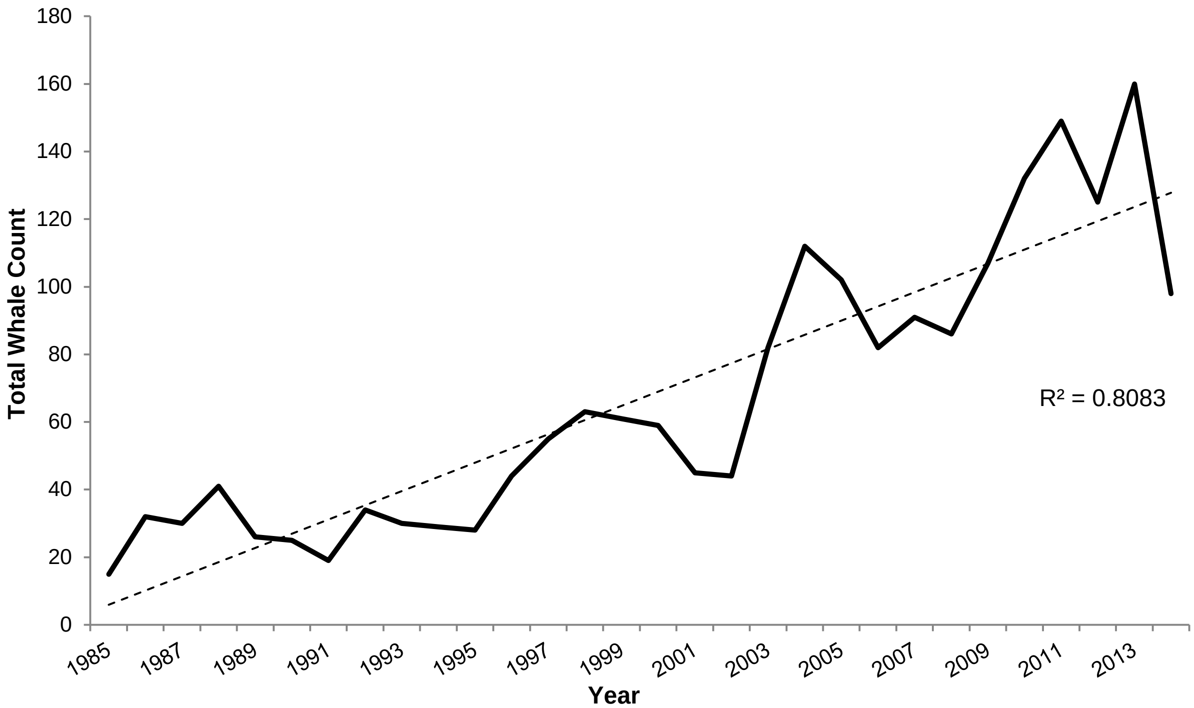

Figure 88. Total humpback whale counts in GLBA between 1985 and 2014. A linear trend line was added to show the long-term trend (Neilson et al. 2015) (R^2 is considered meaningful if >0.5).

<u>Crude Birth Rate</u>

Neilson et al. (2013, 2014, 2015) documented 30 years of crude birth rates of humpback whale residents in GLBA. Crude birth rates varied between 1984-2014, with a peak value of 18.5% (1992) and a low value of 3.4% (2003). The crude birth rate averaged 9.4% over 30 years, which is within the reference condition (6.7-12.1%) (Nielson et al. 2015). Over the last 30 years, 24 years showed a crude birth rate within or above the reference condition range. In the last 10 years, seven years were within or above the reference condition (8.8%). The lower crude birth rate in recent years may mean the population is reaching carrying capacity or that the age structure of the population has changed (Gabriele, written communication, August 7 2014). Figure 89 displays the crude birth rates of humpback whales in GLBA from 1984 to 2014 in comparison to the reference condition range set as a part of the NRCA process.

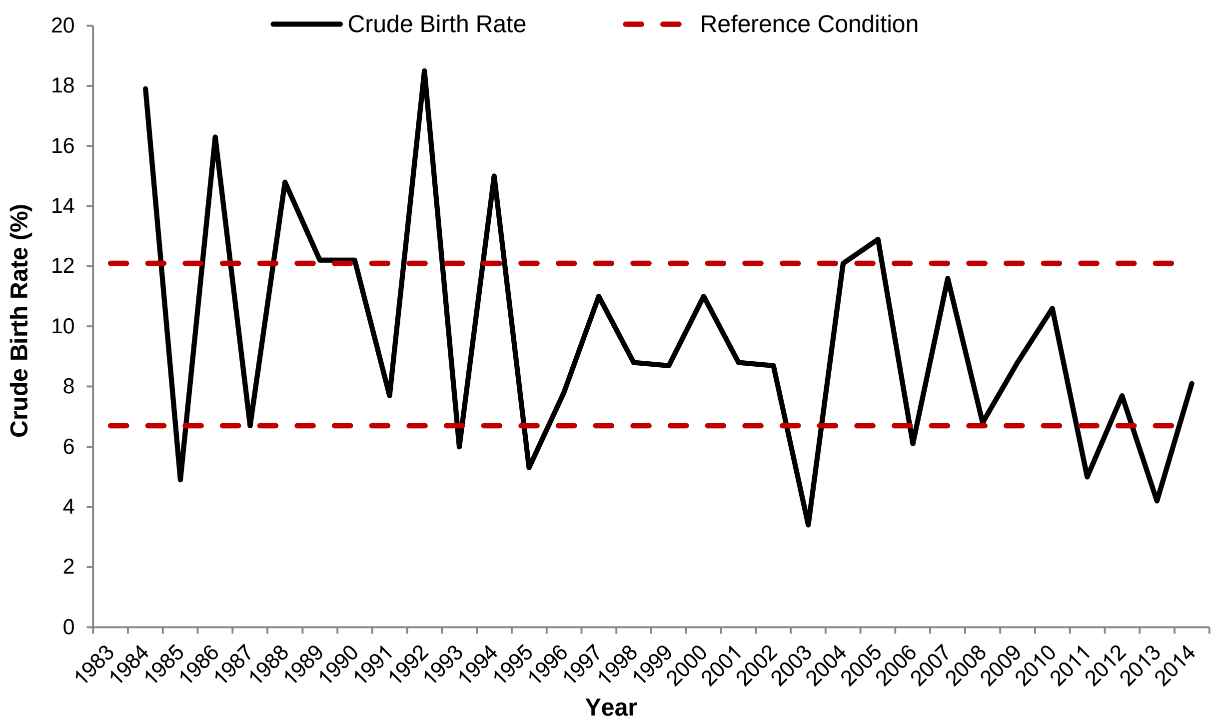

Figure 89. Crude birth rates of humpback whales in GLBA between 1984 and 2014 (Neilson et al. 2015).

Threats and Stressor Factors

GLBA staff identified several potential threats and stressors to the humpback whales in and around the park. Most of the threats are anthropogenic, while others may be naturally occurring. Anthropogenic threats include disturbance from underwater vessel noise, vessel strikes (lethal and non-lethal), entanglement in fishing gear, and competition with commercial fisheries. Natural threats are climate changes and ocean acidification which are likely to cause unanticipated changes in prey resources and habitat suitability.

Disturbance from underwater vessel noise can cause stress to humpback whales in GLBA, which may influence whale distribution in the park. Humpback whales are adapted to rely on sound for everyday activities such as feeding, intraspecific location, intraspecific communication, and detection of predators (Gabriele et al. 2010, Holt et al. 2011); vessel traffic noise interferes with that. A majority of those vessels (e.g., cruise ships, small motorized vessels) traveling through the bay area are for visitors to view the park; these vessel trips usually come at the same time frames every day throughout the visitor season (summer) and are not likely to decline. The park staff has enforced mandatory speed reductions (from 20 knots to 13 knots) during the summer season to reduce the level of noise and intensity of adverse effects on underwater soundscape (Kipple and Gabriele 2003) as well as reduce the probability of whale disturbance and probability of whale-vessel collisions. If increases in the number of vessel entries per year were allowed, it would result in added stress to the natural underwater soundscape (Gabriele et al. 2010).

Vessel strikes, whether lethal or non-lethal, are another threat to humpback whales in GLBA (Photo 33). Accurate assessments of vessel strikes are challenging due to the difficulty of getting accurate and complete information on a whale's cause of death. The GLBA long-term monitoring program was created because there was concern that increased vessel traffic in GLBA may result in whales abandoning the park (Jurasz and Palmer 1981; as cited by Neilson et al. 2013). Neilson et al. (2012) identified the mouth of Glacier Bay as a "hotspot" with elevated risk of whale-vessel collisions (Figure 90). All types of vessels (motorized and non-motorized) performing various activities have been known to hit humpback whales (Neilson et al. 2012). In 25 of 108 instances of vessel collision between 1978 and 2011, the whale died as a result of the strike. However, cause of death could not be conclusively determined in many cases, so this is likely an underestimate (Gabriele, written communication, May 2015). According to Neilson et al. (2012), small, medium, and large vessels all strike whales, but small vessel strikes are most common in Southeast Alaska. In 2011, there were two recorded non-lethal strikes, with one occurring between Willoughby Island and Strawberry Island, and the other occurring in Bartlett Cove (Neilson et al. 2012) (Figure 90). There were 15 collision cases where whales were reported colliding with drifting and anchored vessels, which may indicate that the whales were unaware of the location of the vessel (Neilson et al. 2012). All vessels, whether anchored (which also represent a human safety threat) or moving, appear to be a threat to whales in Alaska.

Photo 33. Propeller scars on an adult humpback observed during 2014 GLBA whale surveys (NPS photo).

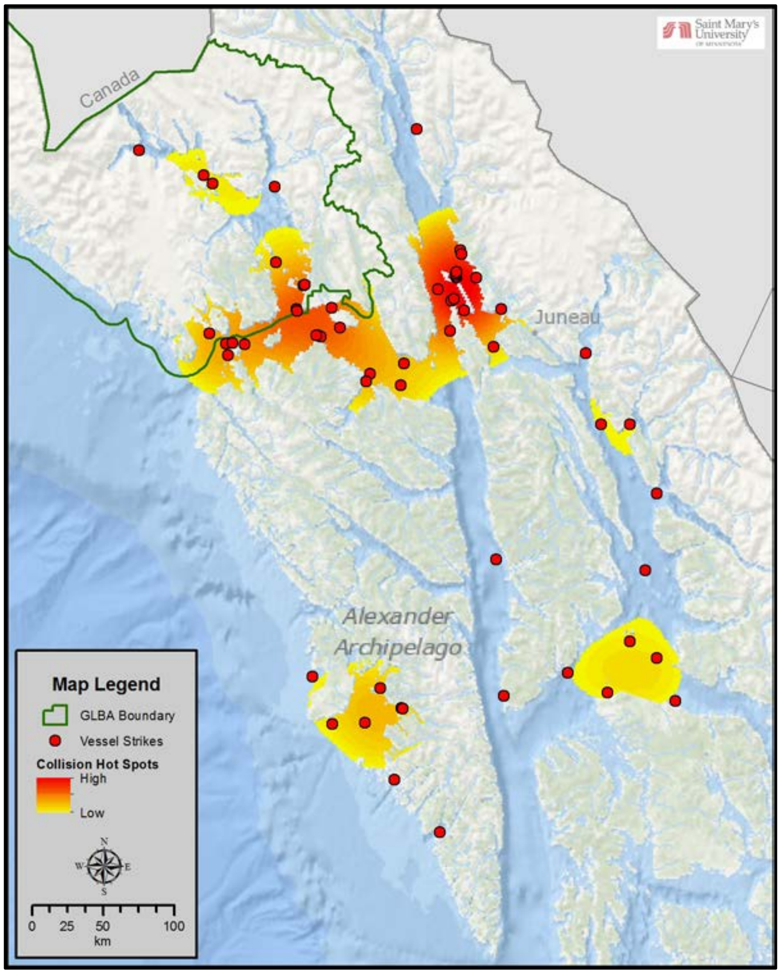

Figure 90. Whale-vessel collision hotspots in southeast Alaska between 1978 and 2011 based on kernel density estimation (Neilson et al. 2012). Yellow indicates low to moderate collision risk and red indicates higher collision risk.

Another threat to humpback whales in GLBA and surrounding waters is entanglement in fishing gear (Photo 34). Adult whales face a higher risk of entanglement than calves. According to Neilson (2006), older whales and male whales are more at risk of entanglement than female whales. In Southeast Alaska, an estimated 2.7 whales per year are seriously injured or killed by incidental entanglement. It should be noted that not all incidents of entanglement are reported. Incidents of entanglement in Alaskan waters are high in the summer months (July, August). There were 52 humpback whale entanglements in fishing lines reported in Alaska between 1997 and 2004.

Photo 34. Whale calf observed during 2014 GLBA surveys with a fishing lure and line entangled on its fluke (NPS photo).

Changes in prey resources (e.g., from natural climate fluctuations, anthropogenic climate change, ocean acidification) are a potential threat to humpback whales in GLBA. Humpback whales consume large amounts of prey each day once in their feeding grounds; estimates range from 370 kg (800 lbs) to 1,360 kg (1.5 tons or nearly 3,000 lbs) per day (Witteveen et al. 2004, Zimmerman and Karpovich 2008). Many humpback whales migrate to the park to feed. Prey availability has been proposed as a limiting factor in humpback whale populations along the west coast of North America (FOC 2013). Whales consume a large variety of prey; however, some prey species are of higher quality than others. Large whales such as humpbacks will target the highest quality prey source first, but will shift to lower quality prey items when primary prey species become low in abundance. If whales are only consuming low quality prey items, the population may experience reductions in growth rate, fat storage, reproduction success, and delayed maturation (FOC 2013). Since fluctuating prey availability is a factor that naturally occurs in growing or declining populations, it often cannot be managed; however, there are factors (e.g., human fishing activity) that may be influencing the prey availability that can be managed (FOC 2013). As of 1993, humpback whales were believed to target small schooling fish such as capelin and sand lance in the park, and were believed to target Pacific herring in surrounding waters (Icy Strait) (Kreiger and Wing 1984, 1986; as cited in Gabriele 1994). In 2004, humpback whales were observed consuming Pacific herring, capelin, sand lance, and one juvenile pink salmon (*Oncorhynchus gorbuscha*) in GLBA and surrounding waters (Doherty and Gabriele 2004). Doherty and Gabriele (2004) believe the higher abundance of whales may indicate a higher abundance of available prey items in the park. In 2012, Neilson and Gabriele (2013) noted fewer whales feeding on capelin in the park and surrounding waters. Pteropods (e.g., *Limacina helicina*) and Pacific sand lance were also believed to be consumed by humpback whales even though they were not directly observed. It should be noted that prey availability in the park may have changed between 2012 and present day.

Competition is a possible stressor for humpback whales, influencing both population abundance and distribution. Intraspecific competition may occur when whales have begun to exhaust higher quality prey items or when prey resources in general are declining (FOC 2013). The diets of several other species in GLBA (e.g., sea lions, salmon, seabirds) overlap with humpback whales; changes in the abundance of these species could impact whale populations. Commercial fisheries can compete with whales by harvesting important prey resources. Commercial fishing is limited in GLBA so impact is lower in the park, but fishing activity in surrounding waters may impact those whales that do not stay within park boundaries.

<u>Data Needs/Gaps</u>

There is consistent long-term monitoring of many humpback whale parameters within GLBA. Data on distribution, population estimates, and crude birth rates date back to 1985 and have been monitored on a yearly basis up until the present date. There are no available data on prey abundance and distribution, nor is there literature pertaining to the resilience of prey species to climate change and ocean acidification. Continued monitoring will aid park staff in managing the GLBA whale population.

<u>Overall Condition</u>

Distribution

The project team defined the *Significance Level* for distribution as a 2. Humpback whales appear to occupy a large area of GLBA and Icy Strait. They are known to use the mid-channel as well as nearshore areas, and while some sightings have occurred in the West Arm and Muir Inlet, a majority of whale sightings have occurred from the entrance of the Bay to the main channel (Neilson et al. 2014, 2015). Humpback whale distribution in recent years is similar to observations from 1985 (Baker 1985) and possibly even more widespread. Accordingly, this measure was assigned a *Condition Level* of 0, or of no concern.

Population Estimate

The project team defined the *Significance Level* for population estimate as a 2. Various population estimates have indicated increasing humpback whale numbers in recent decades in the Glacier Bay region and throughout Southeast Alaska (Straley et al. 2009, Hendrix et al. 2012, Saracco et al. 2013) Saracco et al. (2013) documented an annual rate of increase of 4.4% for the whale population in GLBA between 1985 and 2009 and an even higher rate of 7.7% more recently (2002-2009). This rate of increase is well above the reference condition set for this assessment (4.4%). Neilson et al. (2014, 2015) reported total whale counts in GLBA between 1985 and 2013 and noted an overall increasing trend during the 30-year monitoring period. As a result, this measure was assigned a *Condition Level* of 0, or of no concern.

Crude Birth Rate

The project team defined the *Significance Level* for crude birth rate as a 3. Neilson et al. (2015) documented crude birth rates varying between 1984-2014, with a peak rate of 18.5% (1992) and a low rate of 3.4% (2003). The crude birth rate averaged 9.4% over the study period, which is within the reference condition range (6.7-12.1%). Over the last 30 years, 24 years showed a crude birth rate within or above the reference condition range (Figure 89). The reasons for the lower crude birth rates in recent years are unknown, but may indicate increased predation on calves, or factors related to maternal condition, including the approach of senescence or changes in prey availability. As a result, this measure was assigned a *Condition Level* of 1, or of low concern.

Weighted Condition Score

The *Weighted Condition Score* for humpback whales at GLBA is 0.14, indicating that the component is of good condition. The available data display an improving trend in the GLBA humpback whale population, particularly as a function of the increasing humpback whale numbers.

Humpback Whales			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.14**
Distribution	2	0	
Population Estimate	2	0	
Crude Birth Rate	3	1	

4.12.6. Sources of Expertise

- Chris Gabriele, GLBA Wildlife Biologist, Humpback Whale Monitoring Program

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4.13. Sea Otters

4.13.1. Description

The sea otter (Photo 35) is a member of the Mustelidae family. Similar to other mustelids, sea otters have large heads, short necks and limbs, and elongated bodies (Estes 1980). Male otters are generally larger than females. Males can weigh as much as 45 kg (100 lbs) and be as long as 148 cm (58 in), while females may weigh as much as 32.5 kg (72 lbs) and grow as long as 140 cm (55 in) (Estes 1980). Sea otters have soft, thick fur (called pelage) to provide insulation and assist with thermoregulation, due to their lack of a blubber layer (Estes 1980, Riedman and Estes 1990). Sea otter pelage is the thickest among mammals, with as many as 100,000 hairs per square cm (Kenyon 1969, Estes 1980). The pelage ranges from dark brown to a reddish brown; gray may also be present on the heads of older individuals (Estes 1980).

Photo 35. Sea otter swimming in GLBA (NPS photo).

Sea otters are part of the sub-arctic nearshore ecosystem of Alaska, ranging from southeastern Alaska, through Prince William Sound, the Kenai Fjords, and the Aleutian Islands. Sea otters in the state are further divided into three stocks: Southwest, Southcentral, and Southeast. The Southwest Alaska stock was listed as threatened under the Endangered Species Act in 2005 (USFWS 2013a), but the Southcentral and Southeast stocks do not have any federal protection (USFWS 2014). Sea otters in GLBA are part of the Southeast Alaska stock, which ranges from Dixon Entrance, along the southern Alaska boundary, northwest to Cape Yakataga (USFWS 2014). Sea otters typically prefer shallow coastal water and kelp beds, and feed primarily on aquatic invertebrates (e.g., clams, mussels, crabs, and sea urchins) (Riedman and Estes 1990). Their distribution is limited to shallow nearshore areas by their need to dive to forage, as the maximum diving depth for sea otters is around 50 m (164 ft) for females and 80 m (262 ft) for males (Bodkin et al. 2004). Sea otters are keystone predators that have cascading effects on coastal marine ecosystems (Estes et al. 1978, Mills et al. 1993). The absence of sea otters would allow their prey numbers (e.g., sea urchins [Class Echinoidea]) to increase, which would decrease the abundance of lower trophic levels, particularly kelps (Order Laminariales), with subsequent cascading effects for organisms that rely on kelp beds (Estes et al. 1978, Mills et al. 1993).

A significant decline in global sea otter numbers occurred as a result of commercial harvesting for the Russian fur trade, which began around 1750 (Bodkin 2003). Historic sea otter abundance before commercial harvesting was estimated to be between 150,000-300,000 worldwide (Kenyon 1969, Johnson 1982; as cited in Riedman and Estes 1990). Sea otters declined to an estimated 1,000 to 2,000 animals along the Pacific Coast before the Fur Seal Treaty was established in 1911 (Riedman and Estes 1990). Additional conservation and recovery efforts in Alaska involved translocating 412 sea otters from southwestern and southcentral Alaska to several sites along the outer coast of southeastern Alaska between 1965 and 1969 (Jameson et al. 1982, Bodkin 2003). According to Bodkin (2003), sea otters increased in numbers and expanded their range in the southeastern Alaska region, becoming abundant once again in areas surrounding GLBA (e.g., Cross Sound, Icy Strait) by 1988. Sea otters were first documented in Glacier Bay in 1993 (Bodkin 2003). Sea otter numbers have increased dramatically in GLBA since 1993. Abundance in the park grew from a few individuals in 1995 to approximately 2,400 otters in 2004 (Bodkin et al. 2007) and 8,500 in 2012 (Esslinger et al. 2013). The park provides both an abundance of prey and refuge from human harvest (Bodkin et al. 2007).

4.13.2. Measures

- Trends in distribution and abundance
- Foraging effort (including prey diversity, energy recovery rates, and morphometrics)
- Kelp abundance, invertebrate prey abundance and size
- Strandings

4.13.3. Reference Conditions/Values

While little is known about sea otter numbers in Glacier Bay prior to over-exploitation from the fur trade, in natural situations sea otter density is likely limited by food availability (Riedman and Estes 1990; Esler, written communication, 9 June 2014). Numerous studies over the past decade have explored varying otter densities relative to carrying capacity and have found that foraging observations (which measure energy recovery rates) can be useful in inferring population condition (Esler, written communication, 9 June 2014). According to Bodkin and Monson (2003), the annual growth rate of an undisturbed, translocated, and non-food-limited sea otter population has averaged 18% but could reach 24%.

4.13.4. Data and Methods

Aerial Surveys to Estimate Distribution and Abundance

Bodkin et al. (1999, 2002a, 2006) and Esslinger et al. (2013) conducted aerial surveys of sea otters population in GLBA from 1999-2012. Strip transects (high density, low density) were run (Figure 91) and intensive search units were used to estimate observer correction factors (Bodkin and Udevitz 1999). Transects were 400 m (1,312 ft) wide and spaced 2,000 m (6,562 ft) apart in high strata areas and 4,000 m (13,123 ft) apart in low strata areas. In 2012, three replicate aerial surveys were conducted between 23 May and 26 May, with survey times ranging from 6-8 hours. Geike Inlet, Charpentier Inlet, and Scidmore Bay were excluded from the survey because otters were considered absent during the reconnaissance flights. Data recorded from the transects include date, transect number, location, group size, and group activity.

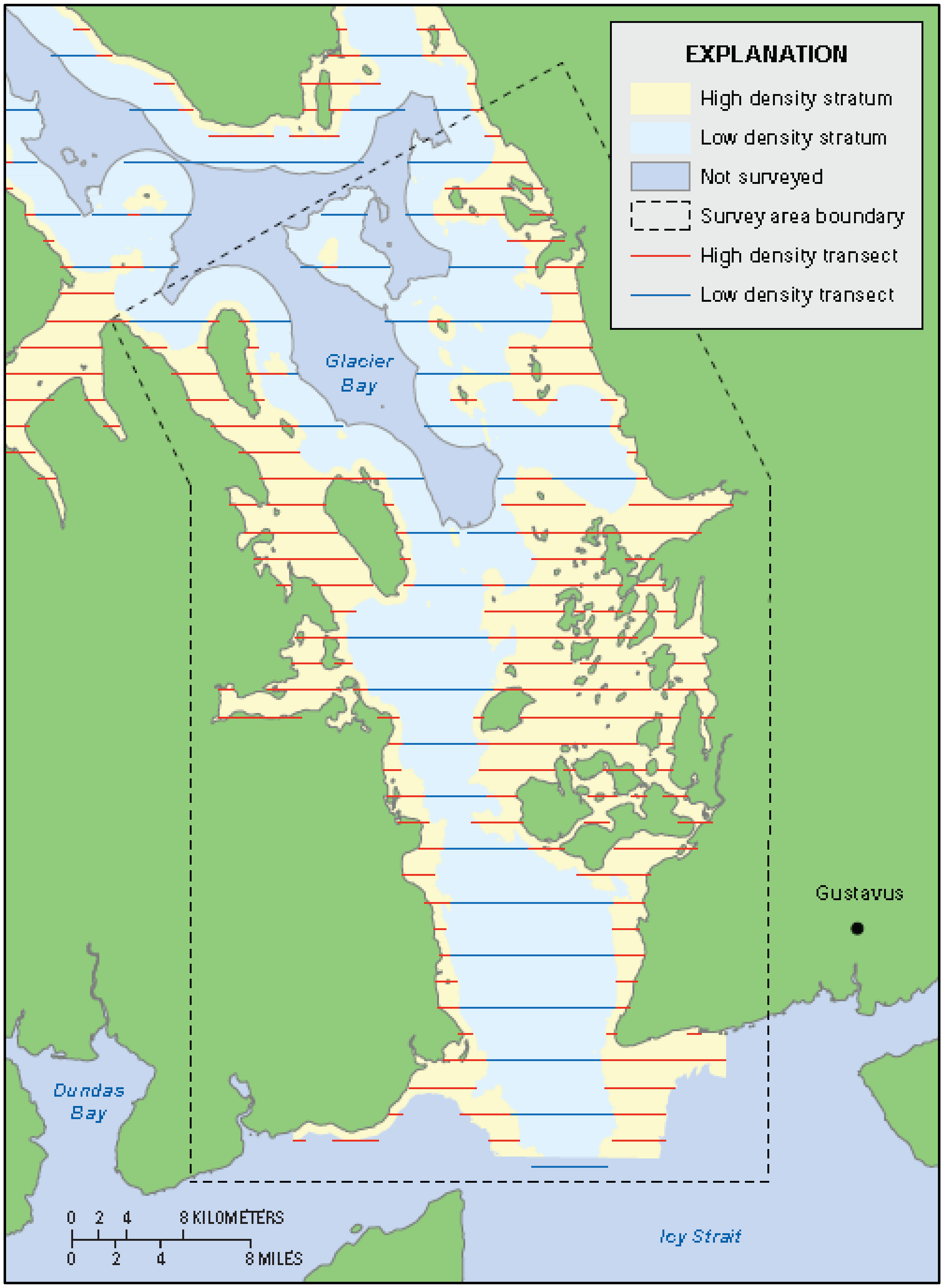

Figure 91. General location of sea otter aerial survey area and transects. The dotted line represents the study boundary through 2006; after 2006, the boundary was expanded northwest to the Scidmore Bay area (Esslinger and Bodkin 2009).

USFWS (2014) summarized sea otter population dynamics in Southeast Alaska based on aerial survey results from the region. The summary includes population size and current population trend for the southeast Alaska stock of northern sea otters. Population estimates and trend were split into five survey areas: North Gulf of Alaska, Glacier Bay, Northern Southeast Alaska, Southern Southeast Alaska, and Yakutat Bay.

Marine Predator Surveys and Opportunistic Observations

Bodkin et al. (2002b) prepared an annual report for marine predator surveys in GLBA between 1999 and 2001. Transects were run along the coast and in open water areas throughout the Bay in June of each year of the survey. In 2001, three vessels were used to survey in March and June. Sightings within 300 m (984 ft) in front of the vessel, and up to 150 m (492 ft) to the side of the vessel were recorded. Records included taxa, count, and activity. GPS units were used to plot transects and locations of predator observations.

Gabriele et al. (2012) provided a summary of opportunistic marine mammal sightings in Glacier Bay between 1994 and 2010. Mammal sightings were reported by NPS biologists during whale monitoring surveys, which occurred from June to August of each year (1994-2010). The main area surveyed fell between Garforth Island, Geikie Inlet, Point Carolus, and Bartlett Cove (Figure 92). This area was surveyed three to four times a week. Areas as far north as Russell Island and Adams Inlet were surveyed intermittently. Most sightings took place within 100 m (328 ft) of the vessel to ensure proper identification. The NPS (2014) spatial database was used to supplement this summary and display sea otter sightings from 1994 to 2013.

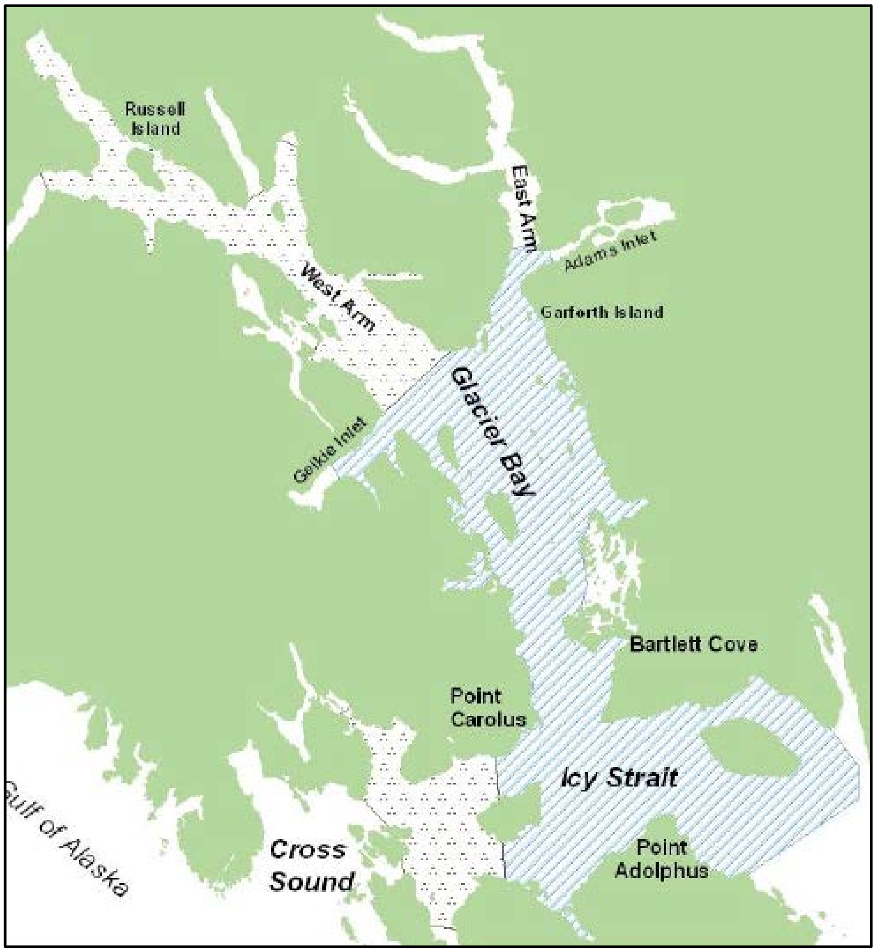

Figure 92. Opportunistic marine mammal survey area (Gabriele et al. 2012). Cross-hatched areas were surveyed weekly and dotted areas intermittently.

<u>Sea Otter Foraging and Intertidal Community Studies</u>

Estes and Duggins (1995) studied variation in the community structure of Alaska kelp forests with and without sea otters present. Study sites in southeastern Alaska with sea otters included Torch Bay on the outer coast of GLBA (Figure 93), which was sampled in May 1988. Data gathered included kelp density, suspension feeder cover, and sea urchin density, biomass, and maximum size (Estes and Duggins 1995). Torch Bay results were compared to surveys from 1978, prior to sea otter recolonization (Duggins 1980). Estes (2003) returned to Torch Bay in 2003 to resample the kelp forest community and report on changes since 1988.

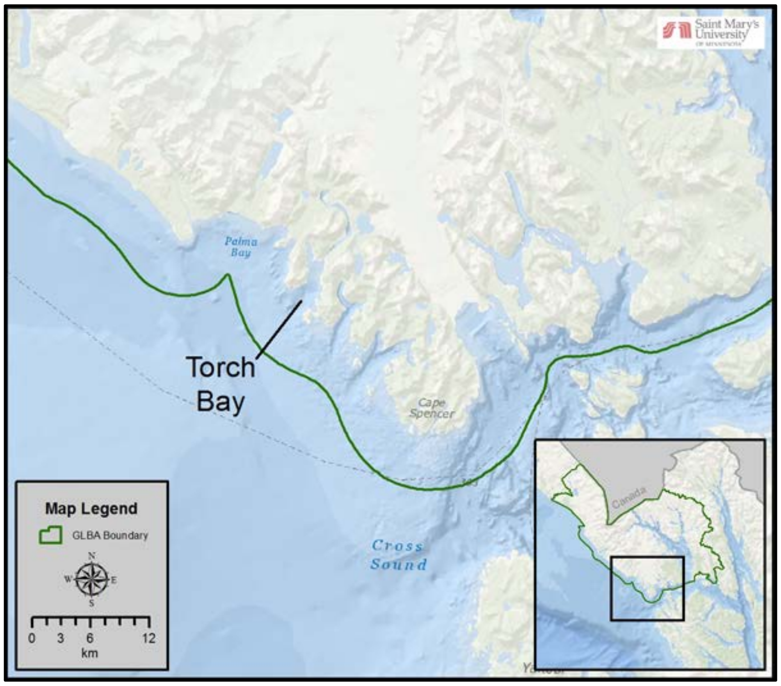

Figure 93. Location of Torch Bay.

Bodkin and Kloecker (1999) conducted a study of intertidal clam communities in GLBA in 1999. The study area consisted of intertidal areas between the entrance of the Bay, Geikie Inlet, and Sandy Cove. Twenty random sites and six preferred clam habitat sites were sampled. Sampling included recording intertidal clam species richness, size distributions, and abundance. A 200 m (1,312 ft) transect was run horizontal to the mean lower low water (MLLW) tide line at each sampling site. Random points on transects were selected for starting points, then ten 0.25 m^2 quadrats were excavated at 20 m (~66 ft) intervals.

Donnellan et al. (2002) sampled the benthic biota of 31 nearshore sites in mid- and lower Glacier Bay from 2000-2002 (Figure 94). These data were used to assess the impacts of sea otter colonization on the nearshore marine community. Taxa studied represented three trophic levels: primary producers (e.g., kelp), primary consumers (e.g., sea urchins), and secondary consumers (e.g., sea stars) (Donnellan et al. 2002). The densities of these taxa were measured at each site, as well as the size of select otter prey species at sites where they occurred.

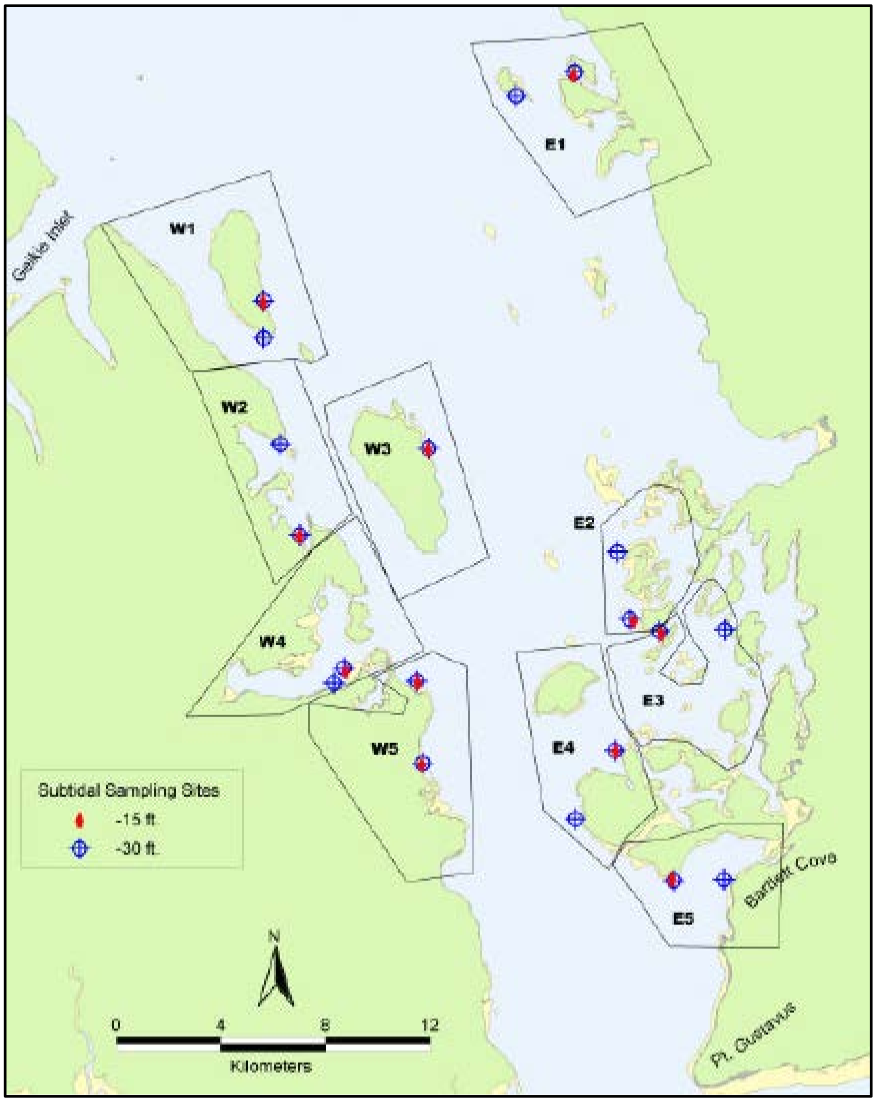

Figure 94. Donnelan et al. (2002) nearshore (subtidal) community study sites (Donnellan et al. 2002).

Weitzman (2013) conducted a study on the effects of sea otter colonization on soft-sediment intertidal prey assemblages in GLBA. Previous USGS sea otter abundance aerial survey data were used for annual distribution and to calculate kernel density. Changes in density were observed along with annual foraging observations to determine a correlation between otter presence and density on different prey species (Weitzman 2013). Over 14,000 foraging bouts (i.e., dives) were documented using high-powered telescopes, from shore or boats. If bouts were successful, the type, number, and size of prey obtained were noted. Changes in energy recovery rates (energy gained from consuming various prey) of foraging otters were also observed. Intertidal community prey populations (e.g., clams, urchins) were sampled at 45 sites in 2010-2011 (Figure 95) and compared to results from 1998-2000 surveys of the same locations. Parameters studied included species richness, biomass, and size distribution of sea otter prey species (Weitzman 2013).

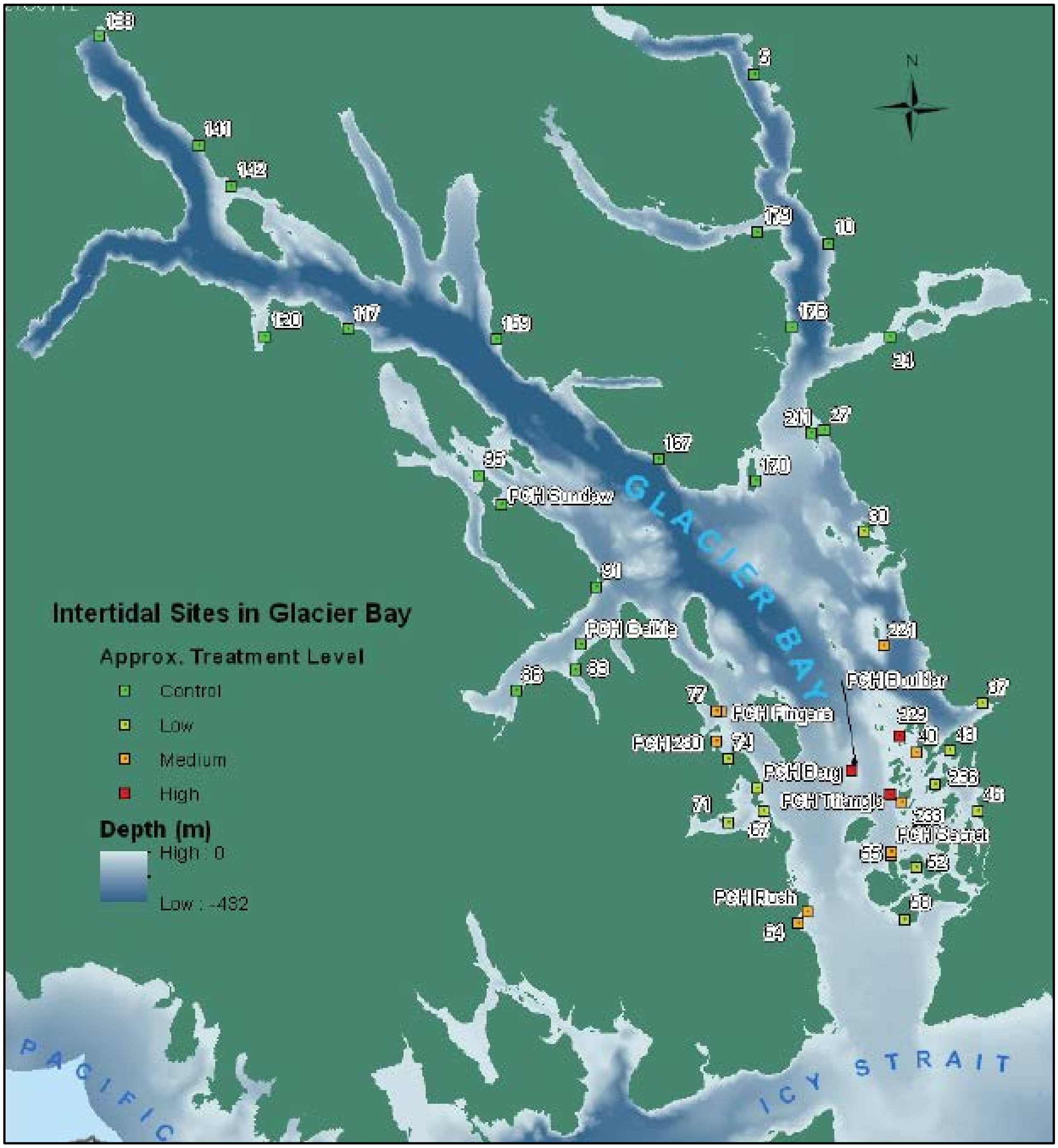

Figure 95. Intertidal community study sites throughout Glacier Bay (Weitzman 2013).

<u>Sea Otter Strandings</u>

NOAA (2014) compiled sea otter stranding data in GLBA and surrounding areas between 2006 and 2013. Stranding records included date, year, location name, global positioning system (GPS) coordinates, necropsy, cause of death (if known), and additional comments about the state of the animal. It should be noted that not all of the records had GPS coordinates.

4.13.5. Current Condition and Trend

<u>Trends in Distribution and Abundance</u>

Distribution

Bodkin et al. (1999, 2002a) documented the distribution of sea otters in GLBA from 1999 to 2001. Observations suggested that sea otters occurred throughout the Lower Bay, from Leland Island in the north to Point Carolus in the south (Figure 96). In 1999, concentrations were noted near Leland Island, Boulder Island, and Point Carolus (Bodkin et al. 1999). Bodkin et al. (2002a) also observed higher concentrations of otters around Sita Reef and from Point Carolus to Rush Point, with occasional sightings north of Sandy Cove. Bodkin et al. (2006) documented the distribution of sea otters in GLBA in 2004. Monthly surveys showed a year-round presence of otters in the Lower Bay. Areas with higher otter concentrations were again Sita Reef, Boulder Island, and between Point Carolus and Rush Point.

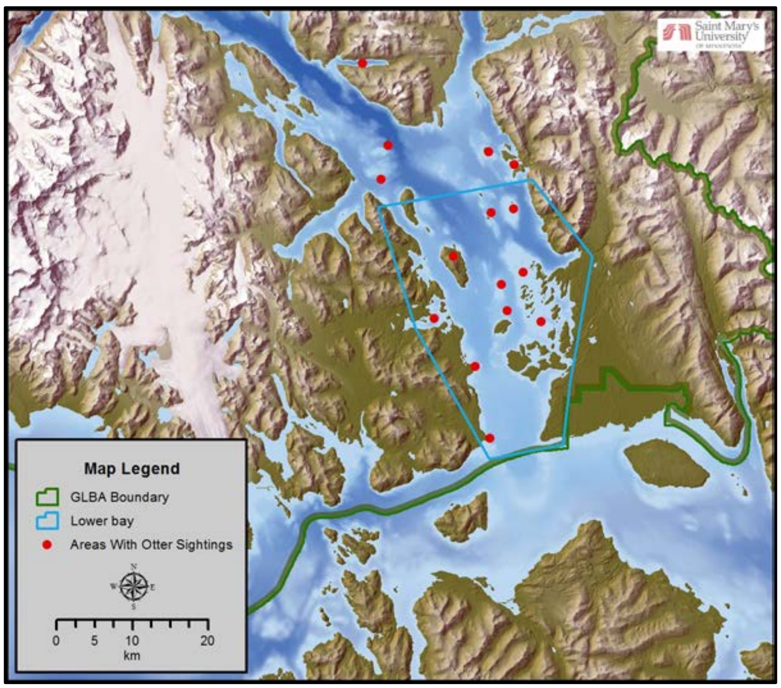

Figure 96. Areas in GLBA occupied by sea otters (Bodkin et al. 1999, 2002a, 2006).

Weitzman (2010) observed changes in sea otter distribution throughout the summer of 2010. Sea otters were observed at locations further north (Lone Island, Sturgess Islands) than previously observed. In May and June, very few or no otters were observed in the Berg Bay area and hundreds of otters were observed by Sita Reef. In June and July, hundreds of otters were observed near Berg Bay, and few or no otters were observed at Sita Reef. Weitzman (2011) then noted areas where sea otters were abundant in 2011. Sea otters commonly use Flapjack Island, Boulder Island, Inner Beardslee, and Geikie Rocks (Weitzman 2011).

Gabriele et al. (2012) reported sea otter distribution in GLBA between 1994 and 2010. A large number of sightings occurred nearshore, particularly in the Lower Bay near Flapjack Island, Idaho

Inlet, and along the coast south of Berg Bay (Figure 97). Sea otters were only sighted in the lower bay area between 1994 and 1999. Distribution began to spread to the middle bay area between 1999 and 2003. Sea otters were sighted in the lower portion of the east and west arms between 2004 and 2008. Expansion of sea otter distribution to the mid-channel and east and west arms occurred between 2009 and 2013. Sea otters sightings have occurred as far north as Tidal Inlet. Figure 97 displays sea otter distribution trends in GLBA between 1994 and 2013 (NPS 2014).

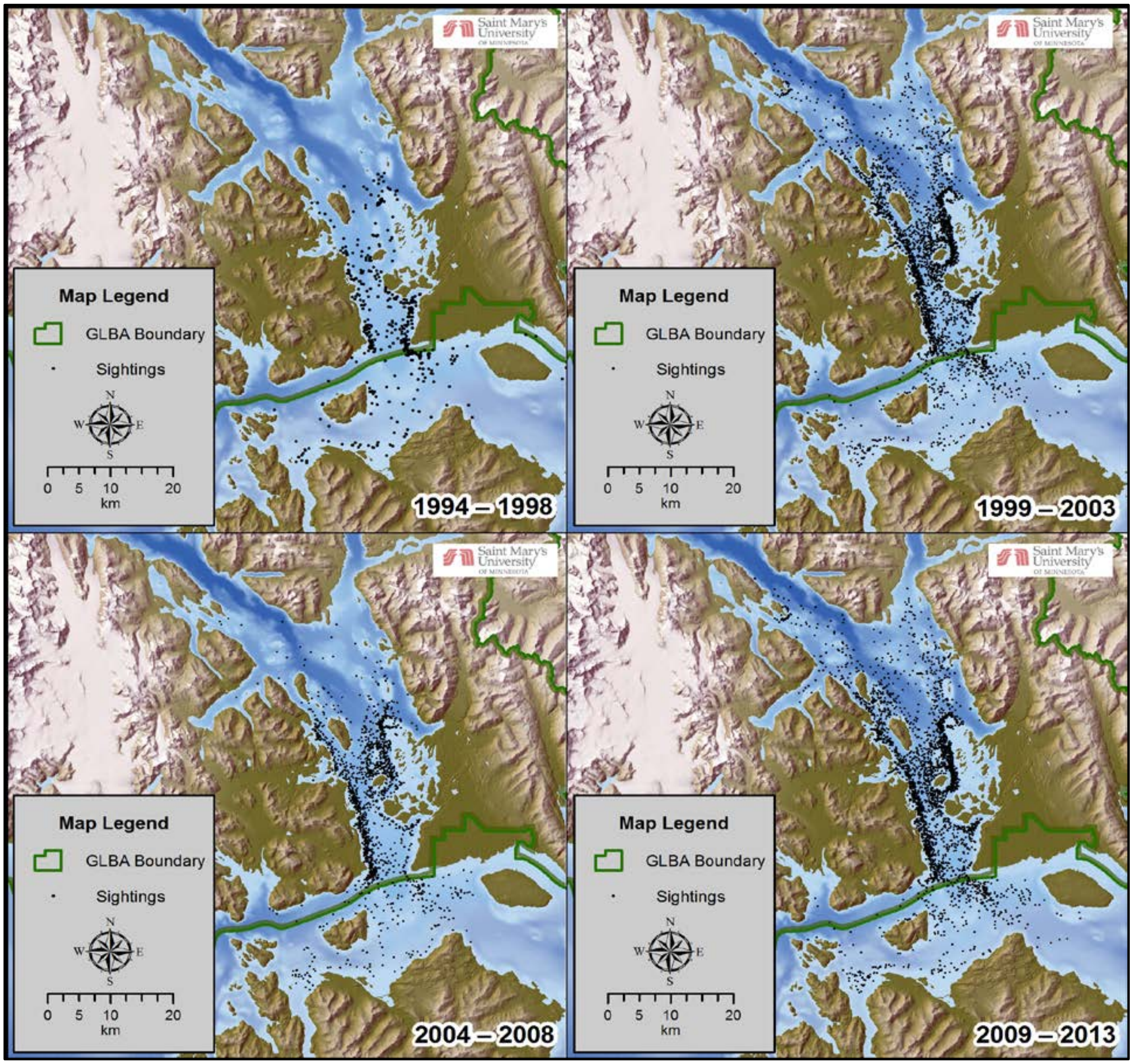

Figure 97. Sea otter distribution according to opportunistic sightings during marine mammal vessel surveys in Glacier Bay and Icy Strait between 1994 and 2013 (NPS 2014).

Esslinger et al. (2013) documented sea otter distribution and abundance with three aerial surveys in GLBA in 2012. Sea otters were observed throughout the Central and Lower Bay areas. Sea otter distribution had expanded since the last abundance surveys in 2006; otters were seen north of Drake and Sturgess Islands in the lower portions of the east and west arms of the bay. It should be noted that Geike Inlet, Charpentier Inlet, and Scidmore Bay were excluded from the survey because otters were considered absent during the reconnaissance flights. Figure 98 displays sea otter distributions from the three replicate surveys in GLBA in 2012.

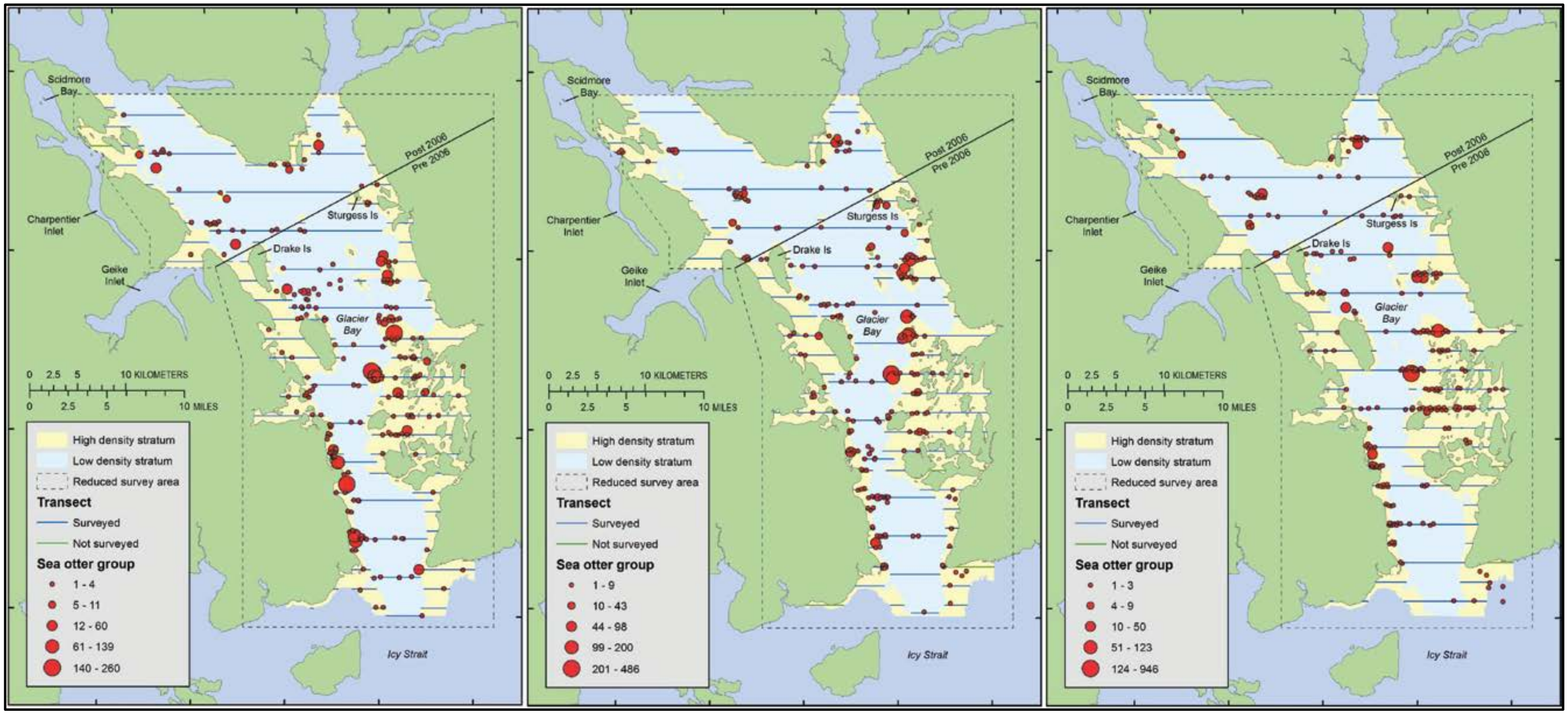

Figure 98. Distribution and abundance of sea otters from aerial survey replicate #1 (left), replicate # 2 (middle), and replicate # 3 (right) in GLBA in May 2012 (Esslinger et al. 2013).

Abundance

Sea otter abundance has been increasing in Southeast Alaska for several decades. According to USFWS (2014), the Southeast Alaska stock (of which the GLBA population is a part) was estimated at 10,563 individuals in 2003. As of 2012, the Southeast Alaska stock was estimated to be 25,712 otters, more than double the estimate from 2003 (USFWS 2014).

Bodkin et al. (1999) reported the mean sea otter abundance estimate for GLBA from the four aerial surveys conducted in 1999 and referenced historical trends. Bodkin et al. (1999) also included otter counts from 1994 to 1998. Sea otters were first observed in Glacier Bay in 1993, and only five individuals were observed in the park by 1995. Since then, numbers have shown rapid growth. Sea otter counts for GLBA remained low between 1996 and 1999. In 1996 and 1997, sea otter counts were 39 and 21 individuals, respectively. The mean estimate for the 1999 survey was 384 otters.

Abundance estimates increased from 384 sea otters in 1999 to 2,381 sea otters in 2004 (Bodkin et al. 2006), with a significant jump between 2000 and 2001. The percent increase in sea otter counts has been consistently positive since 2000, with the smallest increase in 2002 (2.3%) and the largest increase in 2001 (123.5%). According to Bodkin and Esslinger (2006), the abundance estimate increased to 2,785 animals by 2006 (Table 36).

Table 36. Sea otter counts (1994-1998) and population estimates based on aerial surveys (1999-2012) for GLBA, 1994-2012 (Bodkin et al. 2006, Esslinger et al. 2013).

Year	Number of Otters
1994	0
1995	5
1996	39
1997	21
1998	209
1999	384
2000	554
2001	1,238
2002	1,266
2003	1,866
2004	2,381
2006	2,785
2012	8,508
Avg. annual growth rate (1995-2012)	**42%**

Esslinger et al. (2013) documented continuing growth in sea otter abundance in GLBA and southeast Alaska. Based on aerial surveys, the GLBA abundance estimate for 2012 was 8,508 otters (Table 36). The average annual growth rate in GLBA between 1995 and 2012 was estimated to be 42%, which is almost double the reference condition and what was originally expected (Esslinger et al. 2013). This high rate, well above the maximum estimated rate of increase for the species (24% [Estes 1990]), must reflect both reproduction and significant immigration to the Glacier Bay area (Esslinger et al. 2013). The lack of hunting in the park may be contributing to the high growth rate, as annual harvests in other areas of Alaska take approximately 8% of the population (Esslinger and Bodkin 2009, Esslinger et al. 2013).

Gabriele et al. (2012) summarized opportunistic sea otter sightings during vessel-based surveys in GLBA between 1996 and 2010. The number of sea otter sightings has shown an increasing trend since 1996 (with relatively consistent survey effort over time). In 1996, the number of otters observed totaled 340 in 91 sightings. By 2010, there were 1,124 otters observed in 639 sighting events. Figure 99 displays the total number of otters sighted and the number of sightings annually in GLBA between 1996 and 2010.

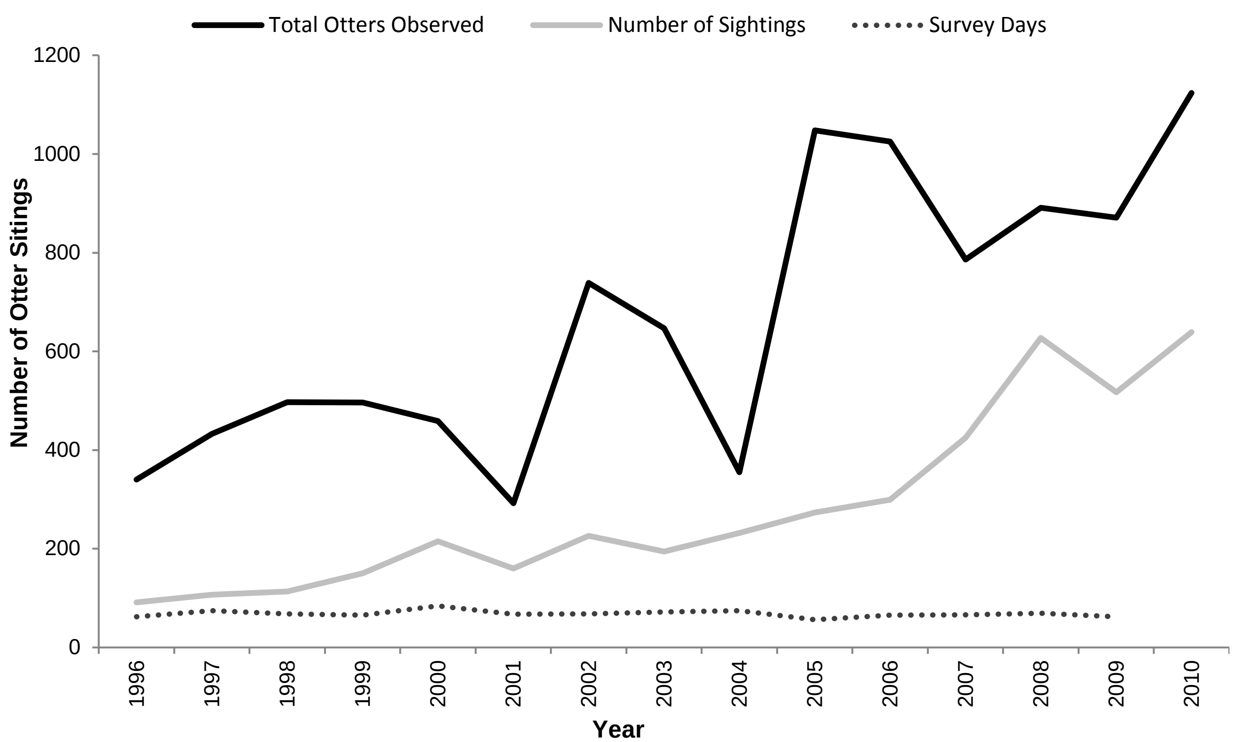

Figure 99. Total number of sea otters observed and the number of sightings during vessel-based marine mammal surveys in GLBA between 1996 and 2010 (Gabriele et al. 2012). Total otters observed is higher than number of sightings, as multiple otters are often observed during each sighting.

Foraging Effort

Sea otter foraging observations can be used to estimate energy recovery rates, which provide some insight into a population's status (Womble, written communication, May 2015). Documenting foraging effort allows managers to evaluate where sea otters in Glacier Bay lie on the continuum from "newly-occupied" (i.e., abundant prey resources) to "approaching carrying capacity" (i.e., maximum utilization of prey resources). When sea otters initially move to an area, foraging effort (time spent foraging) is low, the diversity of prey selected is low (higher energy prey is abundant), and prey size is large (larger prey are targeted first) (Weitzman 2013; Esler, written communication, 13 June 2014). As otter density increases, prey size typically decreases and prey selection diversifies (higher energy prey are no longer abundant), resulting in higher foraging effort (Weitzman 2013).

Foraging Bouts and Prey Diversity

Bodkin et al. (2006) documented 1,232 foraging events at GLBA in 2004, and defined a dive as successful if one or more prey items were retrieved. There were 1,120 successful dives (92.6% success rate). Thirty-six prey types were reported as consumed by sea otters in GLBA between 1993 and 2004, including clams, mussels, scallops, gastropods, mollusks, sea stars, urchins, and crustaceans (Table 37, Appendix H). The classes with the highest number of species consumed were clams and crustaceans, with 10 and seven species, respectively. Commonly consumed prey species included butter clams (*Saxidomus giganteus*), Greenland cockles (*Serripes groenlandicus*), littleneck clams (*Protothaca staminea*), softshell clams (*Mya* spp.), green urchins (*Strongylocentrotus droebachiensis*), horse mussel (*Modiolus modiolus*), tanner crab, kelp crab (*Pugettia gracilis*), and helmet crab (*Telmessus cheiragonus*).

Table 37. Sea otter diet composition in GLBA in 2004 and from 1993 to 2011 (Bodkin et al. 2006, Weitzman 2013).

Prey Item	Percent Diet Composition		
	2004 (Bodkin et al. 2006)	1993 – 2011 (Weitzman 2013)	
		All Bouts	Intertidal Only
Clam	56%	42%	56.4%
Mussel	18%	25%	14.7%
Crab	2%	7%	5.5%
Urchins	9%	21%	17.8%
Snail	--	3%	4%
Other	4%	1%	5.6%
Unidentified	13%		

Weitzman (2013) documented over 14,580 foraging bouts (i.e, dives) between 1993 and 2011. Over 6,759 foraging bouts were considered successful. Forty-six different prey species were identified. Similar to Bodkin et al. (2006), clams, mussels, scallops, gastropods, mollusks, sea stars, urchins, and crustaceans were all preyed upon during this study (Appendix H). Clams, urchins, and mussels were the most common prey items, comprising approximately 42%, 21%, and 25% of sea otter diets, respectively. Weitzman (2013) identified eight species of clams being consumed including butter clams (31.2%), Greenland cockles (5.5%), Mya spp. (2.2%), Nuttall's cockle (*Clinocardium nuttallii*) (2.1%), littleneck clams (1.4%), a combination of rock entodesma (*Entodesma navicula*), Arctic surfclams (*Mactromeris polynyma*), and a *Macoma* species (<1.0%). An unidentified clam species was being consumed during 13.76% of foraging bouts (Weitzman 2013).

Weitzman (2013) also analyzed changes in sea otter energy recovery rates (energy gained from consuming prey [kcal/minute]) over time. Energy recovery rates can indicate overall prey conditions; high energy recovery rates in sea otters occur when preferred prey are abundant and foraging effort is low (Weitzman 2013). Early in the colonization of Glacier Bay, sea otters experienced high average energy recovery rates (18.91 ± 2.12 kcal/min overall, 19.14 ± 2.29 kcal/min in intertidal zones) characteristic of an area with abundant resources (20 kcal/min). Average sea otter energy recovery rates dropped to 15.08 ± 1.74 kcal/min overall (12.19 ± 1.45 kcal/min intertidal) 5-10 years after occupying GLBA, and dropped again to 9.21 ± 1.34 kcal/min (9.63 ± 1.22 kcal/min intertidal) after >10 years of occupying the park (Figure 100; Weitzman 2013). An average energy recovery rate of 9.0 kcal/min was recorded in other locations with otters considered to be at equilibrium density. Lower recovery rates mean that longer foraging times would be required, on average, to obtain the same amount of energy (Weitzman 2013).

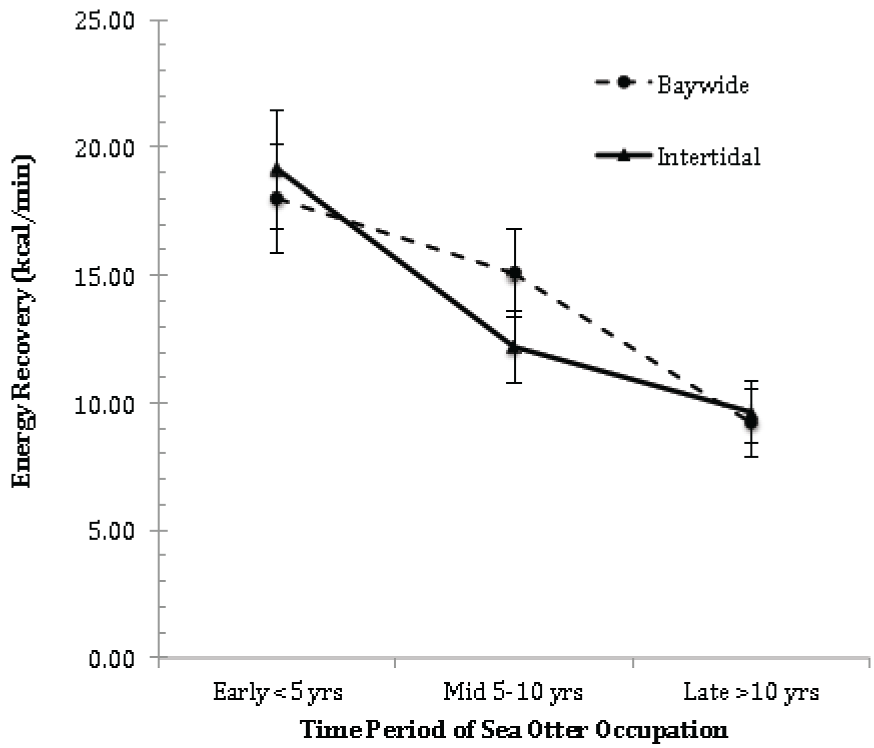

Figure 100. Rates of energy recovery by sea otters overall (baywide) and in intertidal zones alone in GLBA from 1993 to 2011 (Weitzman 2013).

Morphometrics

Morphometrics refers to measurements of various body characteristics (e.g., length, girth, mass), which can serve as an index of body condition (Womble, written communication, May 2015). According to Esler (written communication, 13 June 2014), new relationships between environmental conditions and morphometrics are being studied. For example, if pups are raised under great foraging conditions (e.g., when newly occupying a site), they grow faster and end up at a larger size. As the population gets closer to food-dictated carrying capacity, size declines, so average size can be an index to foraging conditions and hence population status relative to carrying capacity (Esler, written communication, 13 June 2014).

According to Monson and Bowen (2015), information on the average body condition of a sea otter population gained through morphometrics may help managers understand how most of the population will respond to fluctuations in resources (e.g., reductions in prey resources). Individual animals in a population with above average or optimal condition may be less vulnerable to short-term fluctuations. Individuals in a population with less than optimal average body condition may struggle with declines in prey abundance. Morphometric information could also be used to estimate the age at death of sea otter carcasses found in the park (Womble, written communication, May 2015). Currently, there are no available data regarding sea otter morphometrics (average body characteristics) in GLBA. The potential inclusion of morphometric data in SEAN monitoring programs would allow this measure to be assessed in the future.

Kelp Abundance, Invertebrate Prey Abundance and Size

Kelp Abundance

Studies in California and the Aleutian Islands have identified correlations among sea otters, kelp-eating herbivores, and kelp abundance. Several sources state that higher kelp coverage and diversity occur in areas with sea otter populations relative to areas without otters (Estes and Duggins 1995, Bodkin et al. 2006). According to Riedman and Estes (1980), sea otters directly affect urchin populations, which feed on kelp. The presence of sea otters results in lower numbers of sea urchins, allowing the growth of kelp forests. As a result, increasing sea otter populations in the park may mean abundant kelp forests.

Estes and Duggins (1995) and Estes (2003) compared kelp density in Torch Bay in the absence of sea otters (1978) and after otter recolonization (1988, 2003). Kelp increased greatly following recolonization (1978-1988) and then appeared to have stabilized (1988-2003) (Estes and Duggins 1995, Estes 2003). The densities of various kelp species in Torch Bay in 1988 and 2003 are shown in Table 38.

Table 38. Density (number per 0.25 m^2) of kelp species in Torch Bay (Estes 2003). For comparison, the total density of all kelp species in Torch Bay in 1978 was 3.13 plants/ 0.25 m^2 (Estes and Duggins 1995).

	Density (per 0.25 m^2)	
Species	**1988**	**2003**
Agarum cribrosum	–	0.13
Alaria marginata	2.07	–
A. fistulosa	0.41	0.18
Costaria costata	0.22	0.08
Cymathera triplicata	0.13	0.50
Laminaria spp.	8.74	3.35
L. yezoensis	0.35	0.51
Nereocystis leutkeana	0.99	0.15
Pleurophycus	1.59	1.83
Total large kelps	**14.50**	**6.72**
Small kelps	**32.92**	**4.45**

Donnellan et al. (2002) recorded kelp density at study sites in mid to lower Glacier Bay from 2000-2002. Kelp was present at 23 of 31 sites, although density was low (<1 plant/m^2) at 16 sites. A maximum kelp density of 20 plants/m^2 was observed near Lester Point on the east side of the Bay (Donnellan et al. 2002). One study site showed an increase in kelp density from zero in the year 2000 to 7 plants/ m^2 in 2002. Donnellan et al. (2002) also included a map of locations with kelp canopy, as identified during a 2000 aerial survey (Figure 101).

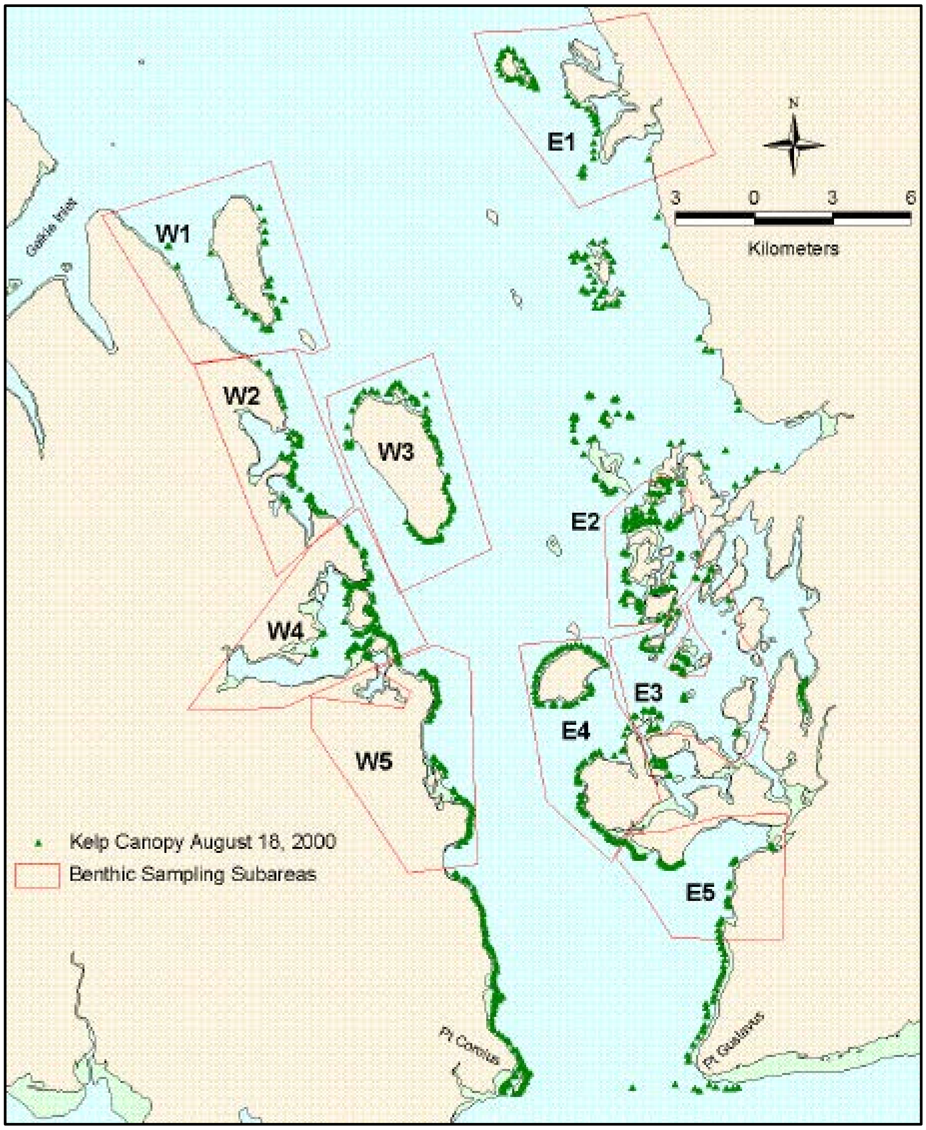

Figure 101. Map of kelp canopy in lower and mid-Glacier Bay identified during a 2000 aerial survey (Donnellan et al. 2002).

Invertebrate Prey Abundance and Size

Invertebrate prey abundance and size can be indicators of sea otter status because low abundances of larger prey may indicate that sea otter densities are reaching carrying capacity (Riedman et al. 1990). For example, sea urchins (a preferred prey item) are usually 65-85 mm (2.6-3.3 in) in areas where sea otters are not established; however, sea urchins may be less than 25 mm (1.0 in) in areas where sea otters are near equilibrium density (Riedman et al. 1990). It should be noted that similar declines in prey size occur among all sea otter prey species with increasing sea otter occupation.

Estes and Duggins (1995) documented the density and biomass of three sea urchin species in Torch Bay in the absence of sea otters (1978) and after recolonization (1988). After sea otters returned, two of the three species became virtually absent from study sites in the bay and the third decreased substantially (Table 39; Estes and Duggins 1995).

Table 39. Sea urchin density and biomass in Torch Bay before (1978) and after (1988) sea otter recolonization (Estes and Duggins 1995).

Species	Density (per 0.25 m²)		Biomass (g/0.25 m²)	
	1978	1988	1978	1988
Strongylocentrotus franciscanus	1.48	–	982.8	–
S. droebachiensis	1.22	–	161.5	–
S. purpuratus	1.45	0.05	76.3	0.3

Bodkin and Kloecker (1999) documented the mean abundance and size of six clam species in GLBA in 1999. Mean clam abundance was estimated for each quadrat and ranged from 0 to 161.2 clams/ 0.25 m^2 (Bodkin and Kloecker 1999). The *Macoma* species and littleneck clams were the most abundant taxa, while Nuttall's cockles were least abundant (Table 40). Mean clam size ranged from 23.7 mm (0.9 in) to 71 mm (2.8 in). The clam species with the largest mean size were the butter clams, while the *Macoma* species had the smallest mean size (Bodkin and Kloecker 1999). More medium sized clams were observed than small and large size classes. Over 20% and over 30% of the *Macoma* species were in the16-20 mm (0.6-0.8 in) class and 21-25 mm (0.8-1.0 in) class, respectively. Approximately 60% of the littleneck clams from GLBA study areas were greater than 35 mm (1.4 in) (Bodkin and Kloecker 1999).

Table 40. Mean abundance and size of clam species in GLBA in 1999 (Bodkin and Kloecker 1999).

Species	Mean Number (per 0.25 m²)	Mean Size (mm)
Nuttall's cockle (*Clinocardium nuttallii*)	0.36	45.0
Arctic hiatella (*Hiatella arctica*)	6.94	26.0
Macoma spp. (*Macoma*)	26.19	23.7
Mya spp (*Mya truncata, Mya arenaria*)	3.7	40.2
Littleneck clam (*Protothaca staminea*)	10.34	39.0
Butter clam (*Saxidomus gigantea*)	2.95	71.0

Donnellan et al. (2002) surveyed sea otter prey species at 31 nearshore sites (Figure 94). The prevalence of each prey type (number of sites at which each was observed) and notes on density are presented in Table 41. Donnellan et al. (2002) also measured the sizes of sea urchins and large sea stars. Average urchin size was 33 mm (1.3 in) with a range of 19-54 mm (0.7-2.1 in). Sea star sizes varied by species and are shown in Table 42.

Table 41. Results from nearshore community surveys of 31 sites in mid and lower Glacier Bay, from 2000-2002 (Donnellan et al. 2002).

Prey type	# of sites present	Notes
Sea urchins	26	Low densities (<0.2/0.25 m^2) at 6 sites, abundant (>20/0.25 m^2) at 3 sites. Average density of 6.9/0.25 m^2 with a max of 180/0.25 m^2
Horse mussels	10	Low density (<0.2/0.25 m^2) at 7 of 10 sites
Clam siphons	28	Low density (<1/0.25 m^2) at 21 of 28 sites
Large sea stars	30	Ten of the 11 most dense sites (>1.5/10 m^2) were in the west bay
Small sea stars	19	Moderate/high densities (>1/10 m^2) at only 3 sites
Metridium (plumose anemones)	16	Densities greater than 1/10 m^2 at only 6 sites
Sea anemones	22	Densities greater than 1/10 m^2 at only 6 sites
Hermit crabs	30	Low densities (<1/10 m^2) at most sites
Other crabs	21	Never exceeded densities greater than 3/10 m^2
Sea cucumbers	16	Mean density >1/10 m^2 at only 1 site
Large whelks	28	Mean density of 4.4/10 m^2, maximum of >20/10 m^2

Table 42. Size ranges and means for large sea star species surveyed in nearshore sites (species with sample size <10 excluded from table) (Donnellan et al. 2002).

Species	Minimum (cm)	Maximum (cm)	Mean (cm)	Standard Deviation
Crossaster papposus	1	11	3	1.63
Evasterias troschelii	2	36	19	6.31
Henricia spp.	1	11	4	2.32
Leptasterias spp.	2	21	11	3.20
Mediaster aequalis	2	4.5	3	0.97
Pycnopodia helianthoides	1	50	21	13.15
Solaster spp.	1	24	8	4.70

Bodkin et al. (2006) documented four prey types and mean prey size in GLBA in 2004. Urchins and mussels were retrieved more on average per dive than any other prey type. On average, 4.6 urchins and 3.5 mussels were retrieved per dive (Table 43). Mussels were the largest prey type collected with a mean size of 85.2 mm (3.4 in). Crabs and clams had mean sizes of 65.0 mm (2.6 in) and 71.8 mm (2.8 in), respectively. Urchins were the smallest prey type at 44.9 mm (1.8 in). Table 43 displays GLBA sea otter prey type, the average number of each prey type retrieved per dive, and mean prey size.

Table 43. Sea otter prey type, average number of each prey type retrieved per dive, and mean prey size in GLBA (Bodkin et al. 2006).

Prey Type	Average Number Retrieved per Dive	Mean Prey Size (mm)
Clams	1.8	65.0
Crabs	1.1	71.8
Mussels	3.5	85.2
Urchins	4.6	44.9

Weitzman (2013) sampled intertidal prey communities at 45 sites across Glacier Bay prior to and after sea otter recolonization. Sites were classified based on sea otter density 10-15 years after colonization: low (~0.1-2 otters/km^2), medium (~2.5-20 otters/km^2), high (>25 otters/km^2), and control (no otters present). Weitzman (2013, p. 26) concluded that "sea otters significantly influenced the structure of intertidal prey communities in Glacier Bay but the effect was variable through space and across species." For example, mean butter clam size decreased at all otter densities, while *Mya* species decreased in size only at high otter densities but increased at low otter densities (Table 44; Weitzman 2013). Green sea urchin mean size also decreased at all intensity levels between 1998 and 2010 (Weitzman 2013).

Table 44. Size characteristics of four common sea otter prey species sampled at intertidal study sites throughout Glacier Bay prior to sea otter colonization (1998) and after re-establishment (2010) (Weitzman 2013).

Species	Pre-colonization (1998)				Post-colonization (2010)			
Density[A]	N[C]	Range (mm)	Mean (mm)	St. Dev.	N	Range (mm)	Mean (mm)	St. Dev.
Butter clam								
Low	105	20-105	78.5	17.7	137	15-114	70.7	21.6
Medium	486	15-113	72.6	20.2	335	14-122	67.5	22.3
High	347	14-100	62.5	15.4	162	14-84	43.9	15.5
Control[B]	4	57-74	64.8	8.1	5	53-73	64.8	9.5
***Mya* spp. (clams)**								
Low	350	14-70	38.6	12.0	533	15-82	44.2	15.4
Medium	438	14-70	40.1	12.5	436	15-73	37.7	12.6
High	94	14-55	33.1	9.1	108	14-67	27.7	11.7
Control	14	15-39	22.1	7.6	5	23-49	33.4	11.5

[A] Sea otter density

[B] Areas where otters were absent for the entire study period

[C] N = sample size (number of individuals observed and measured)

Table 44 (continued). Size characteristics of four common sea otter prey species sampled at intertidal study sites throughout Glacier Bay prior to sea otter colonization (1998) and after re-establishment (2010) (Weitzman 2013).

Species	Pre-colonization (1998)				Post-colonization (2010)			
Density[A]	N[C]	Range (mm)	Mean (mm)	St. Dev.	N	Range (mm)	Mean (mm)	St. Dev.
Pacific littleneck clam								
Low	494	14-69	42.9	14.0	67	14-65	44.0	12.1
Medium	1,373	14-102	39.2	12.6	125	14-64	42.7	12.5
High	362	14-59	33.1	10.6	25	15-52	33.1	11.3
Control	22	25-57	42.1	8.9	1	–	40.0	–
Green sea urchin								
Low	349	11-46	17.7	5.1	2,015	11-45	15.7	5.1
Medium	1,048	11-43	18.6	6.3	1,905	11-43	16.0	5.2
High	319	11-38	16.3	5.1	423	11-33	14.5	3.6
Control	14	11-53	19.0	10.3	16	11-21	14.4	2.7

[A] Sea otter density

[B] Areas where otters were absent for the entire study period

[C] N = sample size (number of individuals observed and measured)

Threats and Stressor Factors

GLBA park staff identified several potential threats and stressors to sea otters in and around the park. Anthropogenic threats include chronic or catastrophic oil pollution, legal harvest occurring outside of GLBA, illegal harvest inside of GLBA, and vessel disturbance and other human activities. Natural threats include predation, disease, and intra-specific competition.

Anthropogenic Threats

Sea otters are highly susceptible to chronic or catastrophic oil pollution. The EVOS in Prince William Sound, Alaska, was the most catastrophic oil spill to affect sea otters. The spill occurred in 1989, and adversely affected many animals, including sea otters. According to Bodkin et al. (2002c), the effects of the 1989 spill were still apparent in the region’s sea otter population in 2000, which suggests that these animals were still recovering over a decade later. Continued monitoring showed that recovery of the region’s population size was ongoing through 2009 (Bodkin et al. 2011) but was complete by 2013, 24 years after the spill (Ballachey et al. 2014).

Legal harvest that occurs outside the park is another potential threat to sea otters in southeast Alaska. The Pacific maritime fur trade was responsible for the depletion of the sea otter population throughout Alaska in the early 1900s (Kenyon 1969, Estes et al. 2009). The state had to reintroduce 400 sea otters back to the region, to encourage the reestablishment of the population (Jameson et al. 1982). Legal harvest is limited to Alaska natives, and each kill must be reported. Legal harvest seems to increase with the increase in sea otter populations. The total harvest in southeast Alaska has increased since 2009, and Table 45 displays the total legal harvest records for southeast Alaska between 1989 and 2013.

Table 45. Reported subsistence harvest of sea otters by village in southeastern Alaska between 1989 and 2013 (USFWS 2013b).

Village	1989-2008	2009	2010	2011	2012	2013	Total
Angoon	101	0	0	0	0		101
Coffman Cove	0	0	0	21	0	0	21
Craig	656	44	136	98	63	71	1,068
Gustavus*	0	0	0	0	0	9	9
Hoonah*	607	27	53	28	127	89	931
Hydaburg	245	0	0	20	14	22	301
Juneau*	334	33	35	3	7	22	434
Kake	106	2	17	21	157	53	356
Kasaan	21	0	0	0	0	0	21
Ketchikan	537	0	11	14	0	4	566
Klawock	922	199	58	49	60	71	1,359
Naukati Bay	0	0	0	3	64	57	124
Pelican*	197	3	4	21	2	0	227
Petersburg	85	0	13	91	49	57	295
Sitka*	2,154	141	218	201	285	406	3,405
Wrangell	67	18	9	0	12	9	115
Yakutat*	402	117	89	149	112	138	1,007
Total	**6,434**	**584**	**643**	**719**	**952**	**1,008**	**10,340**

* Villages nearest to GLBA

Illegal harvest occurring in the park is a potential threat to sea otters in GLBA. The Marine Mammal Protection Act (MMPA) currently prohibits commercial killing and selling of otter pelts. Subsistence hunting by Alaska Natives is legal throughout Southeast Alaska, and there are limited estimates on illegal take.

Disturbance by vessels and other human activities are another threat to sea otters in GLBA. Entanglement/incidental drowning in fishing gear, vessel strikes, and oil spills from vessels are significant threats related to vessel disturbance and human activity (Riedman and Estes 1990, SORT 2007). Trauma from boat collisions and lacerations from propellers can result in mortality or increased vulnerability due to injury (Riedman and Estes 1990). Oil spills are also a possibility in GLBA during the summer as a result of higher vessel use.

Natural Stressors

Predation is a natural source of mortality among sea otter populations. Killer whales have been considered a major cause of sea otter declines in southwestern Alaska (Estes et al. 2009). Although killer whales occur in GLBA, they have not yet been observed preying on otters (Matkin et al. 2007). Matkin et al. (2007) documented four young whales harassing an otter in Glacier Bay on one occasion and hypothesized that the harassment was a kind of “target practice” rather than an effort to obtain food. Other predators of sea otters include sharks, bald eagles, and bears (Kenyon 1969, Bodkin et al. 2000).

Although disease has not been confirmed as a cause of mortality in GLBA, it is still considered a potential threat to the sea otter population. In 1996, a study was conducted to assess the risk of disease on sea otters in California. Approximately 38.6% of sea otter mortality was caused by various diseases (e.g., parasitic, fungal, bacterial) (Thomas and Cole 1996). Some of the diseases were caused by ingesting contaminated prey or from vessel-induced trauma (Thomas and Cole 1996). Contamination of shorelines and vessel traffic could create conditions making sea otters more susceptible to disease in GLBA. Phocine distemper virus, which has caused high-mortality epidemics in North Atlantic seal populations, has been detected in sea otters in Kachemak Bay (near Homer, AK) in southcentral Alaska (Goldstein et al. 2009).

Intra-specific competition is a stressor for sea otters in GLBA. According to Riedman and Estes (1980) food stealing can be common among female otters and their pups. The older a pup becomes the more regularly they may steal prey from mothers during bouts. Food stealing also occurs between territorial males and between males and females (single or with pup) (Riedman and Estes (1980). Males steal food from females and pups foraging in their territory. It is not uncommon for a male to steal prey more than once during a foraging bout. According to Riedman and Estes (1980) males will even steal from their bonded mate. Intra-specific competition may become more of a stressor to sea otters in GLBA if otter numbers continues to increase, as more individuals would be competing for the same resources.

<u>Data Needs/Gaps</u>

Some long-term monitoring data for the GLBA sea otter population are available, including distribution and abundance estimates. Data regarding foraging effort, prey diversity, and prey abundance and size also exist but are not extensive. There are no available data on morphometrics and limited information on kelp abundance in the park. The collection of morphometric data may enable managers to assess body condition of sea otters in the park. Establishment of kelp distribution and abundance monitoring may enable managers to distinguish areas being utilized by sea otters, which could aid in focusing future studies to those areas in the park. Continuing sea otter surveys and studies of intertidal benthic communities will provide valuable information for the management of park resources. Sea otter prey species abundance and size distribution could be re-sampled using previously utilized methods and locations (e.g., Estes and Duggins 1995, Bodkin and Kloecker 1999, Donnellan et al. 2002) to evaluate changes in the intertidal community over community.

<u>Overall Condition</u>

Trends in Distribution and Abundance

The project team defined the *Significance Level* for trends in sea otter distribution and abundance as a 3. According to Bodkin et al. (1999, 2002a, 2006) sea otters were mainly observed in the Lower Bay but seemed to be spreading out. Weitzman (2010) noted that sea otters were observed in the Lower Bay as well as locations further north (Lone Island, Sturgess Islands) than previously observed. This northerly expansion continued into the lower east and west arms of Glacier Bay, as documented in the 2012 survey (Esslinger et al. 2013). Gabriele et al. (2012) documented a large number of sightings in the nearshore region, particularly in the Lower Bay near Flapjack Island, Idaho Inlet, and Berg Bay. Vessel surveys have also observed sea otters expanding their range into the mid-channel and the West and East Arms of Glacier Bay; sightings have occurred as far north as Tidal Inlet (Gabriele et al. 2012). In 1995, the observed sea otter abundance in GLBA was five sea otters, and by 2012 abundance estimates had increased to over 8,000 sea otters (Esslinger et al. 2013; Table 36). As a result, the *Condition Level* for this measure was assigned a 0, or no concern.

Foraging Effort

The project team defined the *Significance Level* for sea otter foraging effort, prey diversity, prey size, and morphometrics as a 3. Bodkin et al. (2006) documented 1,232 foraging events (93% success rate) at GLBA in 2004. A variety of prey items were identified, as 36 prey items were consumed by sea otters in GLBA between 1993 and 2004. Clams, mussels, scallops, gastropods, mollusks, sea stars, urchins, and crustaceans were all preyed upon during this study. The prey classes with the highest species richness were clams and crustaceans, with 10 species and seven species being consumed, respectively. Weitzman (2013) documented over 14,580 foraging bouts between 1993 and 2011, with over 6,759 successful (46%) foraging bouts. Weitzman (2013) documented 46 species of prey. Similar to Bodkin et al. (2006), clams, mussels, scallops, gastropods, mollusks, sea stars, urchins, and crustaceans were all preyed upon. Clams, urchins, and mussels were the most common prey items, comprising approximately 56%, 18%, and 15% of sea otter diets, respectively. Weitzman (2013) observed declines in energy recovery rates in sea otter and intertidal communities, which suggests that otters will have to increase their foraging effort. Since the observed condition and changes in foraging effort are thought to result from the natural process of sea otters reaching carrying capacity, the *Condition Level* for this measure was assigned a 0, or of no concern.

Kelp Abundance, Invertebrate Prey Abundance and Size

The project team defined the *Significance Level* for kelp abundance, invertebrate prey abundance and size as a 2. Kelp abundance increased dramatically in Torch Bay following sea otter colonization (Estes and Duggins 1995). Kelp was present at a majority of study sites in lower Glacier Bay (23 of 31 sites) in the early 2000s and was increasing at some sites (Donnellan et al. 2002). In surveys of prey selection, Bodkin et al. (2006) retrieved urchins and mussels from sea otters more on average per dive than any other prey type. Mussels were the largest prey type collected with a mean size of 85.2 mm (3.4 in). Crabs and clams had mean sizes of 65.0 mm (2.6 in) and 71.8 mm (2.8 in), respectively. Urchins were the smallest prey type (44.9 mm [1.8 in]) (Bodkin et al. 2006). Based on intertidal community surveys between 1998 and 2010, Weitzman (2013) concluded that sea otters significantly influence intertidal prey community structure in Glacier Bay, but that the effects vary between species. While mean butter clam and green sea urchin sizes decreased at all otter densities, *Mya* species decreased in size only at high otter densities and increased at low otter densities (Weitzman 2013). This influence was also seen in Torch Bay, where sea urchins were common prior to sea otter recolonization (1978) but were nearly absent shortly after recolonization (1988) (Estes and Duggins 1995). Although some species may decline in the presence of sea otters, the overall effect may be more diverse and productive intertidal communities. Since the observed condition and changes in kelp abundance and invertebrate prey are thought to result from the natural process of sea otter colonization, the *Condition Level* for this measure is a 0, or of no concern.

Strandings

The project team defined the *Significance Level* for sea otter standings as a 1. Measures with a *Significance Level* of 1 are not discussed in depth in the current condition section of this assessment, but available information is summarized here in the overall condition section. NOAA (2014) compiled marine mammal stranding data in southeast Alaska. Stranding data could alert managers to emerging health and disease threats to sea otters in GLBA. Carcasses could also be used to document distribution of age-classes of dead otters based on determination of age. Then, one could look for deviations from expected patterns in the event of catastrophic events (e.g., oil spills) (Esler, written communication, 13 June 2014).

NOAA (2014) documented 42 sea otter strandings in the GLBA and Gustavus areas between 2006 and 2013 (Appendix I). Twenty-six of those strandings had GPS coordinates (Figure 102). The highest number of strandings were recorded in 2010 (13 strandings; Figure 103). There were 26 reports with an unknown cause of death, four records that described pulmonary congestion/edema as the cause of death, three records of trauma, and three records of gunshot wounds. Other causes of death documented included acute heart failure, orca predation, septicemia, and starvation (Figure 104; NOAA 2014). This may be a result of the increasing sea otter population or increased threats present in the park. A *Condition Level* of 1, or of low concern, was assigned to this measure.

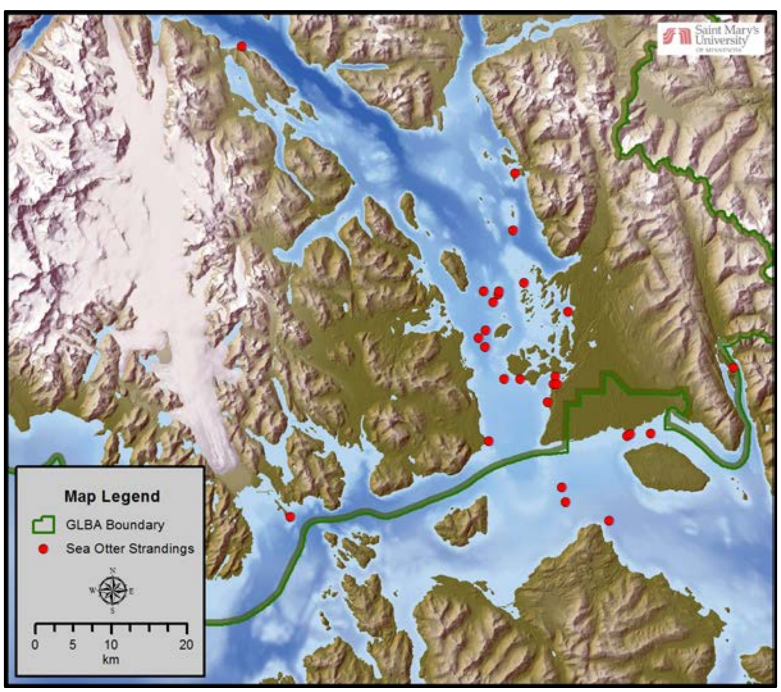

Figure 102. Sea otter strandings in GLBA and surrounding areas between 2006 and 2013 (NOAA 2014).

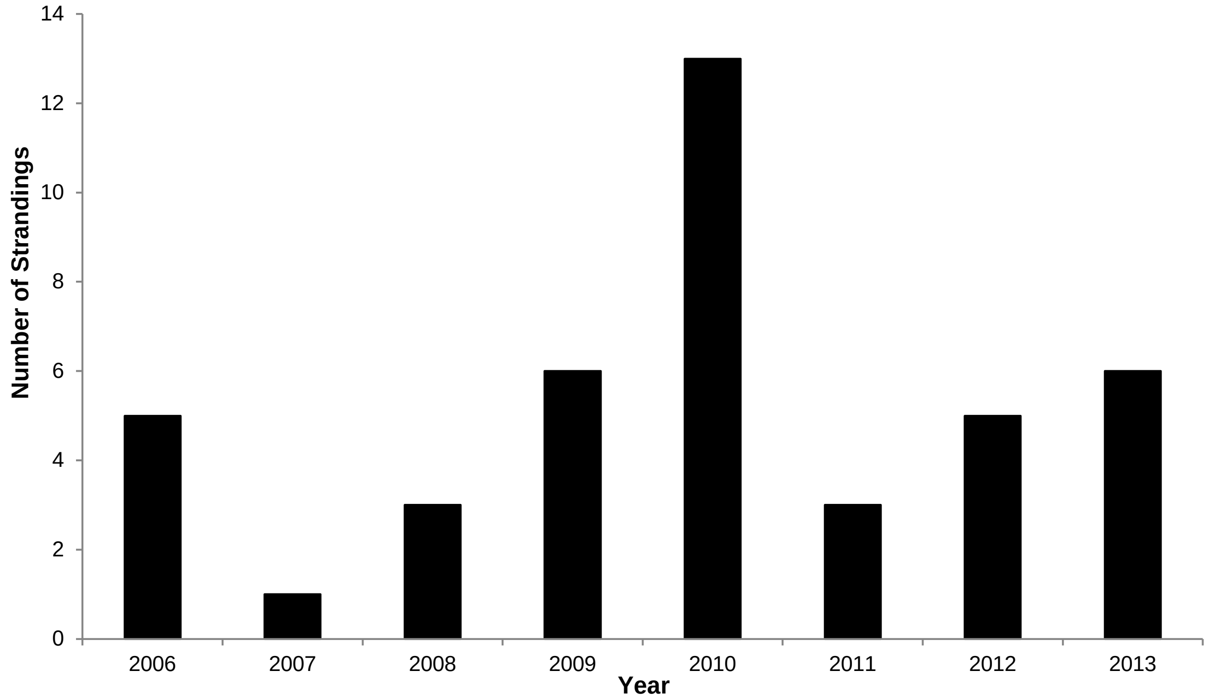

Figure 103. Number of reported sea otter strandings in GLBA and surrounding areas between 2006 and 2013 (NOAA 2014).

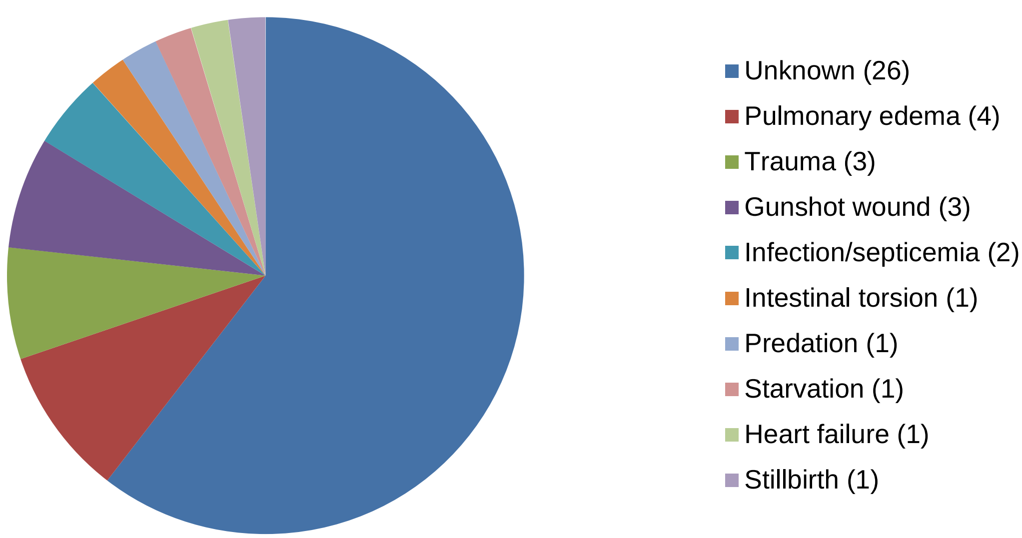

Figure 104. Incidence of various causes of death among reported sea otter strandings (NOAA 2014).

<u>Weighted Condition Score</u>

The *Weighted Condition Score* for sea otters in GLBA is 0.30, indicating that the component is in good condition. Sea otters have rapidly colonized Glacier Bay, and sea otter abundance has greatly increased since their return to the park. Studies on foraging effort and energy recovery rate suggest an increase in time foraging and prey diversity, and a decreasing trend in prey size, which all indicate that the population may be approaching carrying capacity. The available data seem to display an increasing trend in GLBA sea otter abundance as of 2012. More recent data are needed to assess a more current trend.

Sea Otters			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.04**
Distribution, Abundance, and Trends	3	0	
Foraging Effort	3	0	
Kelp Abundance, and Invertebrate Prey Abundance and Size	2	0	
Strandings	1	1	

4.13.6. Sources of Expertise

- Dan Esler, USGS Alaska Science Center Research Wildlife Biologist
- Jim Bodkin, USGS Research Wildlife Biologist
- Jamie Womble, NPS Wildlife Biologist

4.13.7. Literature Cited

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4.14. Harbor Porpoise

4.14.1. Description

The harbor porpoise is both a true porpoise (genus *Phocoena*) and a whale (order Cetacea), more specifically the smallest oceanic whale (Dawson and Taylor 1979). Along North America's Pacific coast, the species ranges from Point Barrow in northern Alaska to central California (Allen and Angliss 2012). The harbor porpoise is considered common in GLBA; they are most common in bays, estuaries, and tidal channels where the waters are more sheltered than open waters (Dawson and Taylor 1979, NPS 2014). The harbor porpoise is of particular interest to GLBA because the area is a regional hot spot for harbor porpoise, and opportunistic sightings indicate population fluctuations (Dahlheim et al. 2012; Gabriele, written communication, 22 May 2015). The species is also a marine predator, which is a Vital Sign in SEAN (Photo 36; Moynahan et al. 2008). However, harbor porpoises are a "shy" species and individuals are not easy to identify, making it difficult to study populations and to accurately assess their condition (Dawson and Taylor 1979; Gabrielle, written communication, 22 May 2015).

Photo 36. Harbor porpoise (NPS photo).

Harbor porpoises are opportunistic feeders, which can be a benefit with the competition for prey in GLBA waters. The harbor porpoise competes with harbor seals, humpback whales, and Steller sea lions. Harbor porpoise diet includes herring (Clupeidae), capelin, sand lance, pollock (*Pollachius* spp.), flatfish (order Pleuronectiformes), and euphausiids (order Euphausiacea) (Gabriele and Lewis 2000). Other prey species include market squid (*Loligo opalescens*), northern anchovy (*Engraulis mordax*), cusk eel (*Chilara taylori*), and rockfishes (*Sebastes* spp.) (Gabriele and Lewis 2000).

Harbor porpoises are sexually mature at 3-4 years and females typically give birth every 2 years (Schmale 2008). Individual life span is approximately 8-10 years, although some porpoises have lived up to 20 years. Harbor porpoises are normally seen alone or in pairs but may form small groups (<10 individuals) (Schmale 2008).

4.14.2. Measures

- Distribution on NOAA sighting tracklines
- Distribution on MARMAM database GIS data
- Density and abundance from NOAA sighting tracklines
- Sightings per year in MARMAM dataset
- Strandings per year in GLBA area

4.14.3. Reference Conditions/Values

The reference condition for harbor porpoise populations in GLBA can be found in several documents. Porpoise distribution in GLBA is described in Dahlheim and Waite (2006) and Dahlheim et al. (2012). Harbor porpoise distribution has been considered clumped, with a majority of sightings occurring in the lower bay and entrance to the bay, but harbor porpoises have also been seen utilizing areas in the West Arm and East Arm. Porpoise density and abundance in the earliest GLBA surveys (1991-1993) are described in Dahlheim et al. (2012). Density and abundance has been relatively high in GLBA versus other areas in Southeastern Alaska. Harbor porpoise sightings from 1994 to 2010 are described in Gabriele et al. (2011), and ranged from 115-229 sightings per year. Historic strandings, as reported by NOAA (2013), have ranged from zero to two strandings per year.

4.14.4. Data and Methods

Dawson and Taylor (1979) conducted a preliminary study of harbor porpoise in GLBA. The study was intended to explore possible methods and study sites for future survey efforts and did not provide focused or comprehensive information on distribution and abundance. Prather et al. (1989) conducted a baseline survey of harbor porpoise in Glacier Bay in 1989. Five transects were located throughout Glacier Bay (West Arm, East Arm, Main Channel) and occurred from May through August. Transect data includes the number of porpoises sighted, and grouping.

Gabriele and Lewis (2000) provided a summary of opportunistic marine mammal sightings in Glacier Bay between 1994 and 1999. Mammal sightings were reported by NPS biologists during whale monitoring surveys, which occurred from June to August of each year. The main area surveyed fell between Garforth Island, Geikie Inlet, Point Carolus, and Bartlett Cove (Figure 105). This area was surveyed three to four times a week. Areas as far north as Russell Island and Adams Inlet were surveyed intermittently. Most sightings took place within 100 m (328 ft) of the vessel to ensure proper identification. A GPS unit was used to record locations of porpoise sightings. Gabriele et al. (2011) and NPS (2014) provide progress reports on these sightings through 2010 and 2013, respectively.

Figure 105. Opportunistic marine mammal survey areas in GLBA (Gabriele and Lewis 2000; Gabriele et al. 2011, MARMAM 2014).

Bodkin et al. (2002) conducted an annual survey for marine predators in GLBA between 1999 and 2001. Transects were run along the coast and in open water areas throughout the bay in June of each year of the survey. In 2001, three vessels were used to survey in March and June. Sightings within 300 m (984 ft) in front of the vessel, and up to 150 m (492 ft) to the side of the vessel were recorded. Records include taxa, count, and activity. GPS units were used to plot transects and locations of predator observations.

Dahlheim and Waite (2006) documented the distribution of harbor porpoise in GLBA between 1991 and 2006. A total of 14 days and 463 surveying hours were spent in GLBA. Marine mammal surveys were conducted by NMML staff on a NOAA vessel. Surveys were conducted from spring through fall to document seasonal occurrence. It should be noted that GPS locations of harbor porpoise in GLBA are not true positions but locations of the vessel along the tracklines when porpoise sightings occurred. Dahlheim et al. (2012) documented the distribution, abundance, and trends of harbor porpoise in GLBA and Icy Strait between 2006 and 2010 using the same methods and tracklines.

NOAA (2013) compiled stranding records of marine mammals in GLBA and surrounding waters from 1988-2012. Strandings have been reported by the general public as well as park staff. Stranding reports include species, date, location, photos, status of animal (e.g., injured, sick, dead), and causation.

4.14.5. Current Condition and Trend

<u>Distribution on NOAA Sighting Tracklines</u>

Dahlheim and Waite (2006) documented harbor porpoise distribution in GLBA between 1991 and 2006. Harbor porpoise were sighted throughout the bay from the entrance to the tide-water glaciers in the West and East Arms. Dahlheim et al. (2012) continued documenting harbor porpoise distribution in GLBA between 2006 and 2010. Harbor porpoise were again distributed throughout the bay; however, the distribution is clumped toward the lower bay and entrance to the bay. Figure 106 displays the harbor porpoise sightings by tracklines in GLBA between 1991 and 2010.

Figure 106. Harbor porpoise sightings by tracklines in GLBA and surrounding areas between 1991 and 2010 (Dahlheim and Waite 2006, Dahlheim et al. 2012).

<u>Distribution on MARMAM Database GIS Data</u>

NPS (2014) compiled harbor porpoise sighting data for GLBA between 1993 and 2013. A large number of sightings have occurred in the lower bay and entrance to the bay, particularly nearshore along the Sitakaday Narrows, Fingers Bay, and Bartlett Cove. Harbor porpoise have been seen in the East Arm in Muir Inlet and Adams Inlet. Porpoise have also been spotted in the West Arm in Reid Inlet and near Russell Island. Figure 107 displays the porpoise sightings in GLBA from 1993 to 2013.

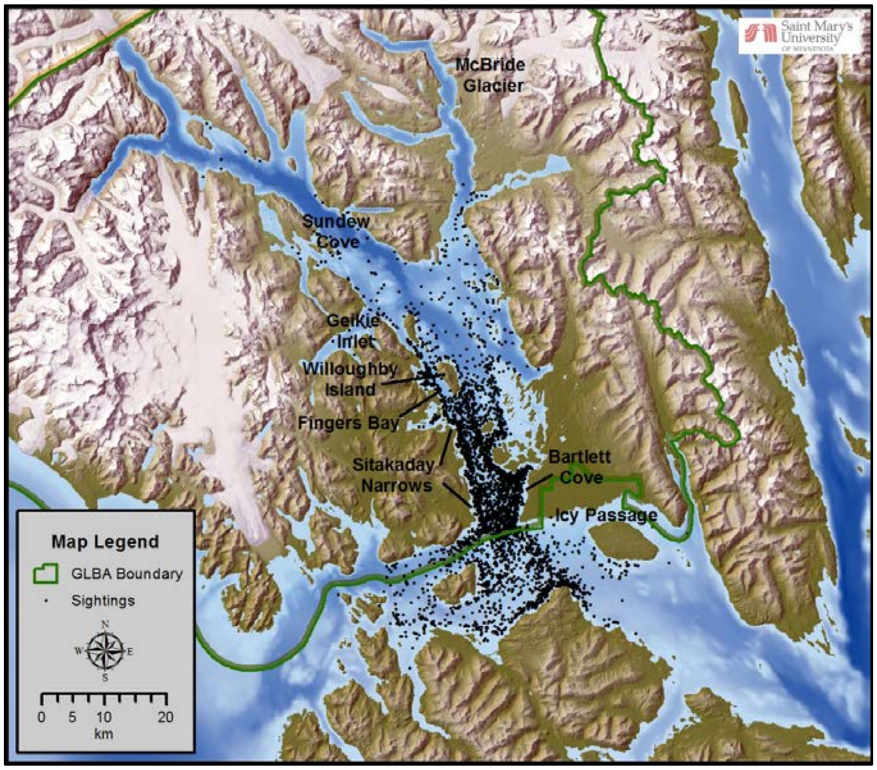


Figure 107. Harbor porpoise sightings during opportunistic marine mammal surveys in GLBA and surrounding areas from 1993 to 2013 (NPS 2014).

<u>Density and Abundance from NOAA Sighting Tracklines</u>

Dahlheim et al. (2012) documented harbor porpoise density and abundance in GLBA and Icy Strait for 6 years. The lowest density and abundance values both occurred in 2006, and the highest density and abundance both occurred in 2010 (Table 46). Although densities were high in some parts of GLBA, the population trend was estimated to have decreased slightly (-0.9%) over the surveying period. Table 46 displays density and abundance estimates of harbor porpoise in GLBA and surrounding areas between 1991 and 2010. Unfortunately, NOAA tracklines were not sampled during the time period that NPS surveys first detected a decline in abundance (1997-1998) (Gabriele, written communication, 22 May 2015).

Table 46. Harbor porpoise density (D), abundance estimates (N), and coeffient of variation (CV) from 1991 to 2010 in GLBA and surrounding areas (Dahlheim et al. 2012).

	Density	Abundance	
Year	D	N	CV
1991	0.163	376	0.24
1992	0.148	342	0.25
1993	0.163	375	0.19
2006	0.083	191	0.23

Table 46 (continued). Harbor porpoise density (D), abundance estimates (N), and coeffient of variation (CV) from 1991 to 2010 in GLBA and surrounding areas (Dahlheim et al. 2012).

	Density	Abundance	
Year	D	N	CV
2007	0.189	304	0.25
2010	0.222	512	0.25

<u>Sightings Per Year in MARMAM Dataset</u>

Gabriele and Lewis (2000) documented 1,157 individual harbor porpoises with 648 sightings between 1996 and 1999. Harbor porpoise numbers decreased by approximately 52% within that time period. The lowest number of porpoise sightings occurred in 1998 with 137 individuals. Most observations occurred within the Sitakaday Narrows and in the 32 m (105 ft) shoal of Willoughby Island (Figure 107). According to Gabriele and Lewis (2000), these areas have strong tidal currents, which is characteristic of higher porpoise prey abundance. Porpoise sightings decreased greatly between 1997 and 1998 and remained low in 1999. Table 47 displays the number of individual harbor porpoises (and number of sightings) observed in GLBA and surrounding waters between 1996 and 1999.

Table 47. The number sightings and individuals of harbor porpoises observed in GLBA and surrounding waters between 1996 and 1999 (Gabriele and Lewis 2000).

Year	Number of Sightings	Number of Individuals
1996	218	378
1997	226	359
1998	92	137
1999	112	183

Gabriele et al. (2011) documented 2,674 individual porpoises in 1,728 sightings between 2000 and 2010. The number of sightings peaked in 2000, then dropped in 2001 and remained relatively stable between 2003 and 2010. Table 48 displays the number of individual harbor porpoises (and number of sightings) observed in GLBA and surrounding waters between 2000 and 2010.

Table 48. The number sightings and individuals of harbor porpoises observed in GLBA and surrounding waters between 2000 and 2010 (Gabriele et al. 2011).

Year	Number of Sightings	Number of Individuals
2000	229	386
2001	131	192
2002	115	239
2003	125	178

Table 48 (continued). The number sightings and individuals of harbor porpoises observed in GLBA and surrounding waters between 2000 and 2010 (Gabriele et al. 2011).

Year	Number of Sightings	Number of Individuals
2004	152	235
2005	157	237
2006	156	241
2007	178	240
2008	164	244
2009	152	227
2010	169	255

<u>Strandings per Year in GLBA Area</u>

NOAA (2013) documented 11 harbor porpoise strandings in GLBA between 1988 and 2012 (Figure 108, Table 49). This averages out to a rate of approximately 0.4 strandings/year over the 25-year period. However, over the most recent 5-year period (2008-2012), the average number of strandings was 1.2/year (Table 49). The greatest number of strandings documented per year was two, occuring in 3 separate years (2005, 2009, 2012) (NOAA 2013). The most recent strandings were documented in Sundew Cove and near McBride Glacier in 2012.

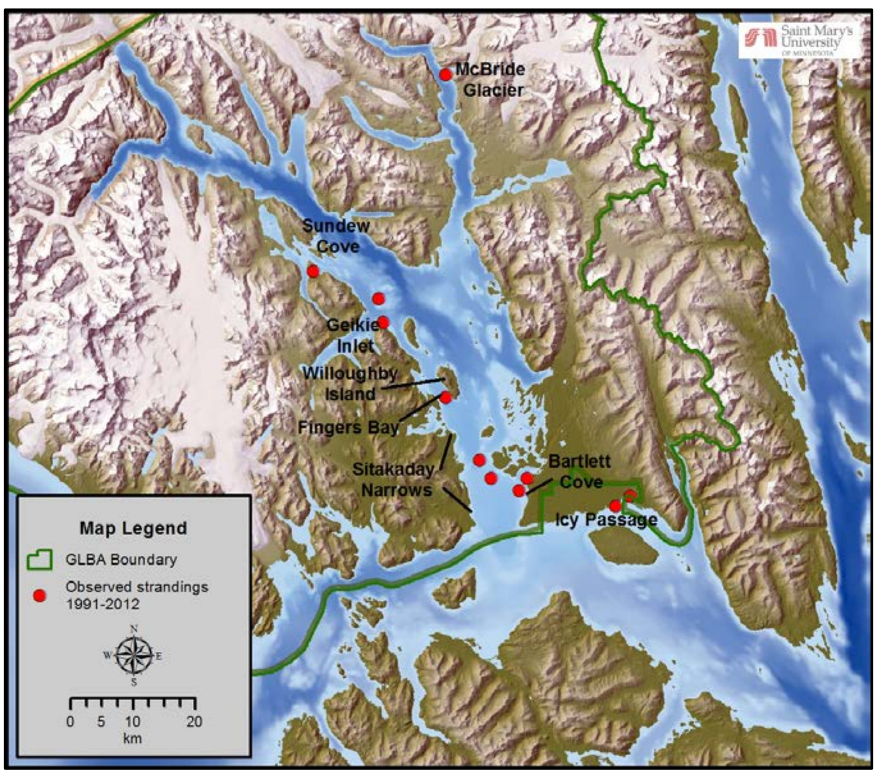


Figure 108. Harbor porpoise strandings documented in GLBA and the surrounding areas between 1991 and 2012 (NOAA 2013).

Table 49. The number of strandings documented in GLBA and surrounding waters between 1988 and 2012 (NOAA 2013).

Year	Number of Strandings
1988-1990	0
1991	1
1992	0
1993	1
1994-1997	0
1998	1
1999-2004	0
2005	2
2006-2007	0
2008	1
2009	2
2010	0
2011	1
2012	2
Total	**11**

Threats and Stressor Factors

Harbor porpoise are threatened by both natural and anthropogenic factors. GLBA staff have indicated that natural threats include predation by killer whales, disease, viruses, and climate change, while anthropogenic threats include pollutants, incidental take/mortality as bycatch in fishing gear outside the park, and human disturbance (mainly vessels). According to Dawson and Taylor (1979), population declines may be heavily influenced by anthropogenic threats such as pollutants (specifically PCBs), fishing net entanglement, increased motor boat traffic, and decline in food abundance (due to competition with fisheries).

Anthropogenic Threats

Pollutants are a potential threat to harbor porpoise in and near GLBA. Harbor porpoise, as well as other cetaceans, can accumulate high levels of pollutants in their blubber layers (Calambokidis and Barlow 1991). Pollutants can also be ingested in higher amounts due to bioaccumulation, since these animals are higher on the food chain. Pollutants also seem to be a factor in restricting harbor porpoise movements and lowering genetic mixing rates with other populations (Allen and Angliss 2012). Pollutants, such as PCBs and mercury, may cause decreased or weakened immune systems in harbor porpoise, making them more susceptible to infectious diseases (Jepson et al. 1999; Bennett et al. 2001).

Incidental take and mortality as bycatch in fishing gear are a threat to harbor porpoises throughout their range, including in southeast Alaska (Dawson and Taylor 1979, Allen et al. 2014). Commercial fishing is limited within GLBA, so the threat to porpoises is lower while inside the park, but increases if they leave park waters. According to Allen and Angliss (2012), incidental mortality estimates from Yakutat salmon set gillnets were approximately 16 and 27 porpoises in 2007 and 2008, respectively. There were three other porpoise mortalities in the Yakutat area in 2008 that were reportedly caused by bycatch and entanglement. Two harbor porpoise mortalities were reported in the Southeast stock in 2009 and one in 2010 from entanglement in gillnet gear (Allen et al. 2014). Vessels and human disturbance are also potential anthropogenic threats to the harbor porpoise in GLBA.

Natural Threats

Predation by orcas is a threat to harbor porpoise in GLBA. Orcas are known to kill and harass as many as 20 cetacean species, including harbor porpoises (Jefferson et al. 1991). The harbor porpoise is one of the most common cetaceans attacked by transient orcas. According to Matkin et al. (2007), transient orcas in the Glacier Bay/Icy Strait region preyed upon harbor porpoise in 23% of observed kills.

Diseases and viruses are also threats to harbor porpoise in GLBA. Physical trauma to porpoises from entanglement or by-catch can lead to parasitic or bacterial infections. Infections that are known to occur and cause mortality include parasitic pneumonia, which was more commonly associated with lungworms (*Pseudalius inflexus* and *Torynurus convolutes*) (Jepson et al. 1999). *Streptococcus canis* and *Salmonella* spp. were common bacterial species that caused infection and porpoise mortality (Jepson et al. 1999). This study suggested that bioaccumulation of pollutants such as PCBs may have been a factor in the weakening of porpoise immune systems prior to infection. Fungal and viral pathogens have also been associated with harbor porpoise mortality after the porpoise experienced physical trauma (Bennett et al. 2001).

Climate change and ocean acidification, as they affect habitat or prey availability, are threats to harbor porpoises in GLBA. Harbor porpoises consume a variety of fish species, but some species may be more prominent in their diet than others (e.g., sand lance). In a study conducted in the Scottish North Sea, there were increasing percentages of harbor porpoise mortality due to starvation between 2002 and 2003 than between 1993 and 2001 (MacLeod et al. 2007). Oceanic changes as a result of climate change have had a negative effect on sand lance populations, which historically made up a majority of the porpoise diet (MacLeod et al. 2007). If the change in oceanic conditions are negatively affecting prey species in the Atlantic Ocean, it is possible that prey species in the Pacific Ocean have been (or will be) affected as well, causing physical stress to harbor porpoise in GLBA.

Data Needs/Gaps

There are long-term data on the distribution and abundance of harbor porpoise in GLBA dating back to 1991; however, these surveys did not occur every year. GPS sighting data from Dahlheim and Waite (2006) and Dahlheim et al. (2012) were taken from the vessel and do not represent actual positions of animals within the bay. More accurate GPS locations may display more specific locations and preferred habitat. Data on strandings date back to 1988 and continue to be reported, but some strandings in remote areas of Alaska likely go undocumented for a variety of reasons.

Continued yearly monitoring efforts will aid park staff on managing the GLBA harbor porpoise population.

<u>Overall Condition</u>

Distribution on NOAA Sighting Tracklines

The project team defined the *Significance Level* for distribution on NOAA sighting tracklines as a 1. Dahlheim and Waite (2006) and Dahlheim et al. (2012) documented harbor porpoise distribution in GLBA between 1991 and 2010. Harbor porpoise were sighted throughout the bay from the entrance of the bay to the tide-water glaciers in the West and East Arms. Harbor porpoise distribution was clumped toward the lower bay and entrance to the bay. The distribution patterns observed on the tracklines appear to be what is expected in this area, and no major deviations have been observed. This measure was assigned a *Condition Level* of 0, or of no concern.

Distribution on MARMAM Database GIS Data

The project team defined the *Significance Level* for distribution on MARMAM database GIS data as a 2. Gabriele and Lewis (2000) and Gabriele et al. (2011) documented porpoise locations for more than 15 years. A large number of sightings occurred in the lower bay and entrance to the bay, more specifically, nearshore along the Sitakaday Narrows, Fingers Bay, and Bartlett Cove. Harbor porpoise have been seen in the East Arm in Muir Inlet and Adams Inlet. Porpoise have also been spotted in the West Arm in Reid Inlet and near Russell Island. Due to the consistent sightings of porpoise in both high use areas of the park (e.g., Sitakaday Narrows, Fingers Bay) and in the East and West Arms, this measure was assigned a *Condition Level* of 0, or of no concern.

Density and Abundance from NOAA Sighting Tracklines

The project team defined the *Significance Level* for density and abundance from NOAA sighting tracklines as a 2. Dahlheim et al. (2012) documented lower density and abundance in 2006, and higher density and abundance in 2010. High densities in areas of GLBA occurred throughout the surveying period. Although densities were high, the population trend was estimated to have decreased slightly (-0.9%) over the surveying period. As a result, this measure was assigned a *Condition Level* of 1, or of low concern.

Sightings per Year in MARMAM Dataset

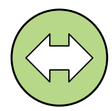
The project team defined the *Significance Level* for sightings per year in the MARMAM dataset as a 2. Gabriele et al. (2011) reported that the annual number of porpoise sightings ranged from 92 (1998) to 229 (2000) sightings. Sightings dropped substantially between 2000 and 2001 (131 sightings), but then the number of sightings remained relatively stable from 2001-2010 (Gabriele et al. 2011). As a result of several years with large reductions in sightings, this measure was assigned a *Condition Level* of 1, or of low concern.

Strandings per Year in GLBA Area

The project team defined the *Significance Level* for strandings as a 1. NOAA (2013) reported 11 harbor porpoise stranding records in GLBA from the time record-keeping began (1988) through 2012. The number of stranding reports per year over the period of record is low (average of 0.4/year) but increased slightly after 2008 (1.2/year from 2008-2012) (NOAA 2013). As a result, this measure was assigned a *Condition Level* of 1, or of low concern.

Weighted Condition Score

The *Weighted Condition Score* for harbor porpoise at GLBA is 0.21, indicating that the component is of low concern. Although there was a slight decreasing population trend, porpoise density is believed to be dynamic, and the combination with the other measures makes it appear that this component is currently stable.

Harbor Porpoise			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.21**
Distribution on NOAA Tracklines	1	0	
Distribution on MARMAM Database	2	0	
Density and Abundance (NOAA)	2	1	
Sightings per Year	2	1	
Strandings per Year	1	1	

4.14.6. Sources of Expertise

- Christine Gabriele, GLBA Wildlife Biologist, Humpback Whale Monitoring Program
- Marilyn Dahlheim, National Marine Mammal Laboratory Wildlife Biologist

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4.15. Steller Sea Lions

4.15.1. Description

The Steller sea lion (Photo 37), the largest member of the eared seal family (Otariidae), ranges along the entire North Pacific rim, from northern Japan, through Alaska, and down to California (Loughlin et al. 1987). The species exhibits sexual dimorphism with males ranging up to 3.2 m (10.5 ft) in length and 1,120 kg (2,469 lbs), while females reach 2.9 m (9.5 ft) and 263 kg (772 lbs) (Loughlin et al. 1987). Males survive about 15 years and females as long as 30 years (Loughlin et al. 1987). Female Steller sea lions reach sexual maturity at 3-6 years old and may continue breeding into their early 20s (Pitcher and Calkins 1981, NMFS 2013). Single pups (twins are rare) are born between late May and early July and undergo an extensive parental care period that may range from 10 months to up to 4 years (Loughlin et al. 1987). Steller sea lions are opportunistic foragers that hunt near shore and in pelagic waters, preying on a variety of fish, invertebrates, and smaller marine mammals (Womble and Sigler 2006, Trites et al. 2007, Womble and Conlon 2010).

Photo 37. Steller sea lions on South Marble Island in GLBA (NPS photo).

In 1990, as a result of significant declines, Steller sea lions were first listed as a threatened species throughout their range under the Endangered Species Act (55 FR 12645). At that time, no subpopulation distinctions were identified (NMFS 2013). In 1997, two DPS of Steller sea lions were recognized: a western and an eastern segment. The eastern segment includes Steller sea lions from Cape Suckling, Alaska, south to California’s Channel Islands, and the western segment includes Steller sea lions west of Cape Suckling (144ºW) (Bickham et al. 1996, Loughlin 1997). Identification of two DPS segments was based on divergent population trends, phenotypic differences, and genetic differences (NMFS 2013). In 1997, the Steller Sea Lion Recovery Team recommended that the western DPS be listed as endangered while the eastern DPS remain listed as threatened (62 FR 24346; U.S. Federal Register 1997). On 25 October 2013, NMFS concluded that the biological (demographic) criterion and Endangered Species Act listing factor recovery criteria for the eastern

DPS had been met and found that the eastern DPS of Steller sea lions could be removed from the list of threatened species under the Endangered Species Act (NMFS 2013, U.S. Federal Register 2013).

Steller sea lions were first documented in the park in the 1960s, at a haulout site near Lituya Bay on the outer coast and swimming in GBP (Streveler 1989, Mathews et al. 2011). Currently there are six known Steller sea lion haulout sites and one rookery (Graves Rocks) within GLBA (Figure 109; Mathews et al. 2011). A rookery is defined as a terrestrial site where at least 50 pups are born annually; haulout sites are primarily used for resting and few or no pups are born there (Bigg 1985, Mathews et al. 2011). Graves Rocks transitioned from a haulout site to a rookery in the late 1990s (Mathews and Dzinich 2001, Gelatt et al. 2007). While most Steller sea lions at GLBA are from the eastern stock, some animals from the western stock have been observed in the park and a few western females have given birth at the Graves Rock rookery (Gelatt et al. 2007, Jemison et al. 2013).

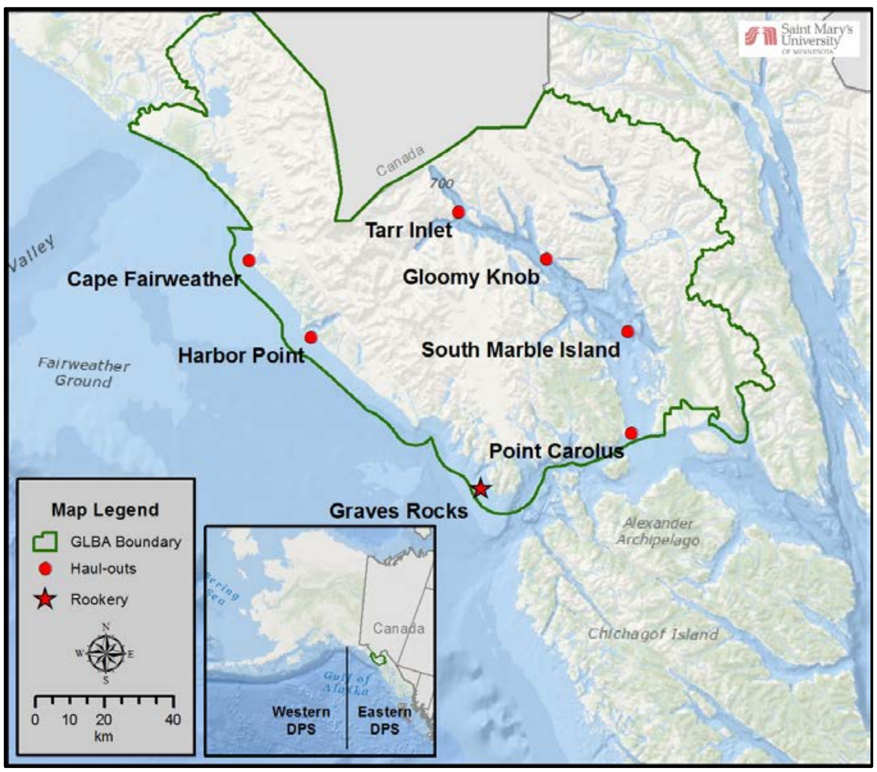

Figure 109. Steller sea lion haulout and rookery sites in GLBA. The inset map shows the dividing line between the eastern and western distinct population segments.

4.15.2. Measures

- Annual estimates of population trends
- Spatial and seasonal distribution at haul outs and rookeries
- Age-specific survival rates

4.15.3. Reference Conditions/Values

For the purpose of this assessment, the reference condition for population trends will be the trend for Southeast Alaska from 1979-2005, calculated by Pitcher et al. (2007) as an annual increase of 3.2%. The reference condition for spatial and seasonal distribution at haul outs and rookeries will be the distribution of Steller sea lions when survey efforts began in the park (early to mid-1970s). At that time, Steller sea lions were only known to haul out at Harbor Point, Cape Fairweather, and Graves Rock (Mathews et al. 2011). Mean age-specific survival rates from four southeastern Alaska rookeries (as reported in Hastings et al. 2011) will be the reference condition for the survival rates measure.

4.15.4. Data and Methods

Steller sea lions have been surveyed at haulout sites in the GLBA area using several different methods since the early to mid-1970s. In 2006, collaboration was initiated between NPS, ADFG, and the University of Alaska Fairbanks (UAF) to compile all historical count data for Steller sea lions in the GLBA region. The project resulted in a long-term trend analysis of Steller sea lions in the GLBA region (Mathews et al. 2011). Mathews et al. (2011) compiled data from aerial, boat, and land-based surveys by the ADFG, UAF, NPS, and the NMFS from 1970-2009 in order to estimate population trends in the GLBA region (see Mathews et al. [2011] for a summary of data sources). Survey sites within GLBA were South Marble Island, Graves Rocks, Harbor Point, Cape Fairweather, Point Carolus, Gloomy Knob, and Tarr Inlet (Figure 109). Aerial surveys have been conducted for all seven sites; boat surveys have been conducted at South Marble Island and Graves Rocks, while land-based surveys are limited to South Marble only (Mathews et al. 2011). The years of data available varied by site, as surveys were rare prior to 1989. Generalized linear models were used to estimate population trends for the three GLBA sites. The remaining park sites did not yield enough survey counts for trend analysis. Mathews et al. (2011) also studied seasonal distribution patterns and documented the establishment of a new Steller sea lion haulout in GLBA (Gloomy Knob) as well as the transition of Graves Rocks from a haulout site to a rookery in the late 1990s.

Womble et al. (2005, 2009) conducted monthly aerial counts of Steller sea lions at 28 sites in Southeast Alaska from 2001-2004. The objectives of the study were to quantify seasonal distribution patterns and to determine if distribution was related to seasonal prey concentrations (Womble et al. 2005, 2009). Surveys were conducted using a single-engine high-winged aircraft at approximately 305 m (1,000 ft). Survey sites located within GLBA were Graves Rocks, South Marble Island, and Point Carolus. A new Steller sea lion haulout site was discovered at Tarr Inlet, but did not yield enough data for inclusion in the distribution analysis (Womble et al. 2009).

Hastings et al. (2011) used mark-recapture models to estimate age-specific survival rates for sea lions from four southeast Alaska rookeries, including Graves Rocks in GLBA (Figure 110). Pups were marked by branding at around 1 month of age. A total of 93 pups were marked at Graves Rocks: 50 in 2002 and 43 in 2005. Surveys to re-sight marked individuals occurred through 2009 (Hastings et al. 2011).

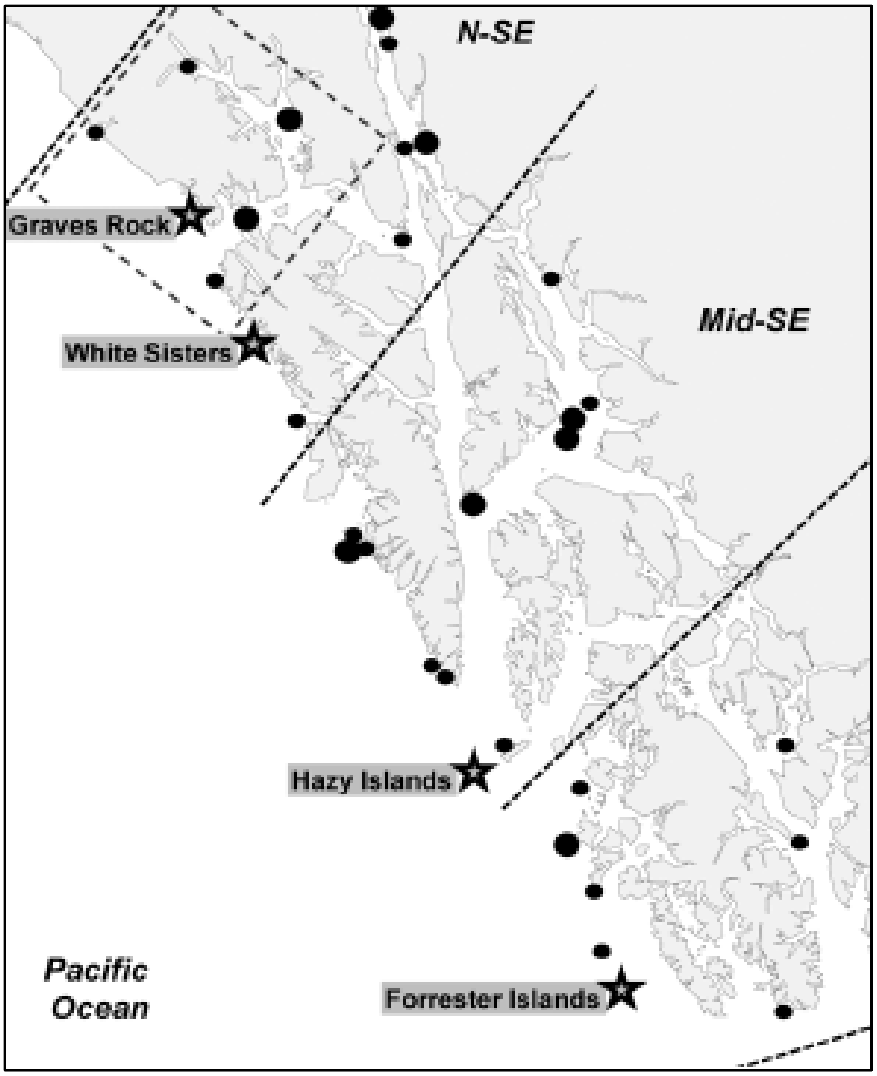

Figure 110. The locations of four southeastern Alaska Steller sea lion rookeries (denoted by stars) studied by Hastings et al. (2011).

4.15.5. Current Condition and Trend

Annual Estimates of Population Trends

Mathews et al. (2011) compiled historical counts of Steller sea lion surveys in the Glacier Bay region and estimated population trends for three GLBA sites: Harbor Point, Graves Rocks, and South Marble Island. The region-wide population trend was estimated at 8.2% (95% C.I. = 6.4–10.0%) from 1970-2009 (Table 50). South Marble Island experienced a much higher rate of population growth, with an estimated annual increase of 16.6% (95% C.I. = 12.2-21.2%) from 1991-2009. This growth rate is one of the highest ever reported for the species (Mathews et al. 2011). Steep increases in growth rate reportedly began on South Marble Island around 1997 (Mathews et al. 2011). Surveys at Graves Rocks, which began in 1989, yielded a similar increasing population trend of 3.8% (95% C.I. = 2.2-5.3%) annually (Table 50). The region-wide population trend of 8.2%, as well as estimates for Graves Rocks and South Marble Island, is above the trend for southeastern Alaska rookeries, estimated by Pitcher et al. (2007) at 3.2%. The above average population growth in the Glacier Bay region is likely due to a combination of factors, including: immigration, population redistribution, expanding habitat availability following deglaciation, reduction in mortality, and ecosystem-level changes (Mathews et al. 2011). Immigration likely influences population trends in Glacier Bay, as sea lions from the western population have been observed in the Glacier Bay region, including at Graves Rocks and South Marble Island (Mathews et al. 2011).

Table 50. Steller sea lion population trend estimates for sites within GLBA and mean trend across all Glacier Bay/Icy Strait/Cross Sound study sites (Mathews et al. 2011).

Site	Years	Trend (% per year)	SE[A]	$C.I._{95}$[B]
Graves Rocks	1989-2009	3.8	0.8	2.2, 5.3
Harbor Point	1970-2009	3.1	1.13	0.9, 5.3
South Marble Island	1991-2009	16.6	2.3	12.2, 21.2
All GB/IS/CS sites	**1970-2009**	**8.2**	**0.93**	**6.4, 10.0**

[A] Standard error

[B] Confidence intervals at a confidence level of 95%

The earliest record of Steller sea lions at a haulout within GLBA boundaries is from 1962 at Harbor Point, when counts of around 150 individuals were noted (Mathews et al. 2011, Table 51). Steller sea lions were first reported at Graves Rocks in 1969, and the site transitioned to a rookery in the late 1990s. Interior sites were colonized in the mid-1980s (South Marble Island) and 1990s (Tarr Inlet). South Marble Island is currently the largest Steller sea lion haulout site within Glacier Bay (Mathews et al. 2011). Colonization of a new haulout site in the Gloomy Knob area appeared to be occurring during the Mathews et al. (2011) study.

Table 51. Highest Steller sea lion counts and date of earliest sea lion estimates for sites within GLBA. Note that there are separate entries for the time period when Graves Rocks was a haulout (pre-1999) and after it became a rookery (Mathews et al. 2011).

Site	Colonization Year/ Earliest Estimate	Highest Count	Date of Highest Count
Cape Fairweather	<1970	407	13 March 1993
Harbor Point	<1962	264	25 June 2009
South Marble Island	1985	1,190	24 June 2009
Point Carolus	~1989	578	16 August 2007
Tarr Inlet	1999	230	8 May 2005
Graves Rocks (haulout)	≤1969	923	24 August 1996
Graves Rocks (rookery)	1999	1455 440 (pups)	18 August 2003 25 June 2009

Surveys of Steller sea lion sites within GLBA have continued since the Mathews et al. (2011) study period, but results have not been analyzed to estimate population trends (Womble, phone communication, 19 May 2015). During June surveys in 2010 and 2013, sea lions were observed at Graves Rocks and South Marble Island, but not at Cape Fairweather, Tarr Inlet, Harbor Point, or Point Carolus (Fritz et al. 2013, NMML 2014).

Spatial and Seasonal Distribution at Haul Outs and Rookeries

Glacier Bay is characterized by strong tidal mixing and high primary productivity levels sustained throughout the summer (Etherington et al. 2007), contributing to favorable forage conditions for sea lions (Womble et al. 2009, Womble and Gende 2010). Spring-spawning fish aggregations occur in several areas and salmon have colonized the Bay and its streams in recent decades, providing a food supply in summer and fall (Womble et al. 2009, Womble and Gende 2010). The colonization of new haulout sites inside Glacier Bay since the 1980s (e.g., Tarr Inlet, Gloomy Knob) could be related to this recent prey colonization and to the exposure of suitable terrestrial habitat following deglaciation (Mathews et al. 2011).

Recent research suggests that the seasonal distribution of Steller sea lions is related to the availability of seasonal prey resources (Womble et al. 2005, 2009; Mathews et al. 2011). Some Steller sea lion haulout sites are occupied year-round while others are utilized only seasonally. During a study of sea lion distribution at 28 Southeast Alaska sites, Womble et al. (2009) grouped sites based on seasonal peaks in sea lion numbers from 2001-2004. Three sites in GLBA (Graves Rocks, South Marble Island, Point Carolus) were included in the analysis. South Marble Island was classified as a Type 2 site, with sea lion numbers peaking in the spring during forage fish spawning (e.g., herring, eulachon) (Womble et al. 2005, 2009). Although South Marble Island was inhabited by sea lions throughout the year from 2001-2004 and abundance also increased in the fall, the highest numbers were observed in spring (Figure 111). Type 3 sites, such as Graves Rocks, experienced peak sea lion numbers in the summer, due to their location along primary Pacific salmon migratory corridors (Womble et al. 2009). Point Carolus was a Type 4 site, where abundance peaked in fall during the migration of fall-spawning salmon (Figure 111). Point Carolus was generally occupied less than 3 months out of the year, and was primarily used by adult and subadult males (Womble et al. 2009).

Steller sea lions were also observed opportunistically during vessel-based marine mammal surveys of Glacier Bay in the summers of 1994 to 2010 (Gabriele et al. 2011). During this period, the vast majority (88%) of hauled-out sea lion observations occurred at South Marble Island. Smaller numbers of sea lions were observed at Point Carolus (9%) and other locations (3%), including Rush Point Buoy (about halfway between South Marble and Point Carolus) and Gloomy Knob (Gabriele et al. 2011).

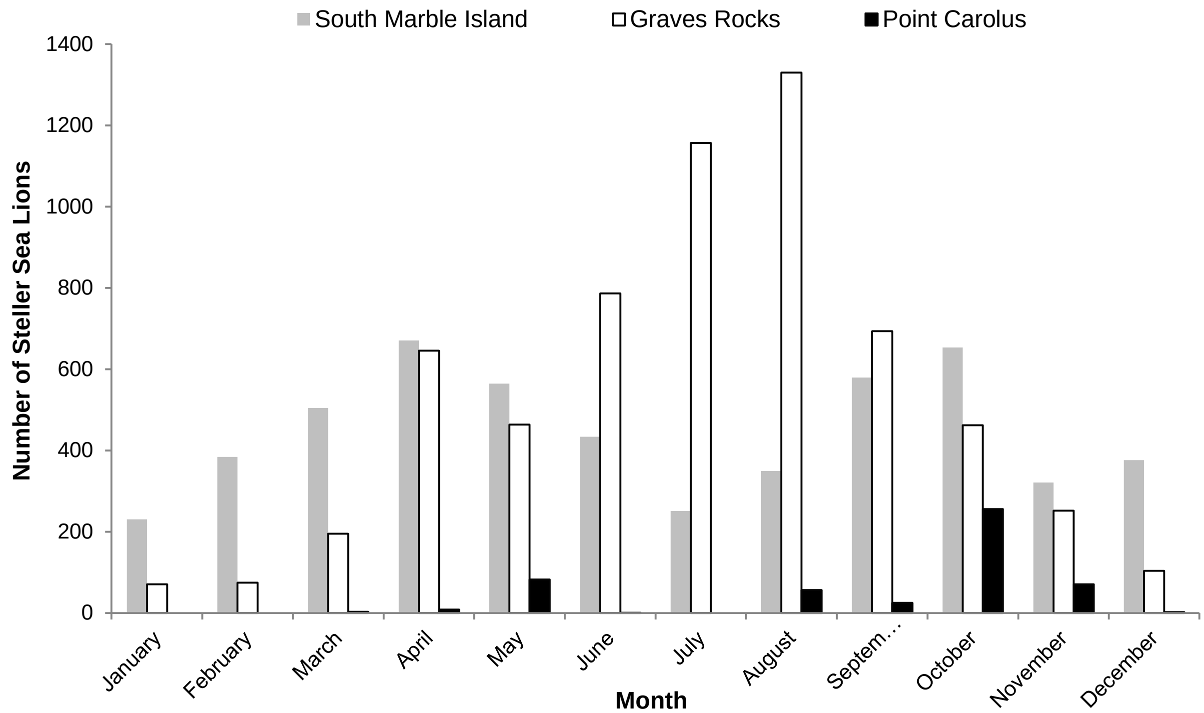

Figure 111. Mean monthly counts of Steller sea lions at three GLBA sites, 2001-2004 (Womble et al. 2009).

Large aggregations of Steller sea lions occur in the spring along the outer coast of Glacier Bay near the Alsek River and Dry Bay (Womble et al. 2005). These large aggregations of up to a few thousand Steller sea lions are associated with spring-spawning aggregations of eulachon along the Yakutat Forelands (Photo 38). Ephemeral aggregations of eulachon and other forage fish species, such as herring and capelin, likely represent an important seasonal prey resource for Steller sea lions at an energetically demanding time of year. Energy demands for adult females are high during spring as they are entering the last trimester of pregnancy and may also be nursing a pup from the previous year (Womble et al. 2005). Arriving at the rookery in good breeding condition is also critical for adult males, as they may fast for several weeks while holding territories at the rookery. Steller sea lions may also aggregate in Adams Inlet and Tarr Inlet during spring, in response to forage fish aggregations (Womble et al. 2005, Mathews et al. 2011).

Photo 38. Steller sea lion aggregation near the Alsek River during eulachon spawning (NPS photo).

Age-specific Survival Rates

A study was initiated in 2001 to quantify the age-specific survival rates of Steller sea lions at Graves Rocks and three other rookeries (Forrester Island, Hazy Islands, White Sisters) in southeastern Alaska (Hastings et al. 2011). Of the four rookery sites, Graves Rocks showed the highest survival estimates in all age classes (Table 52, Photo 39). Pup (0-1 years) survival rates for Graves Rocks were 0.762 for females and 0.729 for males. Estimated survival to 7 years of age was 0.569 for females and 0.439 for males (Hastings et al. 2011). Survival to 7 years old was twice as likely for Graves Rocks pups compared to pups from rookeries further south. The survival estimates for Graves Rocks sea lions are the highest observed to date for the species (Hastings et al. 2011). This may be related to the smaller population size at the Graves Rocks rookery (i.e., less competition). Graves

Rocks pups had larger body sizes at one-month of age than pups from other rookeries, which likely contributed to higher survival rates (Hastings et al. 2011).

Photo 39. Steller sea lions at the Graves Rocks rookery in GLBA (ADFG photo by K. White).

Table 52. Age-specific survival estimates (with 95% C.I.s) for Steller sea lions born at Graves Rocks compared to estimates from three other southeastern Alaska study sites, 2001-2005 (Hastings et al. 2011). Site locations are shown in Figure 110.

Age (yrs)	Graves Rocks	Forrester Islands	Hazy Islands	White Sisters
Females				
0-1	0.762 (0.689, 0.823)	0.567 (0.528, 0.604)	0.583 (0.540, 0.625)	0.665 (0.618, 0.709)
1-2	0.862 (0.809, 0.902)	0.718 (0.676, 0.756)	0.731 (0.687, 0.771)	0.795 (0.753, 0.831)
2-3	0.955 (0.933, 0.971)	0.878 (0.848, 0.903)	0.887 (0.856, 0.912)	0.927 (0.903, 0.945)
3-4	0.970 (0.954, 0.980)	0.915 (0.895, 0.932)	0.922 (0.900, 0.939)	0.950 (0.934, 0.962)
4-5	0.977 (0.964, 0.985)	0.934 (0.913, 0.950)	0.939 (0.918, 0.955)	0.962 (0.947, 0.972)
5-6	0.980 (0.965, 0.988)	0.942 (0.914, 0.962)	0.947 (0.919, 0.965)	0.967 (0.947, 0.979)
6-7	0.980 (0.959, 0.990)	0.943 (0.896, 0.970)	0.947 (0.903, 0.972)	0.967 (0.937, 0.983)
0-7	0.569 (0.462, 0.671)	0.277 (0.240, 0.318)	0.292 (0.247, 0.343)	0.410 (0.352, 0.472)
Males				
0-1	0.729 (0.651, 0.794)	0.523 (0.485, 0.560)	0.540 (0.497, 0.582)	0.624 (0.578, 0.669)
1-2	0.818 (0.754, 0.869)	0.647 (0.601, 0.690)	0.662 (0.613, 0.707)	0.736 (0.688, 0.778)
2-3	0.930 (0.896, 0.953)	0.816 (0.775, 0.850)	0.829 (0.787, 0.863)	0.887 (0.853, 0.913)
3-4	0.945 (0.919, 0.963)	0.851 (0.822, 0.876)	0.861 (0.831, 0.887)	0.910 (0.885, 0.929)
4-5	0.951 (0.926, 0.967)	0.866 (0.834, 0.892)	0.875 (0.843, 0.902)	0.919 (0.894, 0.939)
5-6	0.950 (0.919, 0.970)	0.864 (0.814, 0.903)	0.874 (0.825, 0.911)	0.918 (0.881, 0.944)
6-7	0.943 (0.892, 0.970)	0.847 (0.754, 0.908)	0.858 (0.768, 0.916)	0.907 (0.840, 0.947)
0-7	0.439 (0.326, 0.559)	0.156 (0.124, 0.193)	0.168 (0.130, 0.214)	0.274 (0.218, 0.337)

Threats and Stressor Factors

Human Activities

Anthropogenic threats to Steller sea lions in southeastern Alaska include incidental take (e.g., bycatch in fishing gear, entanglement in marine debris), vessel/human disturbance, and illegal harvest. Legal subsistence harvest outside of GLBA boundaries is a source of mortality, but there is little evidence of contemporary sea lion harvest in the vicinity of GLBA (Wolfe et al. 2009). In 2008, only one community (Hoonah, over 35 km [22 mi] southeast of Gustavus) reported Steller sea lion "take," with six individuals "struck and lost" (i.e., none were successfully harvested) (Wolfe et al. 2009). Subsistence harvest of sea lions declined sharply throughout Alaska in the 1990s, and the 2008 harvest was the lowest recorded since 1992 (Wolfe et al. 2009).

Conflicts or interaction with fisheries are a source of several threats to Steller sea lions. Historically, fishermen could apply for permits to shoot sea lions that were destroying fishing equipment or threatening human safety (Loughlin and York 2000). During the 1970s and early 1980s, sea lions were shot by commercial fishermen using Murphy Cove, an anchorage just 9 km (5.6 mi) northeast of Graves Rocks (Mathews et al. 2011). Shooting Steller sea lions became illegal in the U.S. in 1990 when the species was listed as threatened under the Endangered Species Act (Mathews et al. 2011). However, it is possible that illegal shooting of sea lions still occurs in southeastern Alaska, although the impact on the population as a whole is likely minimal (NMFS 2013).

Sea lions in southeastern Alaska are also impacted by entanglement or ingestion of fishing gear (NMFS 2008, Raum-Suryan et al. 2009, Mathews et al. 2011). Steller sea lions have been accidentally caught in several types of gear in Alaska waters, such as gillnets and longlines (Photo 40, NMFS 2008). Since commercial fishing is limited and being phased out in GLBA waters (NPS 2015a), the risk to sea lions is likely greater when they travel outside park boundaries. However, lost or discarded gear, as well as other hazardous marine debris (e.g., packing bands and rubber bands) can drift into GLBA and endanger marine mammals. While these indirect interactions with fisheries are thought to be only a small factor in southeastern Alaska sea lion mortalities (Mathews et al. 2011), their impact may be underestimated as animals killed by entanglement may die at sea without ever being observed (Raum-Suryan et al. 2009). Ingestion of debris can also be a "silent killer," as the damage is often internal with no external signs of damage (Raum-Suryan et al. 2009).

Photo 40. Sea lions that have ingested salmon fishery flashers (left) and longline gear at end of flashers (right) (Photo from Raum-Suryan et al. 2009).

Steller sea lions in GLBA may be stressed by private and commercial vessels that approach haulouts for wildlife viewing, particularly South Marble Island (Mathews 2000). Repeated disturbance while at haulouts or rookeries may reduce the amount of time sea lions spend resting, disrupt social interactions, and temporarily separate mothers from dependent pups (Mathews 2000). These disruptions could affect pup and juvenile survival as well as adult reproductive success (Mathews 2000); however, the population-level effects have not been quantified. NPS regulations require boaters to remain at least 91 m (100 yds) from Steller sea lions hauled-out on land or a rock to minimize disturbance (36 CFR 13.1178 NPS 2015b). However, during 23 days of monitoring at South Marble Island from 1994-1997, Mathews (2000) observed 19 vessels (21%) approaching closer than 91 m. Sixteen of these vessels (18%) caused disturbance, defined as a >20% increase in sea lion activity or >10% increase in the number of animals in the water (Mathews 2000). Observations suggested that sudden, isolated noise may elicit more of a response from sea lions than steady or gradually changing sounds (Mathews 2000).

Climate Change

Climate change poses a threat to nearly all of Alaska's terrestrial and aquatic natural resources, including Steller sea lion populations. Potential impacts of climate change on sea lions include: shifts in the timing of primary productivity peaks and oceanographic processes; changes in fish migration; shifts in the abundance and range of algae, plankton, and fish at high latitudes; and invasions of new species (e.g., pathogens or competitors) (Harley et al. 2006, NMFS 2013). Steller sea lions are generally considered highly mobile and opportunistic animals, characteristics which could reduce the species' vulnerability to climate-related changes (NMFS 2013). However, the movements of sea lions are limited at certain life stages, particularly when raising young (NMFS 2013; Womble, written communication, 29 July 2015). As a result, changes in prey distribution or abundance near

Steller sea lion haulout and rookery sites could impact the foraging efficiency of nursing females and young sea lions (Merrick and Loughlin 1997, NMFS 2013).

The increase in atmospheric CO_2 that is contributing to climate change is also causing ocean acidification, which could fundamentally alter marine foodwebs (IWGOA 2011). When oceans absorb more CO_2, water chemistry parameters such as pH, bicarbonate ions, and carbonate ions can be altered (NRC 2010); these changes impact the biological functions of marine organisms, including photosynthesis, calcification (e.g., in molluscs and corals), respiration, growth, and reproduction (NRC 2010, IWGOA 2011). Such changes would likely impact prey availability for Steller sea lions.

Health Threats (disease, pollutants)

Little is known about health threats to the eastern stock of Steller sea lions in the northern part of their range. The animals are exposed to chemical contaminants, such as mercury and organochlorines (DDT, PCBs), that are known to impact reproduction, neurological development, and immune system functions in other species (Wang et al. 2011, Castellini et al. 2012). Several male Steller sea lions in Prince William Sound showed PCB concentrations within a range known to cause physiological effects (Wang et al. 2011).

Disease can also cause mortality or reduced reproduction in pinnipeds (NMFS 2008, 2013). Alaskan Steller sea lions are known to have been exposed to two bacteria (*Leptospira* and *Chlamydia*) and one virus (San Miguel sea lion virus) that have impacted reproduction in other species (NMFS 2008). More recently, PDV has reached the North Pacific (NMFS 2013). This virus has caused epidemics, sometimes with high mortality, among North Atlantic seal (Family Phocidae) populations (Goldstein et al. 2009, NMFS 2013). Steller sea lions from the western population have tested seropositive for the virus and it has also been documented in sea otters in areas of Alaska within the range of the eastern population (Goldstein et al. 2009, NMFS 2013). Parasites may also cause mortality in malnourished or otherwise stressed individuals, but does not appear to be limiting the overall population (Haebler and Moeller 1993, NMFS 2008). Hookworms have been implicated in California sea lion (*Zalophus californianus*) pup mortality (Spraker et al. 2007) and may increase if southeastern Alaska sea lion populations continue to grow and density increases at rookeries (Spraker et al. 2007, NMFS 2013).

Natural Stressors

Natural sources of mortality or stressors upon GLBA's Steller sea lion population include predation and interspecific competition. Predation and resource competition could impact GLBA's Steller sea lions. The diets of sea lions overlap with many other marine species, both mammals and birds (NMFS 2008). Population increases or range expansions among these species could influence prey availability for Steller sea lions. Commercial fisheries could also be removing sea lion prey, contributing to nutritional stress (NMFS 2008). In terms of predation, transient orcas are a primary predator of Steller sea lions and are known to prey upon the species in Southeast Alaska (Heise et al. 2003, Dahlheim and White 2010, NMFS 2013). Matkin et al. (2007) observed transient orcas attacking and killing Steller sea lions in the Glacier Bay/Icy Strait region.

Data Needs/Gaps

Information on age-specific survival rates and spatial distribution of sea lions in GLBA is limited to the recent research described in this assessment. This research should be continued or repeated in the future to determine if any changes are occurring in these measures. Age-specific reproductive rates are currently being studied at the Graves Rocks rookery by the ADFG and analyses are currently ongoing (Womble, written communication, 29 July 2015). Further study of Steller sea lion diet in the Glacier Bay region could be helpful in understanding the species' seasonal distribution.

Additional research is needed into the impacts of threats and stressors on the southeastern Alaska Steller sea lion population. For example, the effects of contaminants and parasitism on the species are unknown (NMFS 2008, 2013). The potential synergistic effects of these threats, as well as disease and other natural stressors at Steller sea lion haulouts/rookeries, have not been studied (NMFS 2013). Further research into how climate change-related ecosystem shifts are impacting or will impact the Steller sea lion population would also be useful (NMFS 2008).

Overall Condition

Annual Estimates of Population Trends

The *Significance Level* for this measure is 3. Population trends have been estimated for three sites within GLBA: Harbor Point, Graves Rocks, and South Marble Island. Populations at these sites were increasing at annual rates of 3.1%, 3.8%, and 16.6%, respectively (Mathews et al. 2011). These are all near or above the trend for southeastern Alaska rookeries (1979-2005) of a 3.2% increase annually (Pitcher et al. 2007). This measure was assigned a *Condition Level* of 0, indicating no concern.

Spatial and Seasonal Distribution at Haul Outs and Rookeries

This measure was assigned a *Significance Level* of 2. The seasonal distribution of Steller sea lions appears to be related to the availability of seasonal prey resources (Womble et al. 2005, 2009; Mathews et al. 2011). With the relatively recent colonization of several haulout sites within GLBA (South Marble Island, Tarr Inlet, Gloomy Knob) and the transition of Graves Rocks to a rookery, Steller sea lion distribution appears to be expanding in the area. As a result, this measure was assigned a *Condition Level* of 0 (no concern).

Age-specific Survival Rates

The project team assigned this measure a *Significance Level* of 3. Although not studied until recently, age-specific survival rates for the Graves Rocks rookery in GLBA are higher than at three other southeastern Alaska rookeries and are the highest rates so far observed for the species (Hastings et al. 2011). Therefore, this measure is currently of no concern (*Condition Level* = 0).

Weighted Condition Score

The *Weighted Condition Score* for Steller sea lions is 0.13, suggesting good condition. Given the high survival rates, increasing populations, and relatively recent colonization of new haulout sites, the trend for sea lions seems to be improving.

Steller Sea Lions			
Measures	Significance Level	Condition Level	WCS = 0.00
Annual Estimates of Population Trends	3	0	
Spatial and Seasonal Distribution at Haulouts and Rookeries	2	0	
Age-specific Survival Rates	3	0	

4.15.6. Sources of Expertise

- Jamie Womble, NPS Wildlife Biologist

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4.16. Anadromous Fish

4.16.1. Description

The word anadromous means "upward running" and refers to a relatively uncommon life history strategy utilized by approximately 100 of the more than 28,000 species of fish. Anadromous fish are born in freshwater, spend some portion of their lives in the marine environment and return to spawn in freshwater. Nine anadromous fish species occur in the waters of GLBA, including: Chinook salmon (*Oncorhynchus tshawytscha*), sockeye salmon (*O. nerka*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), pink salmon, steelhead trout, sea run cutthroat trout, Dolly Varden char, and eulachon smelt (NPS 2015a).

Photo 41. Sockeye salmon in the East Alsek River of GLBA (NPS photo).

Pacific salmon range across the Pacific Rim region, from as far southeast as the central coast of California, through much of the Pacific Northwest (Oregon, Washington, Idaho), Alaska, and throughout the U.S., Canadian, and Siberian Arctic to Japan. Salmon are semelparous, meaning they die after spawning. Spawning occurs in freshwater streams, where eggs are laid and fertilized in gravel depressions called redds. After several months, alevin hatch from the eggs and reside in the gravel for an additional 3-4 months while they grow. When salmon emerge from the gravel they are called fry; salmon fry have a variety of residence times in freshwater. Soon after emergence, pink and chum fry immediately begin their migration to the ocean, while sockeye, coho, and Chinook may reside in freshwater and estuaries for 1-3 years. Salmon smolt will spend 1-5 years growing in the ocean before returning to their natal stream. Depending on the species, age, and population, approximately 1-5% of salmon stray to a new stream to spawn (Quinn 2005). Straying allows for the colonization of new habitats, gene flow between populations (Curry 1994, Hendry et al. 2004), and can serve as a buffer against changes in habitat quality (McDowall 2001). Salmon populations are inherently variable, as production is strongly linked to their environment (Knudsen and McDonald 1999). In general, salmon populations have the adaptive ability to deal with changes that fall within the historic temporal and spatial scales of natural disturbance, but larger changes might require a response that is beyond the evolutionary capability of salmon (Waples et al. 2008).

Salmonids are keystone species, as they provide a resource base for several ecosystems (terrestrial and aquatic) (Willson and Halupka 1995, Armstrong and Hermans 2007). Salmon are additionally termed as a "cornerstone species" due to their ability to support several different ecological communities (Willson et al. 1998). Salmon are unique in that they link together several ecosystems (e.g., ocean, freshwater, and terrestrial) as they distribute nutrients, promote the fitness of other species, and influence the survival of other fish, birds (e.g., sea birds, raptors), and mammals (e.g., seals, sea lions, porpoises, orcas, and brown and black bears) (Willson and Halupka 1995). Because salmon are semelparous, they provide a tremendous influx of nutrients to spawning streams and their watersheds. These nutrients benefit the aquatic insects in streams that then serve as food for juvenile salmon and other fish. Bears and other terrestrial vertebrates move the salmon away from water sources, and fish carcasses nourish the soil of the near-stream vegetation and trees (Gende et al. 2002, 2004).

In addition to ecological importance, salmon also possess a great deal of cultural and economic importance in Southeast Alaska (Armstrong and Hermans 2007), and have been a fundamental part of human life in the state for thousands of years (Sisk 2007). The harvest of salmon and other anadromous fish in Alaska has played a major role in the history and economy of the state. Salmon (and their harvest) in Southeast Alaska were at the center of the battle for statehood, and were the focus of the first major industry in Alaska after the Russian sale of the land to the U.S. Currently, Alaska supports and produces approximately 80% of all salmon harvested in the Western U.S. and Canada (Burger and Wetheimer 1995), and the annual value of salmon harvests in the state can be in the range of tens to hundreds of millions of dollars. Culturally, salmon represent a foundation resource for many indigenous Native tribes. Salmon and properties associated with salmon harvest are of critical importance to the Tlingit tribe. As George Emmons explained, writing between 1880 and 1904:

> *The most valuable property of the Tlingit was the fishing ground or salmon stream, which was a family [lineage] possession, handed down through generations, and never encroached upon by others. In the case of a poor family that lacked a stream sufficient for their needs, or if they had suffered a failure of the run, another lineage might extend an invitation to fish in their stream, but only after the owner had satisfied his needs (Emmons 1991, p. 105).*

Each year, many of GLBA's more than 300 streams swell with runs of sockeye, pink, coho, and chum salmon. This wasn't the case just 200 years ago, as the area that is now GBP was inundated by the Glacier Bay Ice Sheet. As the Glacier Bay Ice Sheet receded, salmon began to colonize the many freshwater streams that were formed. These processes of glacial recession, stream succession, and colonization are ongoing in Glacier Bay. Milner et al. (2011) found that the succession of the fish community in GBP streams is strongly related to stream age, and further found that salmonid abundance and distribution may be dependent on habitat complexity and stability. Milner's continued research demonstrated that the flow buffering effects of lakes in watersheds accelerates successional processes and, in turn, salmon colonization (Milner 1987, Milner and Bailey 1989). The park's marine environment is also very important habitat for juvenile and adult salmon, which feed and grow in the rich waters of GLBA (Robards et al. 1999).

Nearly 90% of the forested region of Southeast Alaska is within 5 km (3 mi) of one of the more than 5,000 salmon streams (Willson et al. 2004). Salmon abundance in Southeast Alaska has experienced a great deal of variation, although anthropogenic impacts have been far less substantial as elsewhere in North America. Periods of overfishing, destructive logging, and mining practices have certainly had negative impacts, but Southeast Alaska has not experienced the same extirpations as California, Oregon, Washington and Idaho. There is no evidence of widespread loss of spawning aggregates in Southeast Alaska (Baker et al. 1996). Threats still exist, though currently the State of Alaska protects salmon and salmon habitat in statute. Habitat degradation, anthropogenic climate change and ocean acidification, and invasive species are looming stressors on salmon stocks health in Southeast Alaska.

Salmon are harvested in GLBA through commercial, subsistence, personal use, and sport fisheries. Commercial salmon fishing in GLBA is primarily conducted with troll gear from power and hand troll fishing vessels. The two exceptions are the Dry Bay set gillnet fishery and the Excursion Inlet fall chum seine fishery. There are many different commercial salmon fishing seasons depending on the gear type, season, and location. Salmon are also harvested by Alaska subsistence and personal use fishers with nets, seines, spears, and gaffs, and by sport anglers with rod and reel. While federal subsistence under ANILCA is not authorized within the national park portion of GLBA, it is authorized in the preserve. Subsistence and personal use fisheries are authorized only for designated freshwaters and only for specific periods when fish are present, ranging from 45-151 days in duration (Soiseth, personal communication, 2015). Because NPS regulations (36 CFR 2.3 (d) (1)) prohibit fishing in freshwater in any manner other than with hook and line, personal use salmon fishing is not permitted within freshwater. There is no freshwater prohibition on these gear types for subsistence or personal use salmon fishing in the preserve because of the federal subsistence authorization there. Sport fishing for salmon is open all year in saltwater and freshwater. One exception to this is Chinook salmon, which are not legally authorized to be taken in freshwater in Southeast Alaska.

By nature, the management of salmon harvest is complex. Salmon often migrate thousands of miles in the ocean before returning to their natal stream; migrating groups of fish may be comprised of stocks from many different streams, and it is common for Canadian stocks of salmon to be harvested in U.S. water when they are intermingled and vice versa. Many of the Chinook salmon caught by the Southeast Alaska troll fishery originate from British Columbia and Pacific Northwest streams (Clark et al. 2006). These mixed-stock intercept fisheries necessitate management cooperation between the U.S. and Canada. The Pacific Salmon Treaty (PST) was signed in 1985 (16 U.S.C. 3631) and commits both nations to carry out their respective salmon fisheries and enhancement programs such that: over fishing is prevented, optimum production is sustained, and both countries receive benefits equal to the production of salmon originating in their waters. The Treaty also establishes annual harvest quotas that each country must adhere to.

The main objectives of the Alaska Salmon Management Program are to meet escapement goals, promote harvest of good quality salmon, attain Board of Fisheries allocations among sectors and gear types, and abide by the PST (Clark et al. 2006). The ADFG utilizes a variety of methods to manage salmon harvest.

The mixed stock or "intercept" winter Chinook salmon troll fishery in Southeast Alaska, for example, is managed in accordance with the Alaska Board of Fisheries Winter Troll Management Plan requirements as specified in Alaska Administrative Code (5 AAC 29.080) and the PST (Skannes and Hagerman 2015). The winter troll fishery typically opens 11 October and continues through the end of April or until a total of 45,000 non-Alaska hatchery reared Chinook plus Alaska hatchery Chinook are harvested (5 AAC 29.070 (a) (1) and 5 AAC 29.080 (a)). Hatchery produced Chinook salmon from outside Alaska harvested in the winter fishery count towards the season's treaty allocation. Any remaining treaty Chinook not harvested during the winter fishery are available for harvest in the spring and summer troll fisheries. In-season monitoring of fishery performance, stock composition, and escapement allow managers to issue emergency openings or closures in order to meet annual harvest targets and abide by conditions of the PST.

Terminal salmon fisheries are fisheries that occur on a specific population(s) and are typically within close proximity to, or are in the natal stream, of the targeted stock. These fisheries, such as the ones that occur in the Dry Bay East Alsek set net sockeye fishery and the Excursion Inlet fall chum seine fishery, are managed by monitoring stream-specific run strength and ensuring escapement is met through a variety of methods which may include aerial surveys, sonar counts, or fish weirs. Aerial surveys conducted by ADFG are the primary method used within the park.

Although nine species of anadromous fish occur in GLBA, this assessment focuses only on the five species of commercially harvested Pacific salmon (chinook, sockeye, coho, chum, and pink); eulachon smelt are discussed in Chapter 4.17 of this document. As steelhead trout, sea run cutthroat trout, and Dolly Varden are not commercially-targeted species, and less information is available to evaluate stocks in GLBA or regional stocks as a proxy for GLBA, an assessment for these species is not included.

4.16.2. Measures

- Distribution
- Removals

4.16.3. Reference Conditions/Values

Anadromous fish populations are known to experience a high degree of natural variation in abundance. Species and populations can vary greatly in how they respond to environmental changes. An ideal reference condition could include: no stocks depleted or extinct, fully functioning stream ecosystems, no dams/impoundments/developments, no invasive species, and contaminant loads that are below the thresholds that endanger wildlife or human health. Currently, too little is known about anadromous fishes within GLBA to identify an appropriate reference condition. The condition of this resource will be determined by best professional judgment of the identified park experts.

4.16.4. Data and Methods

Limited information exists regarding anadromous fish distribution and abundance in GLBA, and sources of information currently available are variable regarding their scope, quality, and utility (Soiseth and Milner 1995). The data sources used to assess current condition for the two selected metrics in this assessment (distribution and removals) are summarized by subheading below.

<u>Distribution</u>

The NPS Stream Survey Database (NPS 2015b) provides sporadic and anecdotal salmonid observations across many of the streams found in GLBA. This database contains stream-specific fisheries observations dating from 1954-2009, and indicates the stream, watershed, time of year, species, life stage and number of individuals and/or fish carcasses observed by species. Data originated from published and unpublished information sources (e.g., ADFG aerial survey data, investigator’s annual reports, stream survey data, in-house reports, and ranger patrol log books) as well as a stream resources information survey and questionnaire administered to local area agency personnel.

The database was used primarily in this assessment to provide visual reference to locations in the park where salmonid species have historically been documented but can also be used to evaluate distribution by species. Higher level inferences (i.e., density, relative abundance, etc.) are not possible due to the opportunistic updating process of this database, and lack of a consistent standardized inventory and monitoring approach.

Soiseth (1995) initiated data collection for the NPS Stream Survey database by finding, compiling and databasing a wide variety of reports and records pertaining to anadromous and resident freshwater fish in park streams and to summarize and evaluate the information in order to provide a baseline and foundational data source for other relevant work.

Soiseth and Milner (1995) inventoried and characterized stream systems and catchments in GLBA as part of an effort to develop a model that predicted potential salmonid occurrence in the park. After constructing a database of individual stream systems and the physical parameters of each stream system, Soiseth and Milner (1995) developed a model that predicted streams that may potentially contain salmon populations. Predictions of occupancy in a stream were based on two main criteria:

1) a gradient of $\leq$15% in the first 0.5 km (0.3 mi) or more above the stream mouth;
2) water clarity/flow stability inherent in surface runoff-, lake-, or proglacial lake-influenced stream systems (Soiseth and Milner 1995, p. 176).

Milner et al. (2000) investigated the long-term patterns of stream communities and habitat change in GLBA by examining physical and biological variables in 16 variable aged streams. Selected streams represented up to 200 years post deglaciation stream development. Four questions were addressed:

- How do lakes in different age watersheds affect downstream physical attributes and biological communities?
- How does stream age influence physicochemical variables and biological communities?
- What is the effect of these variables on colonization and abundance of invertebrates and fish?
- What is the relative importance of salmon carcass derived marine nutrients in the growth of riparian and instream vegetation, macroinvertebrates, and juvenile salmonids among sampled streams?

Milner et al. (2000) additionally presented a conceptual stream development summary integrating within-stream trophic interactions and other ecosystem influences (terrestrial, marine, and lake) on stream ecosystem function and successional processes.

Removals

Salmon harvest in GLBA waters is attributed to three primary sectors: commercial harvest, sport fishing, and subsistence/personal use harvest. Regional commercial troll harvest is reported as an indicator of salmonid abundance. There are only two directed terminal (targeting a specific run) commercial fisheries in GLBA (East Alsek set gill-net and Excursion Inlet seine). Data sources are organized under these subheadings.

Commercial Harvest

ADFG publishes an annual salmon harvest report by region each year. The most recent report by Conrad and Gray (2014) provides an overview of both the commercial and personal use/subsistence salmon harvest for the Southeast Alaska/Yakutat Region for the 2014 fishing season. Discussions of historical harvest in the region are also included, dating back to the late 1870s.

Skannes et al. (2015) summarized the 2014 Southeast Alaska/Yakutat commercial troll fishery harvest and earlier years dating back to statehood (1960 fishing season). The Southeast Alaska/Yakutat troll fishery occurs in State of Alaska (5 km [3 mi] offshore) and the Federal Exclusive Economic Zone (EEZ; 322 km [200 mi] offshore) waters. Troll fisheries also take place in some GLBA waters including those along the outer coast to 4.8 km (3 mi) offshore (District 116) as well as portions of District 14A and 14B (Figure 112). Spring, summer and winter troll fisheries occur. The spring troll fishery spatial limits are displayed in Figure 112. Northern Southeast Alaska salmon fisheries harvest management and harvest reporting areas in the Glacier Bay area are presented in Figure 113. Skannes et al. (2015) also summarized the all-gear harvest of Chinook and coho salmon from 1891-2014.

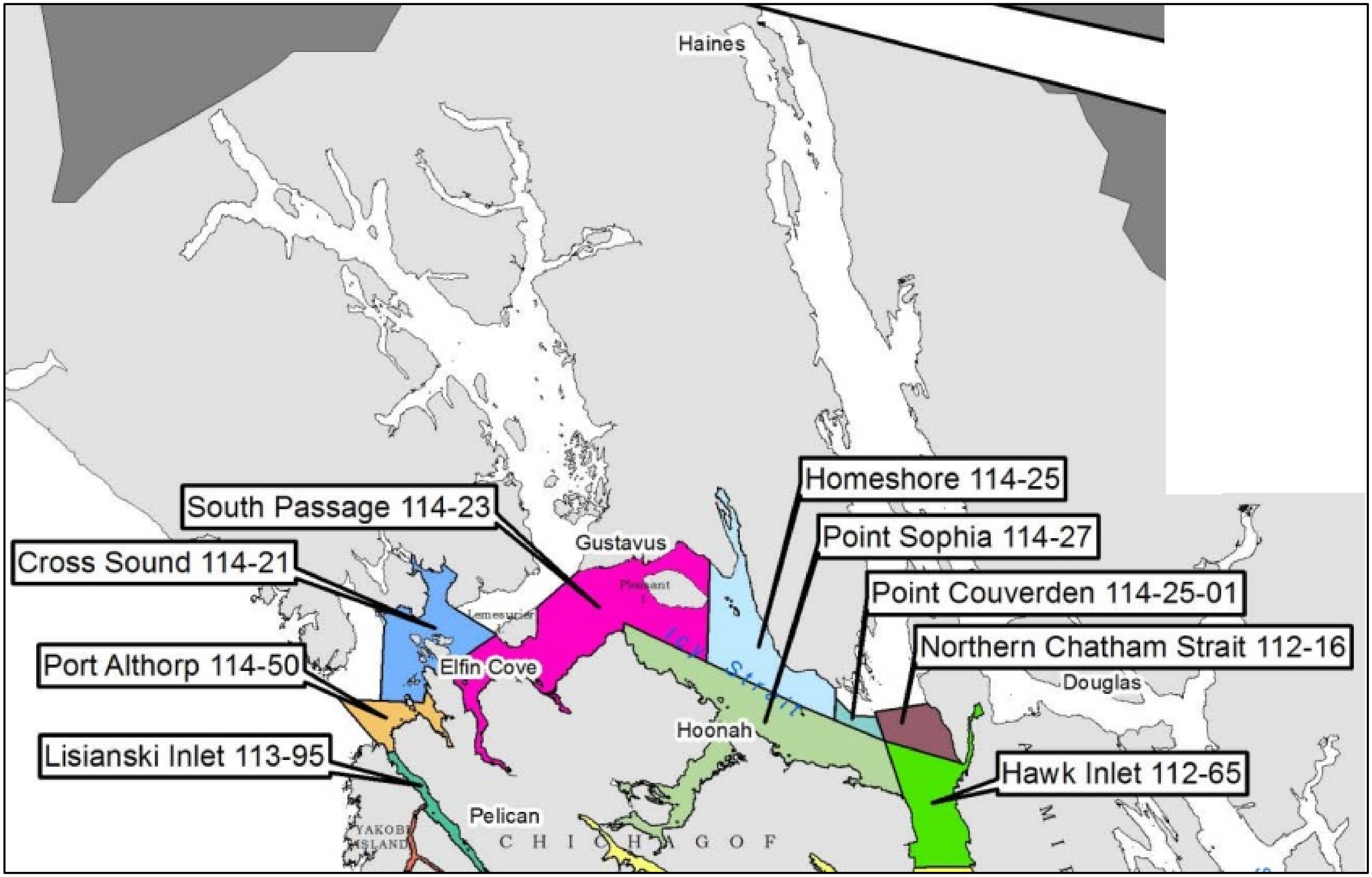

Figure 112. ADFG spring troll fishing areas in the GLBA area of the Southeast Alaska/Yakutat region (ADFG 2015b).

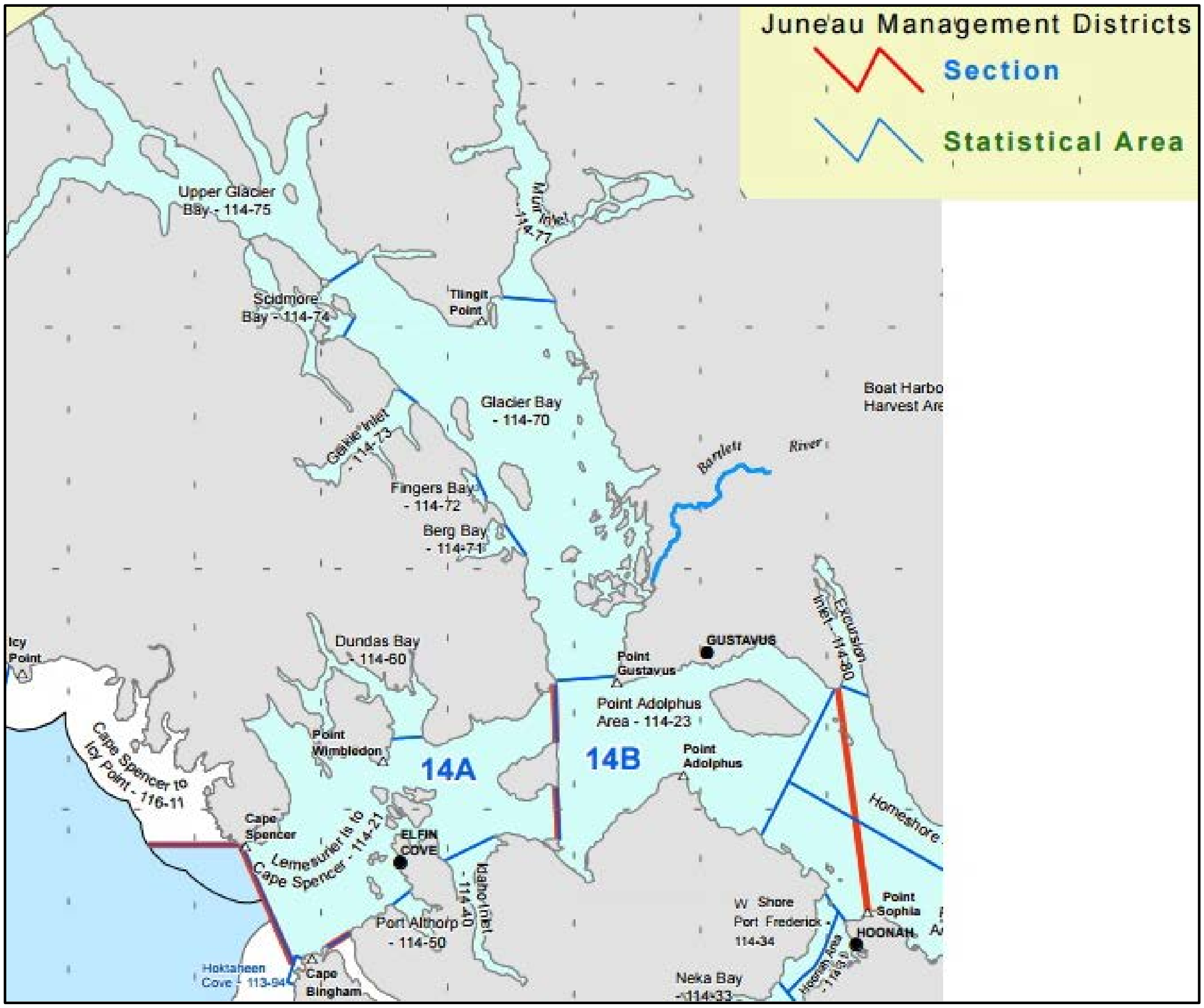

Figure 113. Salmon fishery management districts and statistical areas in the Southeast Alaska/Yakutat region near GLBA (ADFG 2011). Outer coast areas of GLBA are not included in this figure, but fall within District 16.

Eggers and Heinl (2008) describe chum salmon stock status and escapement goals for the Excursion River fall-run chum salmon seine fishery. This terminal fishery occurs in District 14C (east of 14B), specifically within Excursion Inlet (Figure 113; statistical area 114-80) and nearby areas when run-strength is adequate. Peak aerial spawner escapement survey data are available for the Excursion River since 1960 (with the exception of 1963). Purse seine harvest of fall-run chum salmon from statistical area 114-80 is available dating back to 1964.

The East Alsek/Doame River set gillnet sockeye fishery was once (1970s-1980s) one of the most productive sockeye fisheries in the Yakutat area, with annual fishery harvests approaching 200,000 fish (Faber et al. 2008, Heinl et al. 2011). The fishery targets migrating sockeye fresh from the Gulf of Alaska in the lower end of a long, tidally influenced estuarine area on their journey to several natal spawning streams including the East Alsek and Doame Rivers among numerous smaller tributary streams. Periodic flooding events from the large intercoastal Alsek River into the East Alsek River that flushed fine sediments and reset spawning habitat no longer occurred after 1981 when the last of five documented flooding events were reported (Faber 2008). The loss of periodic flooding and subsequent accumulation of fine sediment in this low gradient, slow flowing system are thought to have resulted in decreased spawning habitat quantity and quality and productivity of this system has declined. Heinl et al. (2011) describe sockeye salmon stock status and escapement goals for the terminal set gillnet fishery in the East Alsek/Doame River system within the preserve. Commercial set gillnet harvest data for sockeye salmon in the Yakutat area including the East Alsek/Doame River are available since 1960.

Sport Harvest

Sport fishing total harvest statistics (total number of fish kept) have been estimated by the ADFG in Alaska since 1977, while total catch statistics (total number of fish kept plus fish released) have been estimated since 1990 (ADFG 2015). These estimates of harvest occur primarily through the use of mail surveys which are sent annually to a random sample of fishing license holders. Survey results can be downloaded from the ADFG website (http://www.adfg.alaska.gov/sf/sportfishingsurvey/index.cfm?ADFG=area.results), although online results are only available for 1996-2014. Sport harvest estimates are based solely on Bartlett River (in GLBA) anglers. The Bartlett River is likely the most intensively sport fished river in the park, and is a good barometer of sport fishing effort and harvest for salmon in park streams (Craig Murdoch, GLBA Fisheries Biologist, written communication, 17 November 2015). Years where fewer than 11 people responded to a survey are not displayed in the ADFG data.

Subsistence and Personal Use Harvest

Subsistence and personal use harvest data were obtained by GLBA from ADFG in February 2013 (ADFG 2013 for the Excursion River, East Alsek River, North Berg River, and the Dundas River. This assessment will focus primarily on chum and sockeye salmon harvest, as those harvests occur in the greatest number. As previously noted, Conrad and Gray (2014) present the subsistence/personal use harvest in the Southeast Alaska/Yakutat Region.

4.16.5. Current Condition and Trend

Distribution

NPS Stream Survey Database (NPS 2015b)

An estimated 74% of park streams have been documented to date as containing anadromous fish (Figure 114) based on NPS data (NPS 2015b). Anadromous fish presence is relatively well-reported and documented in the south-central, southeastern and mid-bay (between the West and East arms) area (Figure 114). The more recently deglaciated areas in the upper West and East arm appear to exhibit fewer streams reported to contain anadromous fish. This is consistent with Milner et al. (2000) findings where fish abundance and diversity were significantly reduced in younger, more recently deglaciated streams. Moreover, it's most likely because stream successional processes and hydrological stability in the more recently deglaciated streams are not yet conducive to salmonid colonization. Alternately, more recent salmonid colonization may have occurred but is yet to be observed, reported and documented (Figure 114).

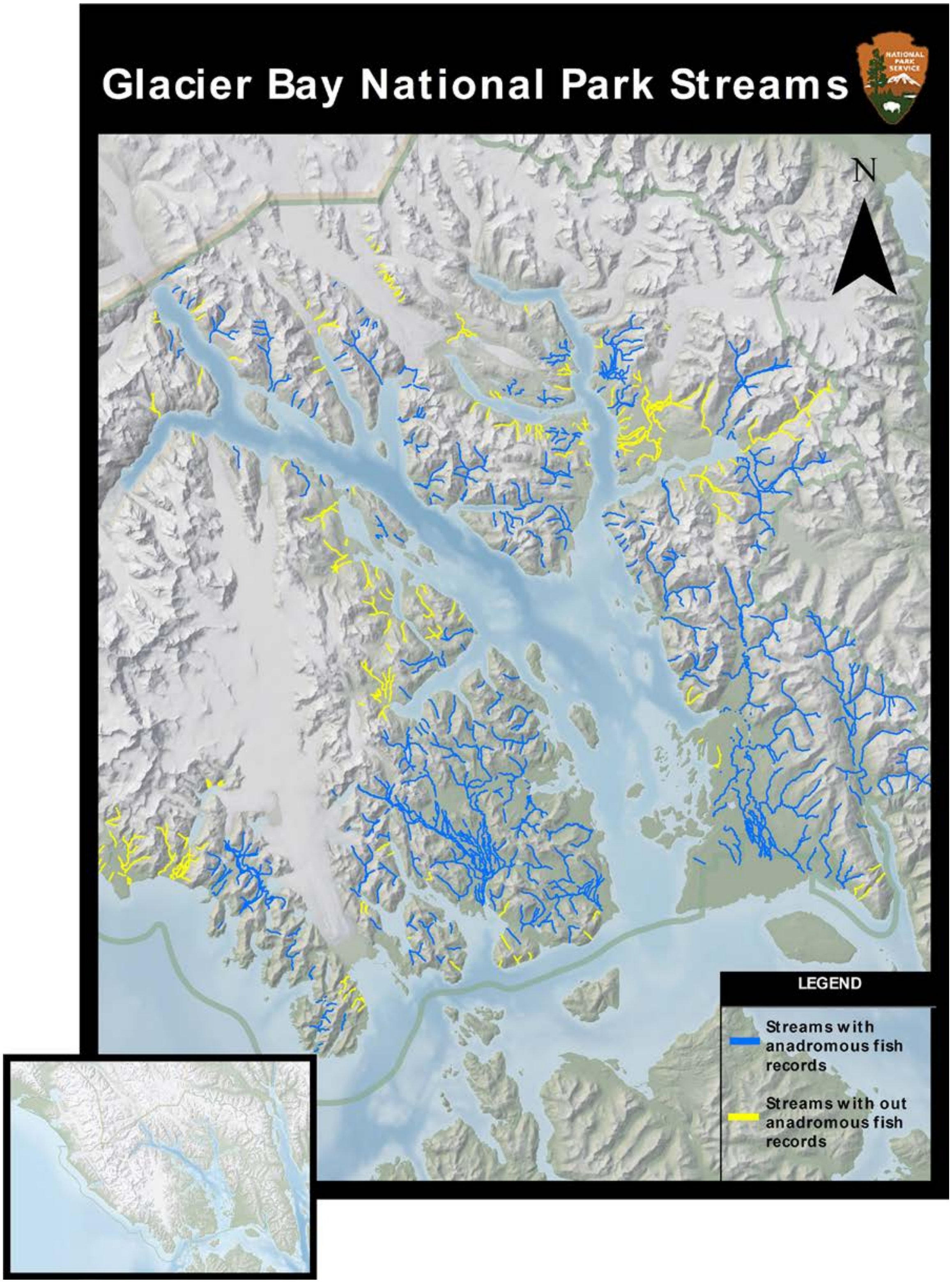

Figure 114. Freshwater streams with anadromous fish records in and near Glacier Bay Proper. Note that the streams that occur near and along much of the outer coast are not visible in this figure (Craig Murdoch, GLBA Fisheries Biologist).

Soiseth (1995)

An unpublished internal NPS report (Soiseth 1995) found that fish presence information existed for about half of the more than 300 identified park streams. However, it is likely that additional unsurveyed streams contained fish but were missed due to a lack of information. Sixty two percent of 156 surveyed freshwater streams were reported to contain fish. This agreed with modeled predictions by Soiseth and Milner (1995). Thirteen species of fish were identified among reported streams, and the highest species richness value observed in a single stream was 11 species. Salmonid species (particularly pink, chum, coho, and Dolly Varden) were the most widely distributed of all fish species, and were observed in over 40 park streams at that time. Seventy percent of fish bearing streams reportedly contained two or fewer species, with pink salmon and Dolly Varden most commonly observed.

Soiseth and Milner (1995)

The coarse and relatively simplistic model developed by Soiseth and Milner (1995) predicted that approximately 60% of GLBA streams contained salmonids. A lack of more detailed and current spatial information as well as more extensive validation and testing undoubtedly hindered model accuracy. Validating the model was challenging, as information on salmonid presence and distribution in the park were and still continue to be limited.

Milner et al. (2000)

During a 1997 survey of 16 streams across GLBA, Milner et al. (2000) found juvenile coho and juvenile Dolly Varden to be widely distributed in the park, regardless of stream age. Juvenile coho salmon were found in 10 of the 16 streams (the youngest stream being ~43 years old), and Dolly Varden were in 13 streams (including three of the youngest streams) (Milner et al. 2000). Milner et al.’s (2000) findings are consistent with those of Milner and Bailey (1989), who found that Dolly Varden are typically the first salmonid colonizers of newly formed streams. Overall, this work, along with earlier work by Milner, indicate that colonization and fish community succession are strongly related to stream age since deglaciation.

Large scale physical processes including isostatic rebound, mountain building and stream incision along with periodic mass wasting events can impede spawner access to specific stream reaches and even entire streams. One of the first known examples was the Vivid Lake north stream system which was historically used by sockeye salmon to access Vivid Lake. By the mid to late 1990’s, at least two research groups began reporting returning sockeye lake access difficulty for the north stream returning spawners (Soiseth, unpublished data). Similarly, a flood regime change as a consequence of isostatic rebound and channel incision in the vicinity of the East Alsek River was postulated by Faber (2008), ultimately affecting sockeye productivity for one of the Yakutat area’s top producing salmon streams. NPS fisheries staff has observed interconnected lake basins within the Bartlett River drainage separated by shallow sills where sockeye will likely also eventually lose access to historical spawning areas (NPS unpublished data). These changes can have significant consequences for the quality and quantity of available sockeye spawning and rearing habitat. But at the same time that traditional spawning habitat access limitation is occurring, new streams are emerging from the ice which will eventually be colonized by salmonids. The relative rate of these two processes is currently unknown but the declines in sockeye habitat quantity and quality are most evident for this species due to their association with lake habitat features. The dynamic nature of colonization in conjunction with spawning and rearing habitat changes will continue to influence anadromous fish distribution and abundance over time.

Removals

GLBA falls within the Region 1 Southeast/Yakutat salmon fisheries management region, encompassed by Cape Suckling to the north and Dixon Entrance to the south. Salmon net registration area A (Southeastern Alaska area) is further divided into regulatory districts (1-16), which are further subdivided into sections (A, B, etc). GLBA waters occur within regulatory districts 16 (outer coast) and 14 (inside waters), with GBP in district 14B, and western Icy Strait and Cross Sound waters in district 14A (Figure 113).

In Registration Area A, salmon are commercially harvested via purse seines, drift gillnets, and hand and power troll gear (Conrad and Gray 2014). The salmon fishery in Registration Area A is a mixed stock and mixed species fishery. Management of this fishery is a complex task, and the ADFG produces management plans prior to each season that outline how the fishery will be managed as well as expected harvest goals and anticipated salmon returns (Conrad and Gray 2014). According to Conrad and Gray (2014, p. 2-3),

> *… the region [Southeast Alaska] contains approximately 5,500 salmon-producing streams and tributaries of various productivity levels, [and] it is impractical to apply stock-specific fisheries management for most individual returns. Additionally, some salmon harvested in the region originate from other states (primarily Washington and Oregon) and Canada. Net and troll fisheries in Southeast Alaska and Yakutat are managed for sustained yield and allocated among users according to Alaska Board of Fisheries regulations and harvest-sharing provisions of the Pacific Salmon Treaty between the United States and Canada.*

In the GLBA area, federal law and regulation allow for the continuation of commercial fishing activities in GLBA outer waters (outside of GBP, including Cross Sound, Icy Strait, and along the outer coast) (Figure 115). Any fishery that occurred prior to the 1999 final rule is allowed to continue, but new or expanded fisheries are prohibited (36 CFR 13.1130(c)). Existing fisheries at the time of the 1999 final rule were documented in NPS (1998), and are listed annually in the GLBA compendium. Troll harvest is the only gear type currently allowed in GBP and participants must possess a Lifetime Access Permit (LAP) in order to participate in the fishery. The LAP is renewed by the NPS at 5-year intervals for the lifetime of qualified fisherman. Permitted fishers are authorized to fish until they are no longer able. The permit is non-transferable. Currently, there are an estimated 45 LAP salmon troll permit holders, but this estimate may be revised downward for the 2015-2020 period as current holders seek renewals (Soiseth, written communication, 26 October 2015). Commercial salmon trolling is authorized in the outer waters of GLBA as is purse seines for chum salmon in Excursion Inlet only.

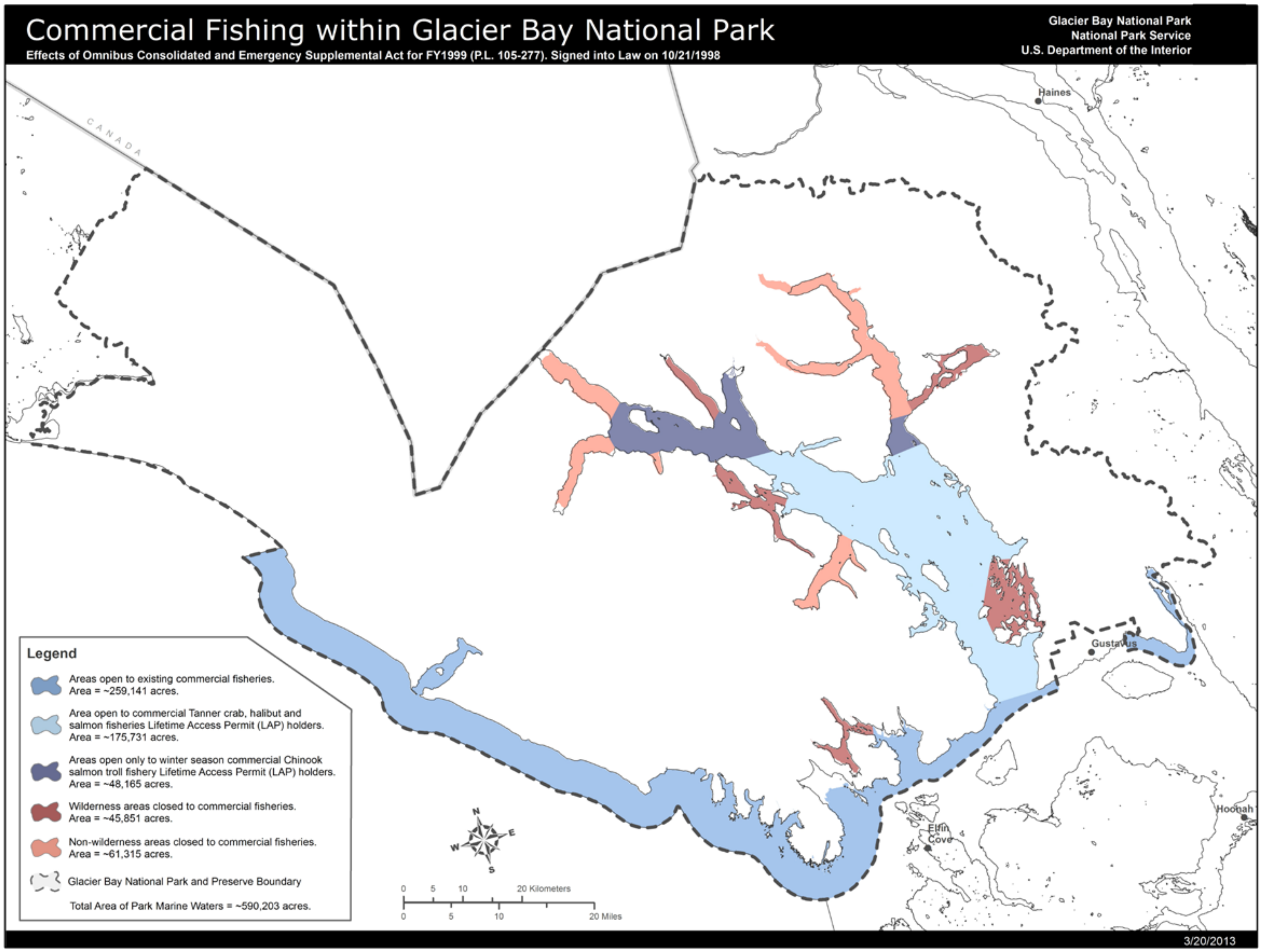

Figure 115. Commercial fishing waters within GLBA (NPS 2015c).

The proportion of GLBA salmon harvested as part of the larger Southeast Alaska commercial salmon troll fishery is unknown. The troll fishery is generally a mixed stock fishery and the Southeast Alaska region contains approximately 5,500 salmon-producing streams and tributaries. Further, some component of salmon troll harvest also originates from areas in the Pacific Northwest and Canada. This section serves instead as an indicator of regional salmon production.

Commercial Harvest

Pink salmon began to dominate harvests beginning in the early 1900s, and this species has comprised 69% of total harvests in the past 10 years (Conrad and Gray 2014). Currently, the order of salmon production (e.g., total number of fish harvested) from highest to lowest in the Southeast Alaska/Yakutat region is: pink, chum, coho, sockeye, and Chinook salmon (Conrad and Gray 2014). The two highest valued (i.e., price per pound) species on an individual fish basis are, respectively, Chinook and coho.

Harvest estimates for salmon in Region 1 peaked at over 60 million salmon in the early 1930s and 1940s (Figure 116), before reaching all-time lows in the following 3 decades (Figure 116). Substantial harvest increases were observed in the region from the 1980s-2000s, with record harvests being reported for many species during this period: coho (5.7 million; 1994), chum (16 million; 1996), and pink (94.8 million; 2013) (Figure 117). The regional record for total commercial salmon harvest is 112.4 million fish, which was established in 2013 (Conrad and Gray 2014).

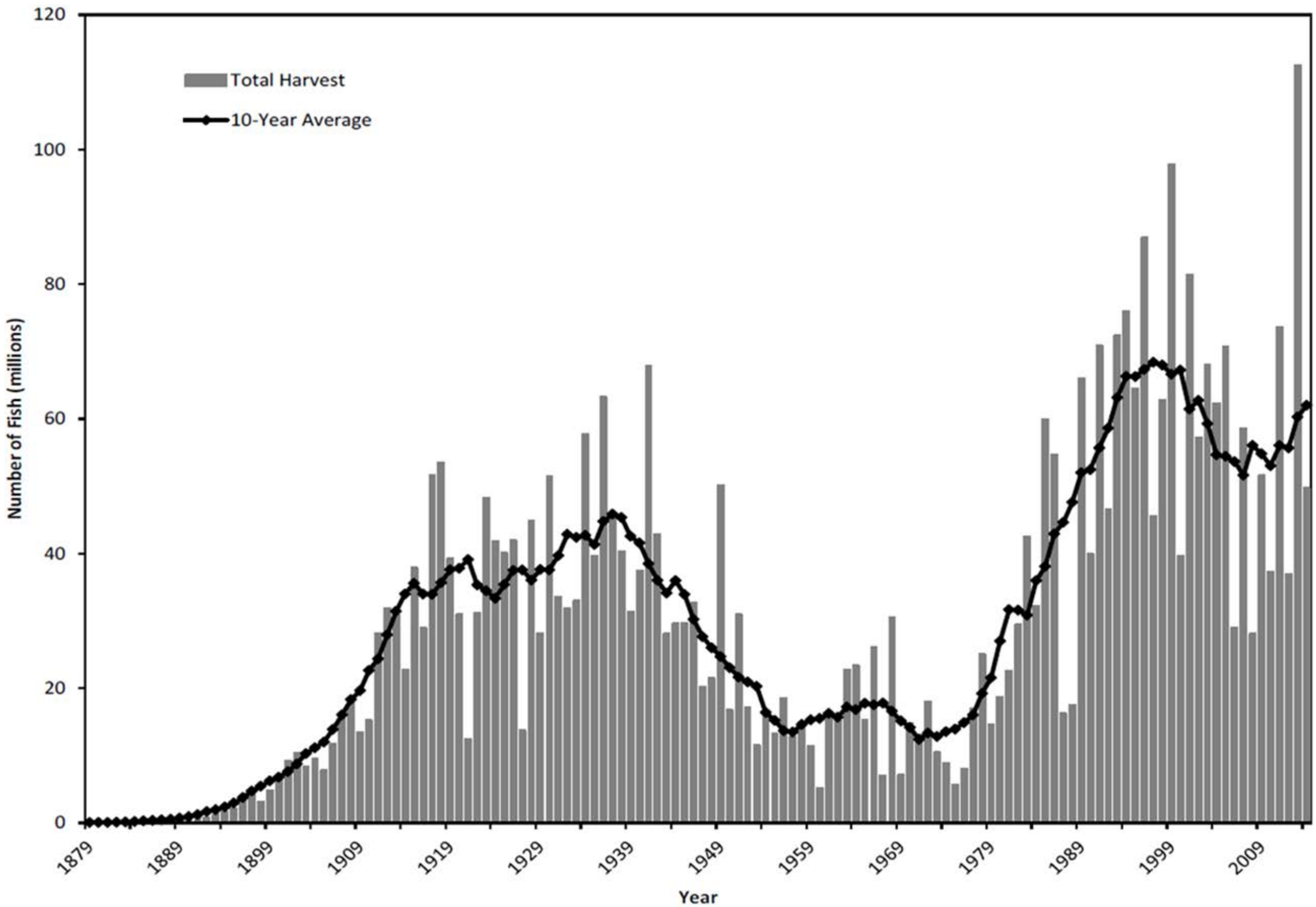

Figure 116. Region 1 (Southeast Alaska and Yakutat) historical salmon harvest and recent 10-year average harvest, from 1878 to 2014 (Conrad and Gray 2014).

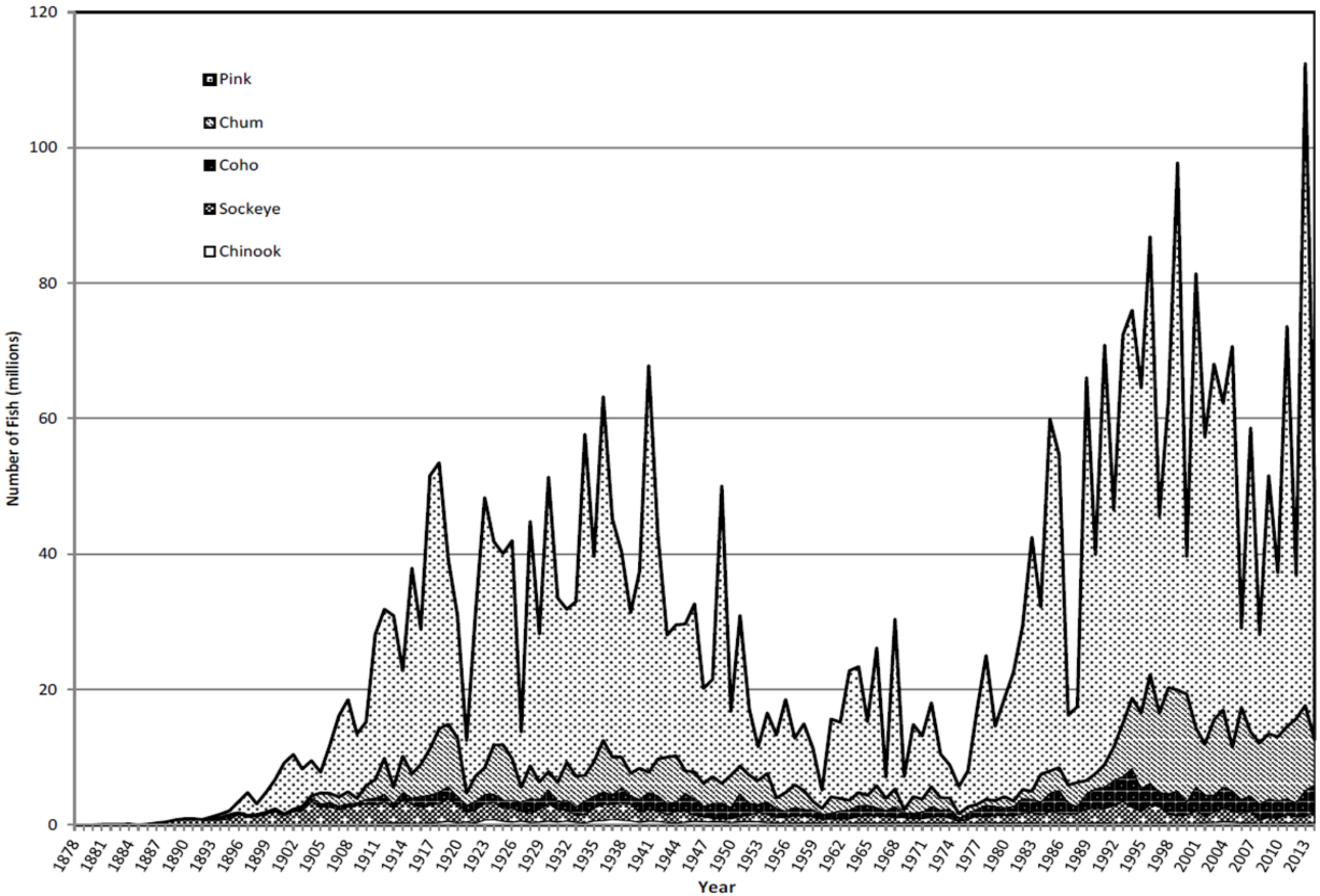

Figure 117. Region 1 (Southeast Alaska and Yakutat) historical salmon harvest by species and season, 1878-2014 (Conrad and Gray 2014).

In 2014, 49.8 million salmon were commercially harvested in the Southeast Alaska/Yakutat region; this estimate includes all gear types (Conrad and Gray 2014). The 2014 harvest estimate was approximately 44% of that reported for 2013 (Conrad and Gray 2014). Harvest estimates by species in 2014 were as follows: 428,000 Chinook (1% of total), 1.7 million sockeye (3%), 3.8 million coho (8%), 6.7 million chum salmon (13%), and 37.2 million pink (75%) (Conrad and Gray 2014).

Purse seine gear accounted for the vast majority of 2014 removals (by numbers of fish), contributing approximately 75% of total salmon harvest (Conrad and Gray 2014). Drift gillnetting (10%) and troll (6%) fisheries were much less significant. While trolling accounted for only 6% of the total salmon harvest in 2014, it represented approximately 83% of all chinook harvest and 59% of coho harvest (Conrad and Gray 2014, Skannes et al. 2015).

There were 428,000 Chinook salmon harvested in 2014, which was above average when compared to both the 10-year average (Figure 118) and the 53 year long-term average (300,000) for the species. The 2014 Chinook harvest was the third highest harvest on record for the Southeast Alaska/Yakutat region (Conrad and Gray 2014; Figure 117). Approximately 355,000 (83%) Chinook salmon were harvested by trolling. This was considerably above both the 10-year (236,622) and historic averages (249,049) for troll harvest.

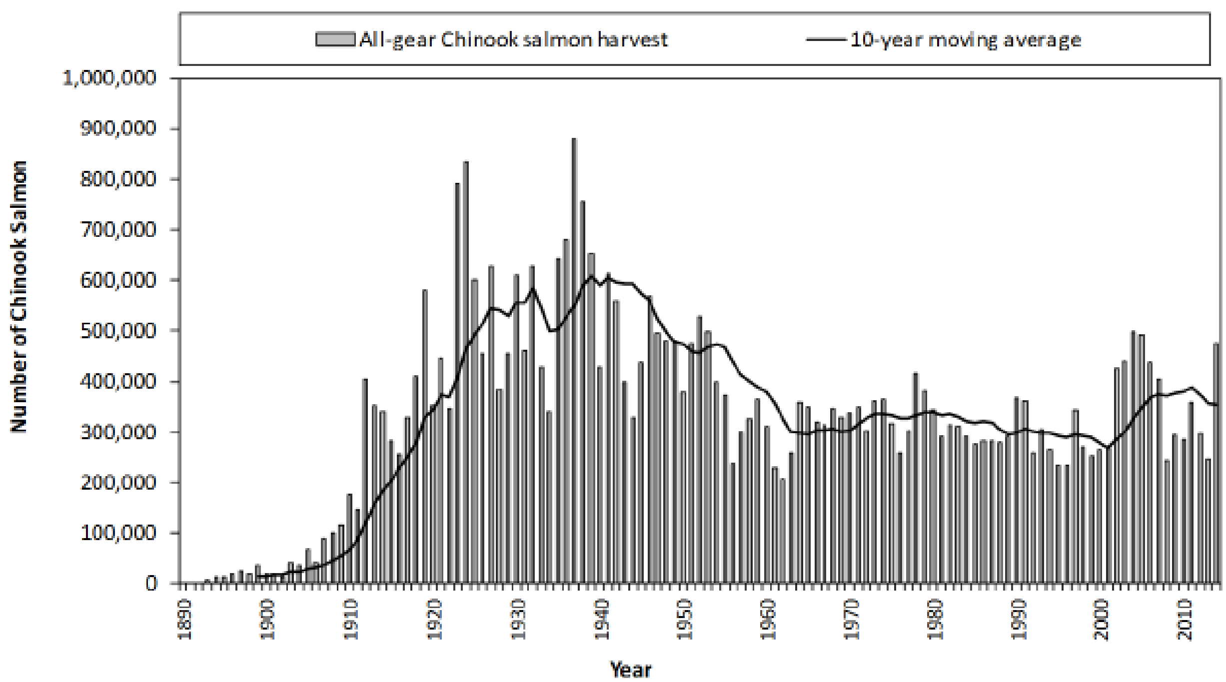

Figure 118. All-gear harvests of Chinook salmon in common property fisheries in Southeast Alaska/Yakutat region, 1891-2014 (Skannes et al. 2015).

Sockeye salmon harvest in 2014 was approximately 1.7 million fish, which is above both the 10-year average harvest (1.2 million) and the long-term harvest average (1.3 million) for the species (Conrad and Gray 2014). The 2014 sockeye harvest was the 14th largest harvest in the region since 1962. Most sockeye salmon in the Southeast Alaska/Yakutat region were harvested in the purse seine fishery

(54%), followed by the drift gillnet fishery (30%), and the set gillnet fishery (7%) (Conrad and Gray 2014).

Approximately 3.8 million coho salmon were harvested in the Southeast Alaska/Yakutat region in 2014 (Conrad and Gray 2014). This ranks as the third largest harvest since 1962 (Figure 119). The 2014 harvest was above the 10-year average of 2.6 million for the region and well above the historic harvest average of 2.1 million. The majority of coho salmon harvest in this region (2.2 million, 59%) was attributed to the troll fishery in 2014, which was nearly double the historical harvest average for this gear type of 1.2 million.

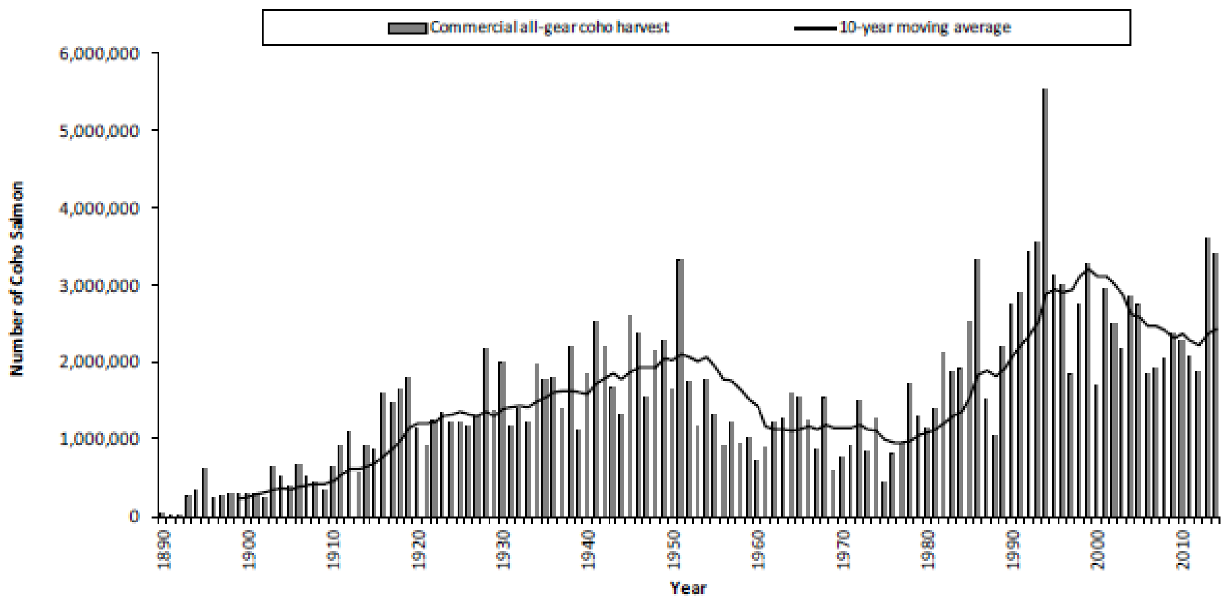

Figure 119. Commercial all-gear harvests of coho salmon in common property fisheries in the Southeast Alaska/Yakutat region, 1890-2014 (Skannes et al. 2015).

The 2014 Southeast Alaska/Yakutat region pink salmon harvest was 37.2 million fish, which was the highest of any salmon species harvested in the region during this season. Harvest in 2014 was below the 41.4 million 10-year regional average. But pink harvest in 2014 was still above the 31 million historic average (Conrad and Gray 2014). The 2014 harvest ranked 21st out of 53 years. The vast majority of pink salmon were harvested in the purse seine fishery. Conrad and Gray (2014) noted that, pink salmon exhibit a pattern of strong odd-year harvests and weak even year harvests.

Chum salmon harvest in the Southeast Alaska/Yakutat region increased beginning in the mid-1980s, with annual harvest prior to 1984 averaging 1.6 million salmon. Hatchery production of chum became significant in 1984 and regional harvest estimates increased accordingly. The 2014 6.7 million chum salmon harvest was below the 10.5 million 10-year average but was above the 5.8 million 53 year historical average. The 2014 chum harvest was the 21st highest harvest in the region since statehood. Regional chum harvest was approximately equal between the purse seine (36%) and driftnet fisheries (36%), while the hatchery fishery contributed 24% of the harvest (Conrad and Gray 2014).

Chinook harvest from fishers who possess LAPs occurs during the winter troll season in GBP. There are no Chinook salmon spawning runs in GBP, so the fish harvested in this fishery are likely winter feeding fish from other areas. Currently, approximately 45 salmon troll LAPs exist that cannot be passed on. Fisher age-based modeling results suggest the GBP LAP fishery will cease by 2050 or later when participants are unable or no longer choose to fish (Soiseth, written communication, 2015). GBP Chinook harvest and effort estimates are available from 2002-2013. The number of boats fishing in GBP has ranged from 10 (2003) to zero (2011-2013), and annual harvest estimates have never exceeded 300 fish in a season (Figure 120). Data for two seasons (2005/2006, and 2006/2007) are confidential, as fewer than three boats fished in GBP. But 25 or less fish were probably harvested.

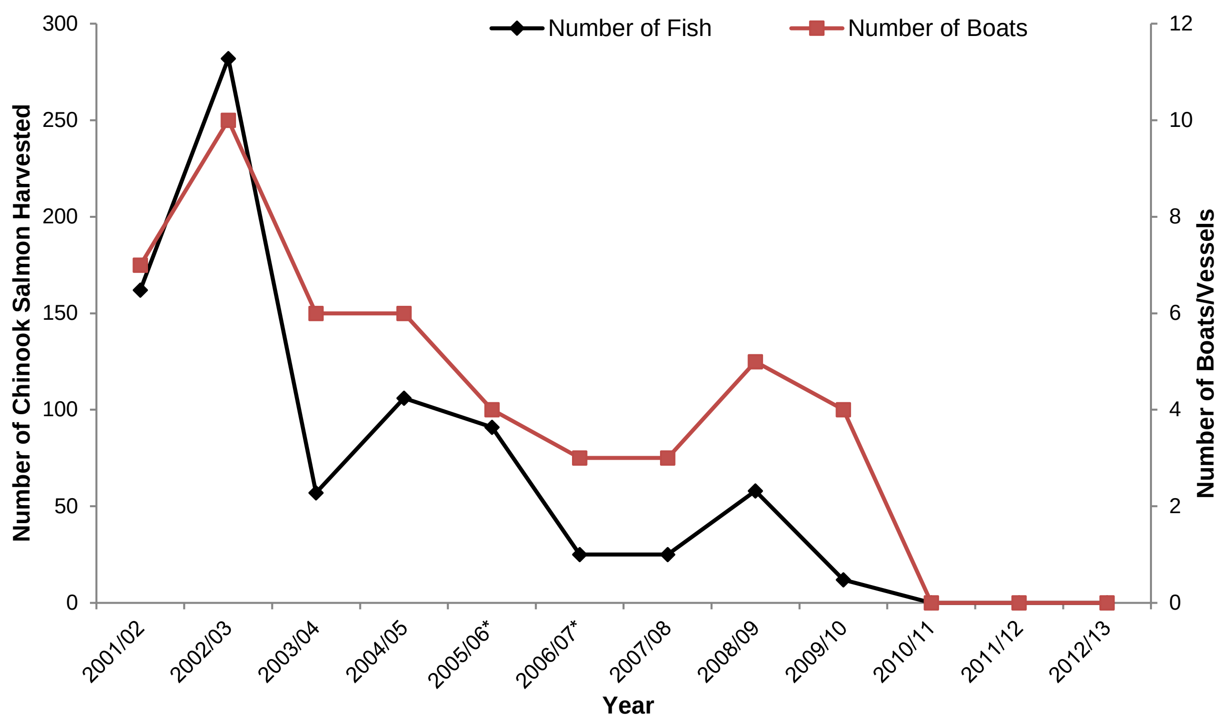

Figure 120. Lifetime Access Permit holder harvest estimates for Chinook salmon in Glacier Bay Proper during the winter troll season from 2001-2013. Note: * indicates years where three or fewer boats fished in the Bay. These data are confidential under state statute and 25 or fewer fish were presumed harvested (ADFG Data).

Aerial survey and harvest data suggest Excursion River chum runs were larger in the 1960s and 70s relative to more recent years (Eggers and Heinl 2008). Harvest averaged 95,000 fish from 1960 to 1981 in years when openings were authorized but more recently (1982-2008) harvest has averaged only about a third of this (30,000 fish, Figure 121). Peak aerial survey estimates of escapement averaged 20,000 sockeye from 1960 to 1981 but, similarly, were reportedly reduced two thirds (to 7,000 fish) in successive years (1982-2008). Reported fall-run chum salmon harvest within Excursion Inlet (statistical area 114-80) during 2007 was 18,149 fish. ADFG has established a sustainable escapement goal of 4,000-18,000 index spawners for the Excursion River late-run chum salmon stock. An index spawner count is the peak aerial survey count for the Excursion River system.

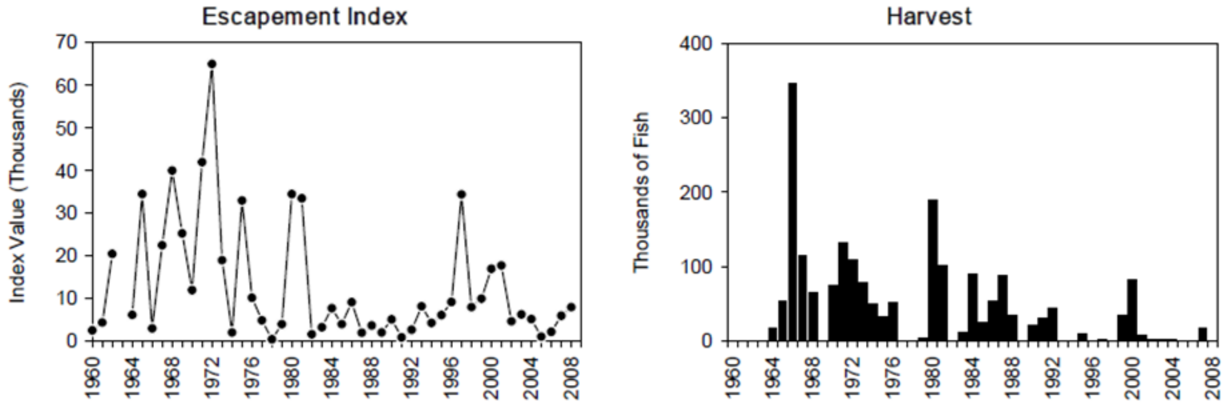

Figure 121. Annual escapement index and harvest of fall-run Excursion River chum salmon in the Excusion Inlet seine fishery (Eggers and Heinl 2008).

East Alsek sockeye production increased markedly through the 1980s resulting in a corresponding increase in average commercial harvest from 25,000 fish annually in the 1970s to more than 100,000 fish per season during the 1981-1994 period (Heinl et al. 2011; Figure 122). ADFG established a peak aerial survey biological escapement goal of 26,000-57,000 sockeye in 1995 but this was subsequently revised downward to 13,000-26,000 fish in 2003 as a consequence of declining stock productivity (Clark et al. 2003). In more recent years, the fishery has periodically not opened (1999-2002, 2008 & 2010) due to an inability to achieve escapement goals (Woods and Zeiser 2013a). Average sockeye harvest during the five year 1994-1998 period was 37,500 sockeye (Woods and Zeiser 2013a). Subsequently, the average sockeye harvest over the three fishery openings during the 2007-2011 period was approximately 17,100 fish (Woods and Zeiser 2013b). More recently, reported sockeye harvest during the 2012 opening was just over 12,100 fish with a peak escapement of 16,000 fish observed on July 22.

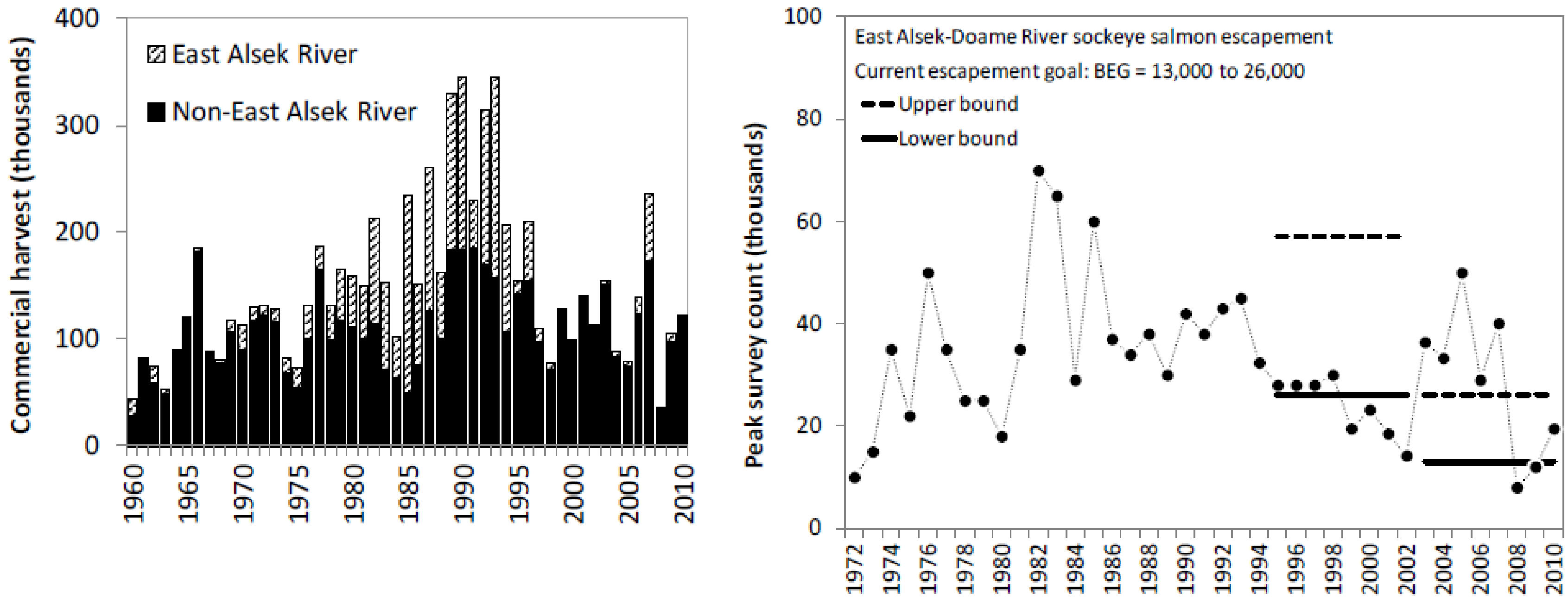

Figure 122. Commercial sockeye set gillnet harvest in the East Alsek/Doame River and other Yakutat area streams (left). East Alsek/Doame River sockeye escapement index (peak aerial survey counts) and two subsequent escapement goals (upper and lower bounds) (Heinl et al. 2011).

Sport Harvest

Based on angler survey data provided by the ADFG, 2013 sport harvest in the Bartlett River accounted for 1,447 salmon removals. Fishing effort in 2013 consisted of approximately 459 anglers and over 1,140 days fished (ADFG 2013). Sockeye and coho salmon were the species harvested in the greatest numbers from 1997-2013 (Figure 123), with pink and chum salmon harvested in low numbers. The 2013 Bartlett River sport harvest was estimated at 135 sockeye salmon and 1,168 coho salmon. Coho harvest in 2013 was well above the 7-year (nonconsecutive) average of 455 salmon. The 2013 sockeye harvest was nearly double the 7-year average of 73 fish.

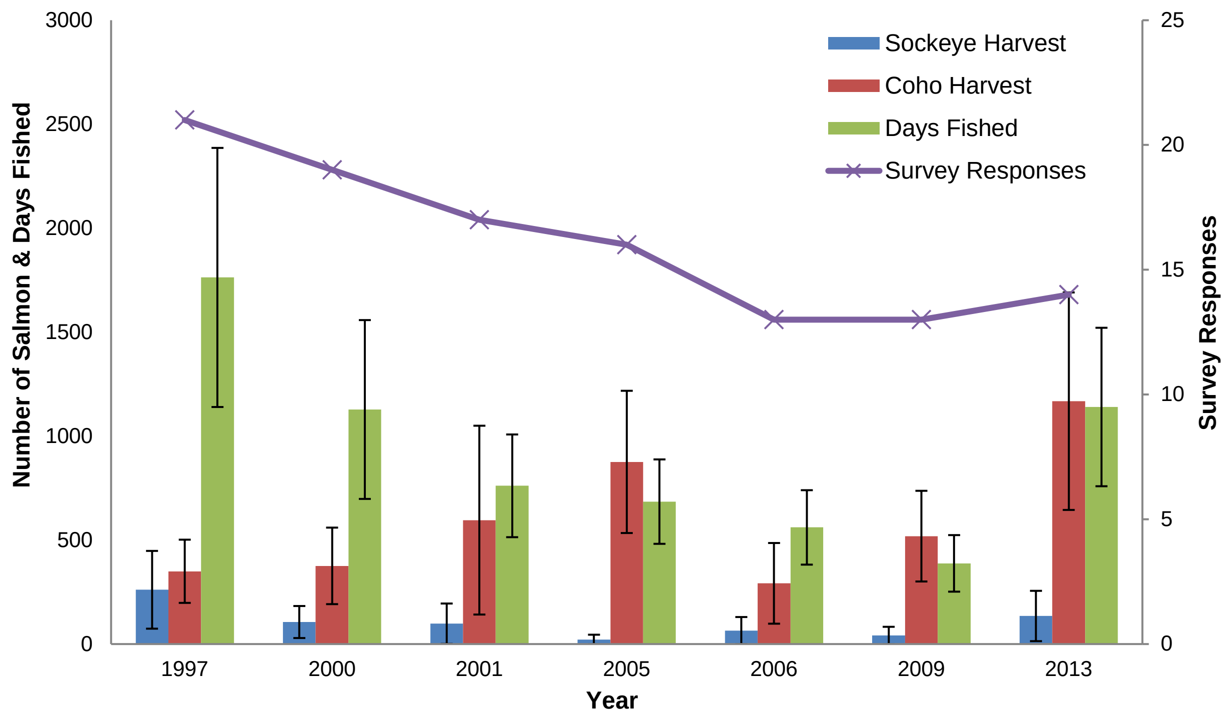

Figure 123. Bartlett River sportfish harvest and effort, 1997-2013. Note, only years where ≥11 people responded are included in this figure (ADFG 2013).

State-authorized Subsistence and Personal Use Harvest

<u>*ADFG (2013)*</u>

State-authorized subsistence and personal use harvest occurs primarily in the vicinity of four rivers in GLBA: Excursion River, East Alsek River, Dundas River, and North Berg River. ADFG (2013) reports subsistence/personal use harvest and effort for all anadromous fish in GLBA from 1985-2011. Participation in this activity is currently quite limited (Table 53) as NPS regulations (36 CFR 2.3 (d) (1)) prohibit fishing in freshwaters in any manner other than by hook and line. State authorized gear for these fisheries include gaffs, spears, beach seines, dip nets, drift gillnets, and cast nets. While five anadromous fish species are harvested in these rivers, only sockeye and chum salmon were harvested in great numbers (Table 53), and the average harvest and number of permits in each river is presented in Table 53.

Table 53. Average state-authorized subsistence and personal use harvest and harvest effort for anadromous fish in four GLBA rivers from 1985-2011 (Data from ADFG). Harvest doesn't necessarily occur in these systems during each year.

River	# Permits	Chinook	Sockeye	Coho	Pink	Chum
North Berg	2	0	27	0	21	1
Excursion	3	0	2	4	10	582
East Alsek*	3	4	129	2	15	2
Dundas	1	0	26	0	3	1

* Federal subsistence salmon harvest is legally authorized only within the East Alsek River located in the preserve.

Sockeye salmon subsistence/personal use harvest has occurred in greatest numbers in the East Alsek River, with annual harvest estimates ranging from 30 (1990) to 335 (1994) (Figure 124). Estimated harvest was 113 sockeye salmon in 2011 by four permit holders, which is a slightly below the long-term average harvest for this river.

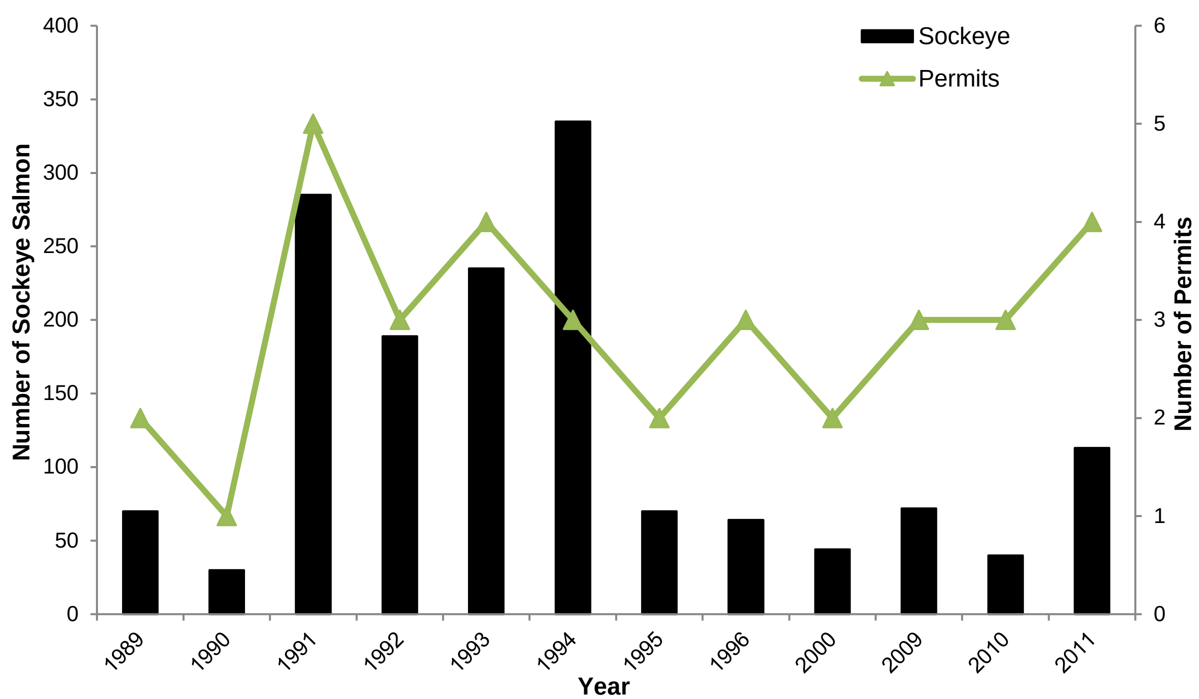

Figure 124. East Alsek River subsistence/personal use sockeye salmon harvest, 1989-2011 (ADFG 2013).

Sockeye harvest in the North Berg River has been lower on average compared to the East Alsek River, with an average state-authorized subsistence/personal use harvest of 27 salmon per year (Table 53). Annual harvest estimates (excluding years when no sockeye were harvested) have ranged from two (1995) to 150 fish (1990). Sockeye harvest in the most recently reported year (2010) was zero; only one permit was issued that year. The most recent year with harvest occurring in the river was 2009, when six sockeye were harvested (Figure 125).

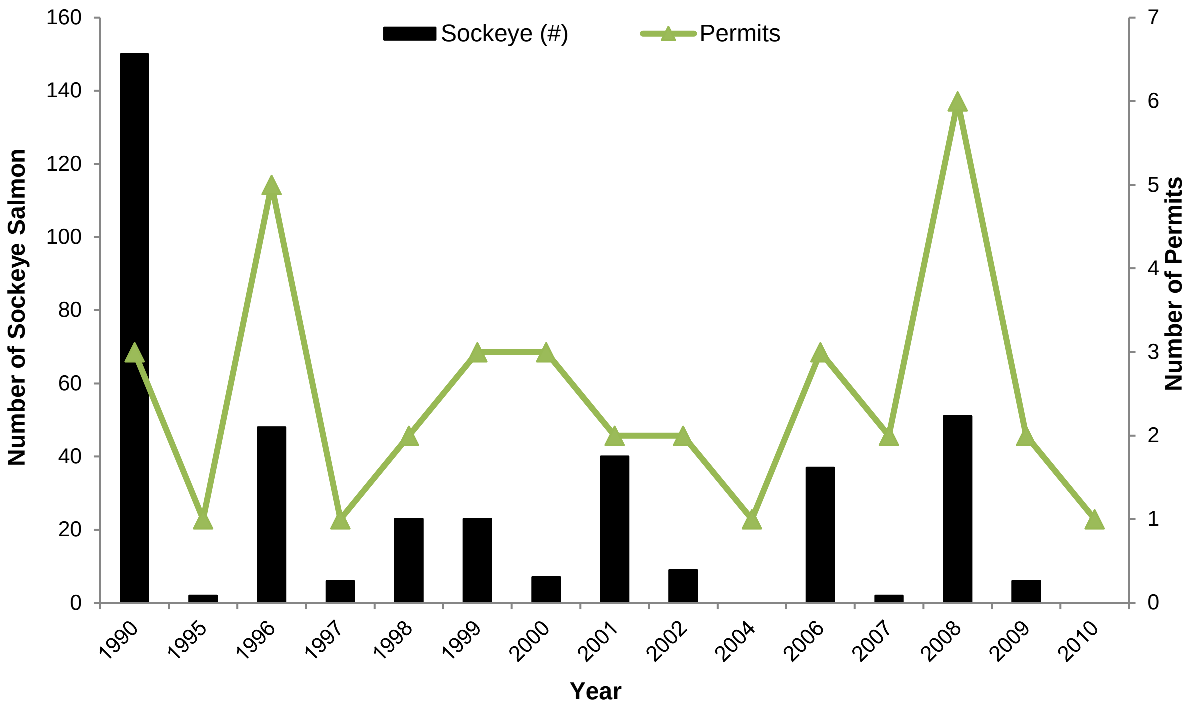

Figure 125. North Berg River subsistence/personal use sockeye salmon harvest, 1990-2010 (ADFG 2013).

The Excursion River exhibits the highest reported state-authorized subsistence/personal use harvests of chum salmon, with the highest harvests occurring from 1985-1999 (Figure 126). Average annual chum harvest from 1989-2010 has been 582 salmon (Table 53), ranging from 7 (2000) to 1,539 (1999). Only one permit was issued in 2010, and no chum salmon were reported as harvested (Figure 126). The most recent year with harvest statistics was 2009, when 550 chum salmon were harvested.

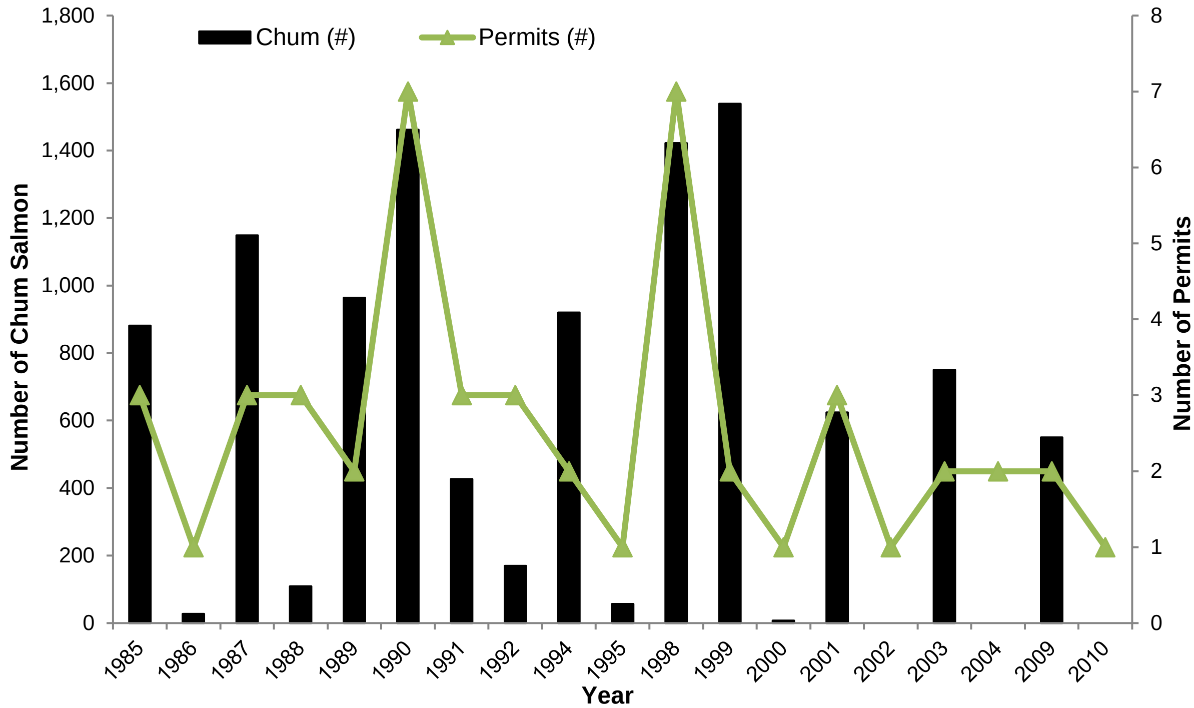

Figure 126. Excursion River state-authorized subsistence/personal use chum salmon harvest, 1990-2010 (ADFG 2013).

Threats and Stressor Factors

GLBA staff identified several threats to the anadromous fish community during project scoping, and the most pressing of these threats and stressors were harvest, climate change, ocean acidification, contaminants (especially mercury), and invasive species.

As has been discussed in depth in this document, commercial harvest occurs widely throughout the outer waters of GLBA, and on a much more limited basis through LAPs in GBP. Sport and state-authorized subsistence/personal use harvest also occur throughout the waters of GLBA although the state-authorized subsistence/personal use has historically been relatively small. ADFG annual reports on commercial troll harvests have indicated relatively stable and healthy salmon stocks in the Southeast Alaska/Yakutat region in recent years (Conrad and Gray 2014, Skannes et al. 2015). The mixed stock fishery that exists in Southeast Alaska makes management of anadromous fish difficult in this region; particularly because fish harvest cannot be attributed to a specific region or stock as the region contains thousands of salmon-producing streams and tributaries. Thus, there is no way of knowing how heavily harvested a particular stock is being harvested at when harvest occurs on these mixed stocks. Weaker (i.e., less productive) stocks in Southeast Alaska are at particular risk of overharvest, as described in NRC (1996, p. 282):

> *When fishing occurs on a mixture of populations with different stock-recruitment functions and fishing cannot be regulated at a rate appropriate for each component population, the stage is set for overfishing of the less-productive components (Ricker 1958, 1973; Hilborn 1985). For example, extinction of wild coho salmon in the lower Columbia River has*

occurred as fishing pressures at sea and in the lower Columbia increased to take hatchery returns; catch levels of 85–95% were directed at the returning fish (Cramer et al. 1991).

Climate change, and the predicted hydrological change because of it (i.e., shifts in precipitation storage from snow to runoff, increased stream temperature, etc.), has the potential to be the most pressing issue and challenge to the overall health of Pacific salmon in the 21st Century (Shanley and Albert 2014). General climate change models indicate predicted increases in air temperatures ranging from 1-5°C (33.8° 41°F) during the next 100 years (Hengeveld 1990, Bryant 2009). Predicted temperature increases in Southeast Alaska, especially in the winter months when precipitation could change from snowfall to rain, could have dramatic landscape-wide effects on watersheds (Hodgson and Quinn 2002, Mote et al. 2003, Bryant 2009).

Salmon are affected by hydroclimatic factors at every life stage (Shanley and Albert 2014), and an alteration in temperature or flow regime due to climate change could have dramatic impacts on salmon in Southeast Alaska. Temperature has the largest effect on egg development rates in freshwater habitats (Quinn 2005). In fact, a review of current literature by Shanley and Albert (2014) theorized that, under current climate change scenarios, egg-to-fry lifestage survival may be the most important limiting factor for salmon productivity in Southeast Alaska freshwater habitats. Additional hydroclimatic factors that could influence salmon survival and productivity include winter flow extremes removing salmon eggs from stream substrates (e.g., scouring) or completely altering or removing nesting habitat from freshwater streams (Mantua et al. 2010). Salmon fry and juveniles require appropriate stream flows to maintain high quality habitat to mature in (Quinn 2005); and at maturity, global ocean circulation patterns, prey availability, and appropriate stream flows and temperatures are required for successful rearing, migration and spawning (Quinn 2005).

Ocean acidification (OA) represents another climate-change driven threat to the anadromous fish of Southeast Alaska. The world's oceans absorb CO_2 from the atmosphere, with most sources of CO_2 coming from the burning of fossil fuels and cement production (Raven et al. 2005). As CO_2 is absorbed, chemical changes occur in the ocean that result in the overall pH decreasing (i.e., making it more acidic). If global emissions of CO_2 continue to increase based on current trends, the average pH of the oceans could fall by as much as 0.5 units by the year 2100 (Raven et al. 2005). The pH of the oceans has already decreased from 8.17-8.09 over the past 200 years (Haufler et al. 2010). Species that are dependent upon calcium carbonate availability are directly impacted by OA, as the amount of calcium carbonate available to organisms decreases as the oceans become more acidic. Declines in these calcium carbonate dependent species (e.g., shellfish, mollusks, crustaceans, pteropods) could have indirect impacts on anadromous fish communities, as several of these species represent critically important food sources for salmon and other species of fish. (PMEL 2008, Haufler et al. 2010).

Mercury is a metal that is entering national parks in Alaska through atmospheric deposition from local, regional, and trans-Pacific sources (Landers et al. 2008). Mercury is an elemental pollutant with a complex life cycle in the atmosphere and biosphere, which leads to some difficulty in detecting its origin (Landers et al. 2008). Anthropogenic sources (e.g., combustion, smelting, and petroleum refining) are thought to account for 75% of the mercury that enters the atmosphere, with the remainder originating from geologic and biogenic sources (Landers et al. 2008).

In water, mercury is often converted to methylmercury (MeHg), a neurotoxin 100 times more toxic than elemental mercury (USGS 2000). MeHg biomagnifies in the aquatic food web and can be transferred to new environments through animal movements (i.e., “biovectors”). For example, developing salmon can accumulate MeHg and other contaminants as they grow in pelagic waters and ultimately deposit them into their riverine spawning environment (Quinn 2005, Baker et al. 2009). A detailed discussion of mercury and other heavy metals and contaminants in GLBA is presented in the Water Quality component (Chapter 4.22) of this NRCA.

Invasive species also represent a threat to the anadromous fish communities of GLBA. Of primary concern to GLBA managers are Atlantic salmon and the New Zealand mudsnail (*Potamopyrgus antipodarum*). The Atlantic salmon is an anadromous fish species native to the Atlantic Ocean that has been cultured in fish farms and ocean-based net pens in British Columbia and Washington since the 1970s (Schrader and Hennon 2005). Several thousands to hundreds of thousands of Atlantic salmon have escaped (and continue to escape). They periodically appear in commercial saltwater catches and occasionally in freshwater catches (ADFG 2002a). Atlantic salmon have been caught in Southeast Alaska in both commercial and recreational fisheries, and have been documented in Icy Strait and the Doame River in the Dry Bay preserve. Reproduction of Atlantic salmon has been documented in some freshwater streams of British Columbia (Volpe et al. 2000, ADFG 2002a), and the establishment of Atlantic salmon in Southeast Alaska could represent a threat to other native salmonid species such as the steelhead, cutthroat trout, Dolly Varden, and coho salmon (ADFG 2002b).

New Zealand mudsnail is another invasive species that potentially threatens the anadromous fish of Southeast Alaska. This snail species was introduced to the U.S., and occurs in nearly all of the Western U.S. (USFWS 2004, MSU 2005). The New Zealand mudsnail colonizes freshwater environments very rapidly, and often takes over and dominates the invertebrate community of the freshwater sources where it is present. The mudsnail dominates these habitats as a consumer, ingesting large portions of the food resources and outcompeting and overcrowding all native species (Schrader and Hennon 2005). Their colonization alters food chain/web dynamics of streams and rivers and can outcompete and dominate salmonid prey species. This species is not yet confirmed in Alaska, but it is easily transported via sport fishing gear (e.g., boots, waders) and by boating equipment (Schrader and Hennon 2005).

Myxobolus cerebralis is a salmonid parasite that causes a disease commonly referred to as “whirling disease.” This affliction has caused considerable health problems in both hatchery and wild salmonid populations, with symptoms including whirling behavior in fish, blackened tails, and skeletal deformities (Gilbert and Granath 2003). The disease affects juvenile fish in higher numbers as the fish’s bones are not yet fully calcified, and heavy infections in juveniles are often accompanied by increased mortality (Gilbert and Granath 2003). Whirling disease was primarily a disease found in hatchery fish, with rainbow trout being among the hardest-hit of all species. However, beginning in the 1990s, the disease began to spread to wild salmon populations across the Western U.S. and continues to spread today. Whirling disease has only been observed in Alaska in a hatchery in Anchorage in 2006 (Arsan et al. 2007), with other instances of the disease either undocumented or unobserved.

Data Needs/Gaps

While the park has a strong history of research regarding stream succession and fish colonization (Milner et al.'s many publications), there exist large gaps in knowledge on basic salmonid species distribution and abundance in most park streams. Environmental DNA sampling may be one method that could greatly increase what is known about the distribution of anadromous fish in the park. Estimates of abundance are often challenging to obtain and tend to be highly variable. It may not be feasible to obtain abundance estimates from park streams on a regular basis, but monitoring several small index streams may be a less intensive option to monitor salmonid abundance over time. Invasive species monitoring is also absent in GLBA streams. Environmental DNA monitoring may also be a viable method for periodic surveys for invasive species such as Atlantic salmon or New Zealand mud snail.

Additionally, the terminal salmon fisheries that exist in the GLBA area, including the Dry Bay East Alsek set gillnet sockeye fishery and the Excursion Inlet fall chum salmon seine fishery, are not discussed in depth in this document. These fisheries are managed by monitoring run strength and ensuring escapement is met through aerial surveys. Because of this, estimates of removals may not be complete for the region as they are currently depicted in this assessment.

Overall Condition

Distribution

During project and component scoping, the *Significance Level* for distribution was determined to be a 3. Few studies have monitored the distribution or abundance of the various salmonid species in the park, although very limited distribution information exists for many freshwater streams (ca. 229 of the over 300) in the park (Soiseth and Milner 1995). The general perception among park researchers and managers is that salmon distribution, although clearly dynamic with respect to stream colonization and the potential for large scale physical change affecting access, has remained relatively unchanged over the last several decades in the park is as one should expect. There are no known extirpations of any stocks in the park, and stream habitat is generally pristine, with new habitats continuing to become available as glacial retreat occurs. Figure 114 provides a presence and absence visual representation of anadromous fish distribution in the park, with distribution being more sparse in the newly formed streams and habitats. There is no current cause for concern regarding anadromous fish distribution in the park, but additional research and monitoring could be used or potential declining trends. A *Condition Level* of 0 was assigned to this measure.

Removals

A *Significance Level* of 3 was assigned to the removals measure. The mixed stock nature of many of the area salmon fisheries plus the lack of detailed stock health information for nearly all salmon populations within the park pose a substantial challenge in assessing current condition of the GLBA stocks. Area fisheries (commercial [all gear types], sport and subsistence/personal use) clearly harvest some component of GLBA stocks, but the relative proportion of GLBA fish harvested compared to other stocks cannot be determined. Salmon fisheries are managed by the ADFG, with annually established harvest guidelines for mixed stocks and escapement goals established periodically for terminal fisheries. Harvest thresholds, minimum escapement goals relatively well-managed fisheries and protected pristine spawning and rearing habitat allow for viable salmonid populations in GLBA waters. However, removals of fish from these fisheries continues to be of concern to park managers, and the uncertainty of the actual proportion of GLBA stocks harvested is troublesome at times. The potential to overharvest weak stocks exists in mixed fisheries like Southeast Alaska troll fisheries, and the variation in harvest trends and patterns per year warrants some level of concern. A *Condition Level* of 2 was assigned to this measure.

<u>Weighted Condition Score</u>

A *Weighted Condition Score* of 0.33 was assigned to the anadromous fish component. This score is at the upper threshold of the good condition designation, and managers should continue to monitor the overall condition of this resource to determine if adjustments in condition need to be made in the future. A trend was not assigned to this component due to the overall uncertainty regarding the actual proportion of GLBA stock that are harvested, and the limited amount of information that exists for the distribution measure. Because of these uncertainties, a border indicating medium confidence was selected.

Anadromous Fish			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.33**
Distribution	3	0	
Removals	3	2	

4.16.6. Sources of Expertise

- Chad Soiseth, GLBA Fisheries Biologist
- Craig Murdoch, GLBA Fisheries Biologist

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4.17. Mid-Trophic Level Marine Forage Community

4.17.1. Description

The mid-trophic level forage community in GLBA includes many species of fish such as Pacific sand lance, capelin, eulachon, herring, juvenile walleye pollock, northern smooth tongue (*Leuroglossus schmidti*), northern lampfish (*Stenobrachius leucopsarus*), and juvenile Pacific salmon (Photo 42; Springer and Speckman 1997). Many of these species are often collectively referred to as “small schooling fish” (SSF) or “forage fish” (Renner et al. 2012). SSF are preyed upon by many species of mammals, birds, and other fishes. They are critical in the transfer of energy from primary producers to the higher trophic levels, including humpback whales, halibut, harbor seals, and numerous marine bird species (Photo 43; Springer and Speckman 1997).

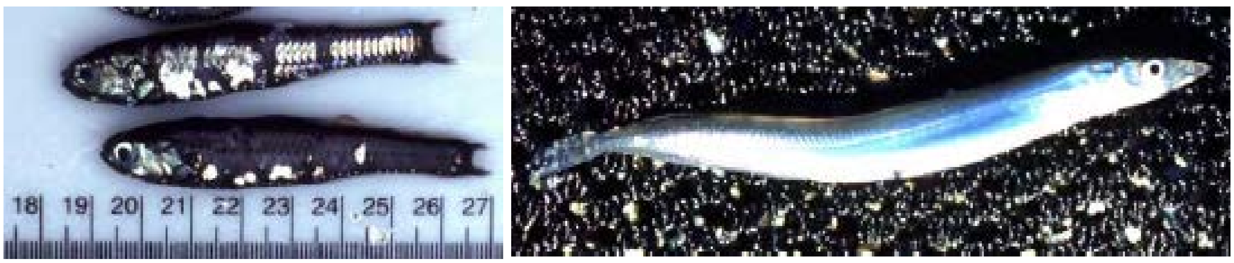

Photo 42. Examples of the mid-trophic level forage community from GLBA: northern lampfish (left) and young-of-the-year Pacific sand lance (right) (USGS photos).

Photo 43. Humpback whales feeding on members of the mid-trophic level forage community (photo in public domain).

Fish are not the only group of animals that make up the mid-trophic level forage community in GLBA. Jellyfish (*Haliclystus* spp.), euphausiids (i.e., krill), and cephalopods are all prey species that are considered members of GLBA’s mid-trophic level forage community. Many of these organisms are considered “zooplankton.” Zooplankton are the primary food source for many fish, whales, and seabird species in the marine environment, and directly influence their predators’ distribution and survival (Robards et al. 2002, 2003). Knowledge of zooplankton and SSF abundance and distribution is a key to understanding the larger ecological community that is the marine environment in GLBA (Robards et al. 2002, 2003).

The diets of the mid-trophic level marine forage community vary by species and with age within species. Smaller community members, whether small species or the young of larger species, typically feed on smaller food items such as phytoplankton, diatoms, copepod eggs and nauplii (crustacean larvae) (Springer and Speckman 1997). As the forage community members grow, food items increase in size and may include other small fishes as well as animals of their own species (Springer and Speckman 1997).

Glacier Bay Proper is a unique, tidally influenced estuary that experiences well-defined stratification during the summer months (Robards et al. 2002, Etherington et al. 2007). The bay splits into two arms with input from glaciers feeding freshwater into both of the upper arms, causing them to behave more like traditional estuaries (Robards et al. 2002, 2003). GBP is approximately 100 km (62 mi) in length and varies in width from 4–8 km (2.5–5 mi) in the lower Bay, with 15 km (9.3 mi) widths in the mid-Bay (Hooge and Hooge 2002). Both arms of GBP have depths that exceed 300 m (984 ft), with a maximum depth of 458 m (1,502 ft) found in the west arm (Hooge and Hooge 2002). Unusually high and sustained levels of productivity within Glacier Bay, when compared to other fjords in the area, were documented by Hooge and Hooge (2002). These levels of productivity are believed to be driven by frequent water mixing and renewal events (Hooge and Hooge 2002). The mixing and renewal events distribute nutrients, allowing for an abundance of phytoplankton and zooplankton which supply the energy base for the large variety of predatory species within GLBA (Robards et al. 2002, 2003).

4.17.2. Measures

- Species-specific spatial distribution of forage species
- Density of forage species
- Biomass of forage species
- Depth of forage species
- Changes in species composition over time

4.17.3. Reference Conditions/Values

Due to a lack of current studies and historic data, reference conditions have not been defined for the mid-trophic level forage community component. In general, the trophic linkages within the Glacier Bay marine food web are poorly understood at this time, so it is difficult to identify meaningful reference conditions (Chris Sergeant, SEAN Ecologist, written communication, 9 November 2015). Robards et al. (2002, 2003) and Arimitsu et al. (2007) identify much of the information pertinent to the measures; however, the data used in these studies are now 10-15 years old. The current condition of this resource will be determined, if possible, by the best professional judgment of the identified park experts.

4.17.4. Data and Methods

Springer and Speckman (1997) attempted to refine and further delineate an actual definition for the term "forage fish," as well as identify and clarify their role in the food web. Forage fish stock variability and management, the effect of fluctuating biomass and density on predators, and the importance of forage fish to Alaska's ecosystem were all discussed. The dependence of the higher trophic levels on the presence or absence of forage fishes was the focus.

Hooge and Hooge (2002) studied the oceanographic processes within Glacier Bay. The study made use of data collected between 1992 and 2000 using a conductivity-temperature-depth probe (CTD) capable of recording depth, temperature, salinity, light penetration, amount of sediment, and the amount of phytoplankton. Hooge and Hooge (2002) focused on the oceanographic reasons for Glacier Bay's unusually high productivity among Alaskan fjords. Etherington et al. (2007) further described the oceanographic characteristics of Glacier Bay and their influence on biological productivity.

Robards et al. (2002, 2003) investigated primary and secondary production, SSF, and marine bird and mammalian predators within GBP and Icy Strait from 1999-2000. Hydroacoustic surveys collected with a Biosonics DT4000 echosounder combined with predator and trawl surveys were the primary methods of data collection. While this study constitutes most of the research available for this component, the data are now more than 15 years old and may no longer be representative of GBP.

Arimitsu et al. (2007) sampled the GLBA marine waters between 1999 and 2004 to characterize marine predator and forage fish resources. Using advanced very high resolution radiometer satellite imagery (AVHRR), Arimitsu et al. (2007) analyzed sea surface temperatures (SST) in relation to the distribution and abundance of midwater-schooling forage fishes. Trawl sampling, temperature, salinity, chlorophyll-a, and turbidity were measured and examined for any relation (ANOVA) between these oceanographic measures and species occurrence.

Renner et al. (2012) conducted a study of the marine predator and prey communities within Glacier Bay. This study tested two competing hypotheses: 1) predator communities depend on and shape the community composition of their prey, 2) predator communities are structured more directly by their physical environment rather than their prey community. Simultaneously utilizing trawl nets, visual surveys, and electronic sampling, Renner et al. (2012) sampled 87 locations, collecting data on the lower, mid, and upper-level trophic communities in addition to the oceanographic conditions.

4.17.5. Current Condition and Trend

Species-Specific Spatial Distribution of Forage Species

Species composition, distribution, and frequency of SSF varied by collection method within Glacier Bay (beach seine or trawl) (Robards et al. 2002, Arimitsu et al. 2007). Robards et al. (2002) presents a more spatially detailed analysis for beach seine collected samples that includes temporal, geographic, and species-specific data (Table 54).

Table 54. Species composition of small schooling fish beach seine samples from Glacier Bay (Robards et al. 2002).

Location	June 1999	June 2000	July 2000	August 2000
Lower Bay	Pink salmon (85%)	Pink salmon (47%)	Walleye pollock (74%)	Sand lance (32%)
				Pink salmon (21%)
				Great sculpin (18%)
		Unidentified sculpins (41%)	Great sculpin (13%)	Pacific herring (7%)
	Unidentified sculpins (11%)			Dolly Varden (5%)
		Dolly Varden (6%)	Dolly Varden (9%)	Kelp greenling (4%)
				Silverspotted sculpin (3%)
Middle Bay	Sockeye salmon (51%)	Pink salmon (77%)	Pink salmon (46%)	Pacific herring (87%)
	Pacific sand lance (19%)		Great sculpin (13%)	
	Pink salmon (12%)	Rock sole (11%)	Pacific herring (11%)	
	Coho salmon (6%)		Butter sole (10%)	
			Slender eelblenny (5%)	Pacific sand lance (9%)
	Pacific herring (3%)	Unidentified sculpins (8%)	Pacific sand lance (4%)	
			Coho salmon (3%)	
West Arm	Unidentified sculpins (37%)	Unidentified sculpins (52%)	Capelin (44%)	Capelin (73%)
	Pacific sand lance (26%)	Pink salmon (10%)	Slender eelblenny (30%)	
				Slender eelblenny (17%)
	Slender eelblenny (18%)	Pacific sand lance (9%)	Great sculpin (11%)	
		Slender eelblenny (7%)		
	Walleye pollock (9%)	Butter sole (7%)	Rock sole (10%)	Pacific sand lance (4%)
		Capelin (5%)		
East Arm	Unidentified sculpins (34%)	Slender eelblenny (29%)	Pacific herring (42%)	Pacific sand lance (83%)
		Dolly Varden (22%)	Great sculpin (19%)	
	Slender eelblenny (34%)	Sockeye salmon (15%)	Rock sole (14%)	
	Pacific sand lance (15%)	Unidentified sculpins (9%)	Capelin (14%)	
		Butter sole (6%)		Capelin (7%)
	Dolly Varden (12%)	Pacific sand lance (6%)	Slender eelblenny (3%)	
		Coho salmon (5%)		

Robards et al. (2002) also performed a similar species distribution analysis on trawl captured samples. Figure 127 represents the distribution of prominent mid-trophic level community members captured in the 1999 sampling effort. The sampling results from 2000 are similarly represented in Figure 128.

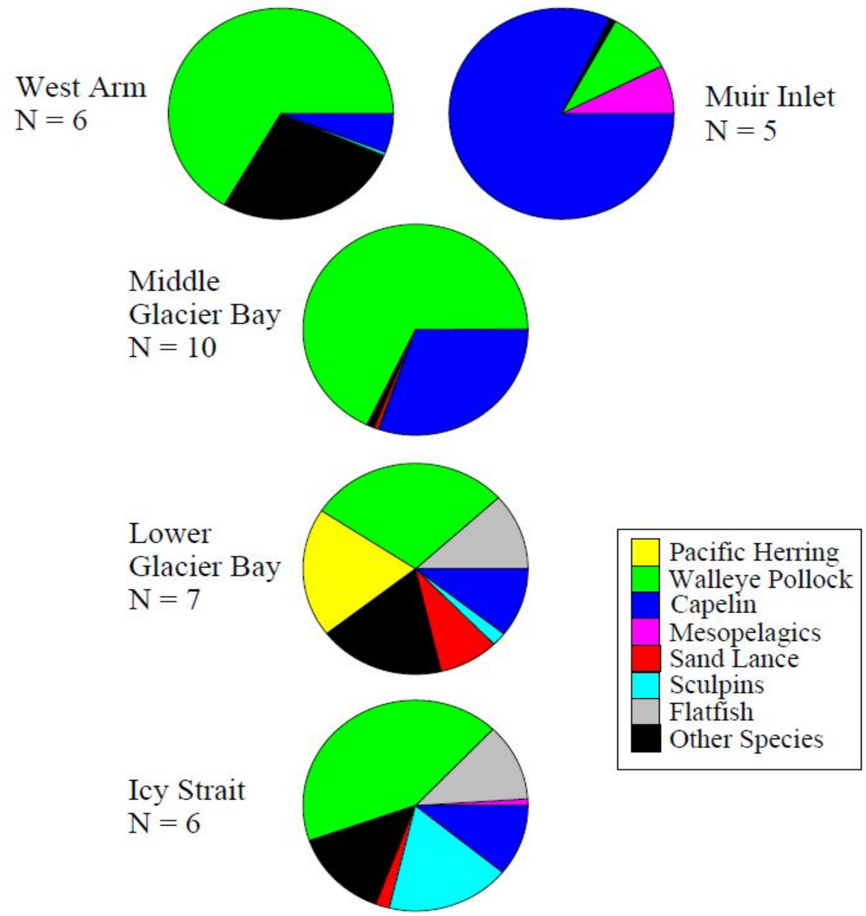

Figure 127. Distribution and composition of prominent mid-trophic level community species captured by mid-water trawl in June 1999 (Robards et al. 2002).

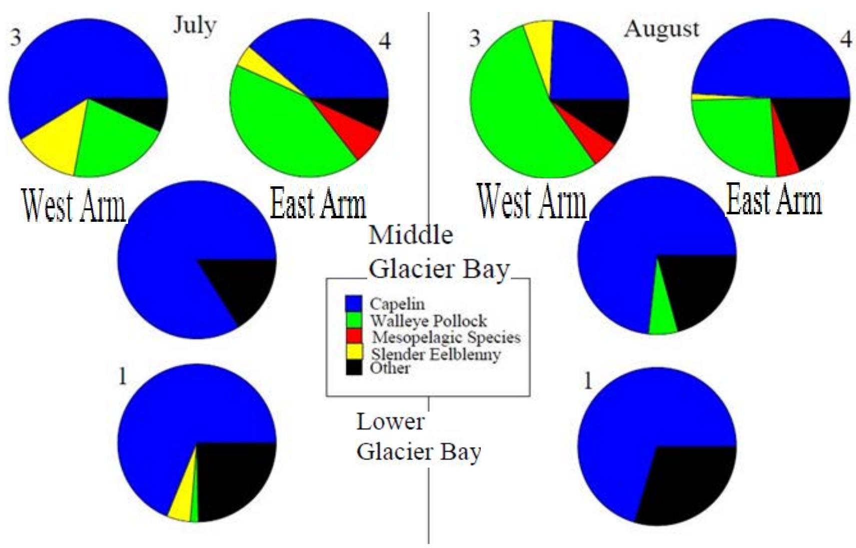

Figure 128. Distribution and composition of prominent mid-trophic level community species captured by mid-water trawl during July and August 2000 (Robards et al. 2002).

Pacific herring were captured frequently and in large quantities in the Middle and East Arm of Glacier Bay during July and August of 2000 (Robards et al. 2002). For safety and in the presence of an abundant food supply, herring are known to congregate in shallow water during summer and fall months. While they were present throughout GBP, they are believed to become more prominent in the later summer months of July and August (Figure 129, Robards et al. 2002). Unlike other coastlines adjacent to the park, herring are not known to spawn at any observable density within GBP (Robards et al. 2002).

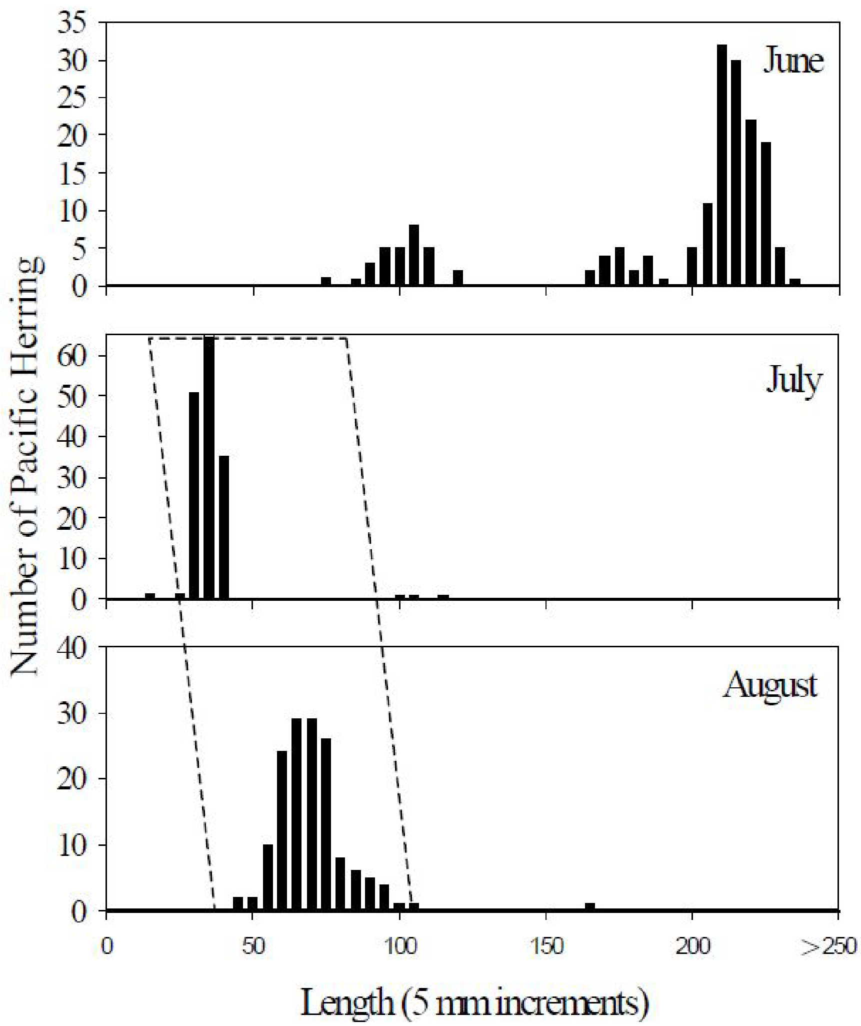

Figure 129. Length/Frequency distribution of Pacific herring caught during 1999 and 2000. Young-of-the-year fish indicated by dashed line (Robards et al. 2002).

Arimitsu et al. (2007) recorded larval herring in 14% of their trawls while adult herring were collected in 25% of trawls. Herring were encountered in relatively low numbers in GBP, but their abundance increased just outside the park boundaries in Icy Strait (Arimitsu et al. 2007). Robards et al. (2002) noted that juvenile herring may also be adept at avoiding capture by nets or may simply be missed due to their patchy distribution. This may explain the species' absence within the Bay during summer sampling, but their presence during fall sampling.

Robards et al. (2002) found walleye pollock and capelin in abundance and widely distributed throughout GBP. The number and distribution found are presented in Table 55 and Table 56 (Robards et al. 2002). Arimistu et al. (2007) further segregated pollock into three size classes based on fork length: <30 mm (1.2 in), 31-60 mm (1.2-2.4 in), and 110-180 mm (4.3-7.1 in). Larval pollock (<30 mm) were captured in 46% of their sample trawls, 31-60 mm pollock were the most abundant and captured in 37% of trawls, and finally 110-180 mm pollock were captured in only 12% of trawls.

Table 55. Number of walleye pollock larvae per 1,000 m^3 (35,315 ft^3) captured via trawl (Zeroes omitted for clarity-all stations were sampled every month) (Robards et al. 2002).

	Lower Bay		Middle Bay		West Arm			East Arm			
Station #	**00**	**02**	**04**	**06**	**08**	**10**	**21**	**14**	**16**	**18**	**20**
May	–	–	–	3.9	–	–	–	–	–	–	–
June	–	3.8	49.9	187.5	158.2	491.3	–	68.3	86.1	–	–
July	–	0.1	–	–	4.1	1.0	3.6	–	–	7.9	7.9
August	–	–	–	–	0.1	1.0	35.3	0.4	0.8	2.4	4.2

Table 56. Number of capelin larvae per 1,000 m^3 captured via trawl (Zeroes omitted for clarity-all stations were sampled every month) (Robards et al. 2002).

	Lower Bay		Middle Bay		West Arm			East Arm			
Station #	**00**	**02**	**04**	**06**	**08**	**10**	**21**	**14**	**16**	**18**	**20**
May	–	–	–	–	–	–	–	–	–	–	–
June	–	4.4	15.6	–	10.9	244.9	–	–	8.4	–	–
July	1.9	3.6	–	2.4	0.1	0.5	24.0	0.1	–	8.4	6.0
August	0.9	3.5	0.4	0.8	0.8	0.4	26.5	7.6	8.7	4.9	0.6

While pollock were the most abundant species captured via trawl, capelin were also a dominant species (Robards et al. 2002). Arimistu et al. (2007) reported larval capelin captured in 69% of trawls with adult capelin captured in 54%.

Some distinct spatial patterns of distribution were observed between capelin and pollock throughout GBP. In the 1999 sampling season (June), walleye pollock dominated the samples in the lower and middle bay as well as the west arm. While capelin were present in those areas, they dominated the east arm around Muir Inlet (Robards et al. 2002, Arimistu et al. 2007). The capelin/pollock relationship is reversed, however, in 2000 (July–August) when capelin dominate in all areas with the exception of the West Arm in July and the East Arm in August where pollock dominate (Robards et al. 2002). These results may be related to the maturation of each species, as Glacier Bay is thought to be a nursery to both species (Arimitsu et al. 2007).

Sampling also revealed a high relative abundance of juvenile fish across species when compared to similar work done in Lower Cook Inlet (Robards et al. 2002). One possible explanation for this phenomenon is that GBP serves as a nursery for many mid-trophic level community members (Robards et al. 2002).

The young of Pacific salmon, Dolly Varden, slender eelblenny (*Lumpenus fabricii reinhardt*), sand lance, sculpins (family Cottidae), rock sole (*Lepidopsetta polyxystra*), English sole (*Parophrys vetulus*), northern lampfish, and northern smooth tongue were all captured frequently within GBP. Dolly Varden and other salmonids exhibited a preference for the east arm of the Bay. Robards et al. (2002) speculates the topography in the east arm, with its wide benches of vegetation and lake systems, is the primary reason for the preference. Slender eelblenny were widely distributed in the upper arms of the Bay. Rock sole and English sole seemed to prefer the lower Bay (Robards et al. 2002).

Both northern smooth tongue and northern lampfish were captured unusually close to the surface (10m [32 ft]) in the upper arms of GBP. Robards et al. (2002) speculates that prey resource availability in the upper water columns or high turbidity due to glacial silt may explain these species being captured at unusual depths in these locations.

<u>Density of Forage Species</u>

High-density locations of SSF and other mid-trophic level forage community members were rarely detected within the survey area of Robards et al. (2002) during their sampling period (see Table 55 and Table 56 above). Areas of acoustically determined biomass were concentrated in a few locations around GBP and generally in shallow, coastal waters. Although seabird and humpback whale abundance is currently quite high within GBP (Sergeant et al. 2014, Neilson et al. 2014), Robards et al. (2002) found that less than 8% of the total area surveyed exceeded forage fish densities suitable for seabirds (0.01 fish/m^3), and less than 1% contained densities suitable for whales (0.1 fish/m^3).

<u>Biomass of Forage Species</u>

Robards et al. (2002) conducted a hydroacoustic survey to study the biomass of forage fish and zooplankton within GLBA (Figure 130). Not surprisingly, the survey found that forage fish biomass was greatest at the heads of both the east and west arms of GBP, as these locations coincide with high concentrations of zooplankton and phytoplankton (Robards et al. 2002). This concentration is likely best explained by the freshwater input from the glaciers contributing both nutrients and energy to mix the waters in the marine environment, thus providing ideal conditions for high productivity (Robards et al. 2002). Other areas of high biomass concentration, such as Berg Bay, Beardslee Islands, South and North Marble Islands, Geikie Inlet, and Scidmore Bay were also in areas of high levels of mixing and renewal (Robards et al. 2002).

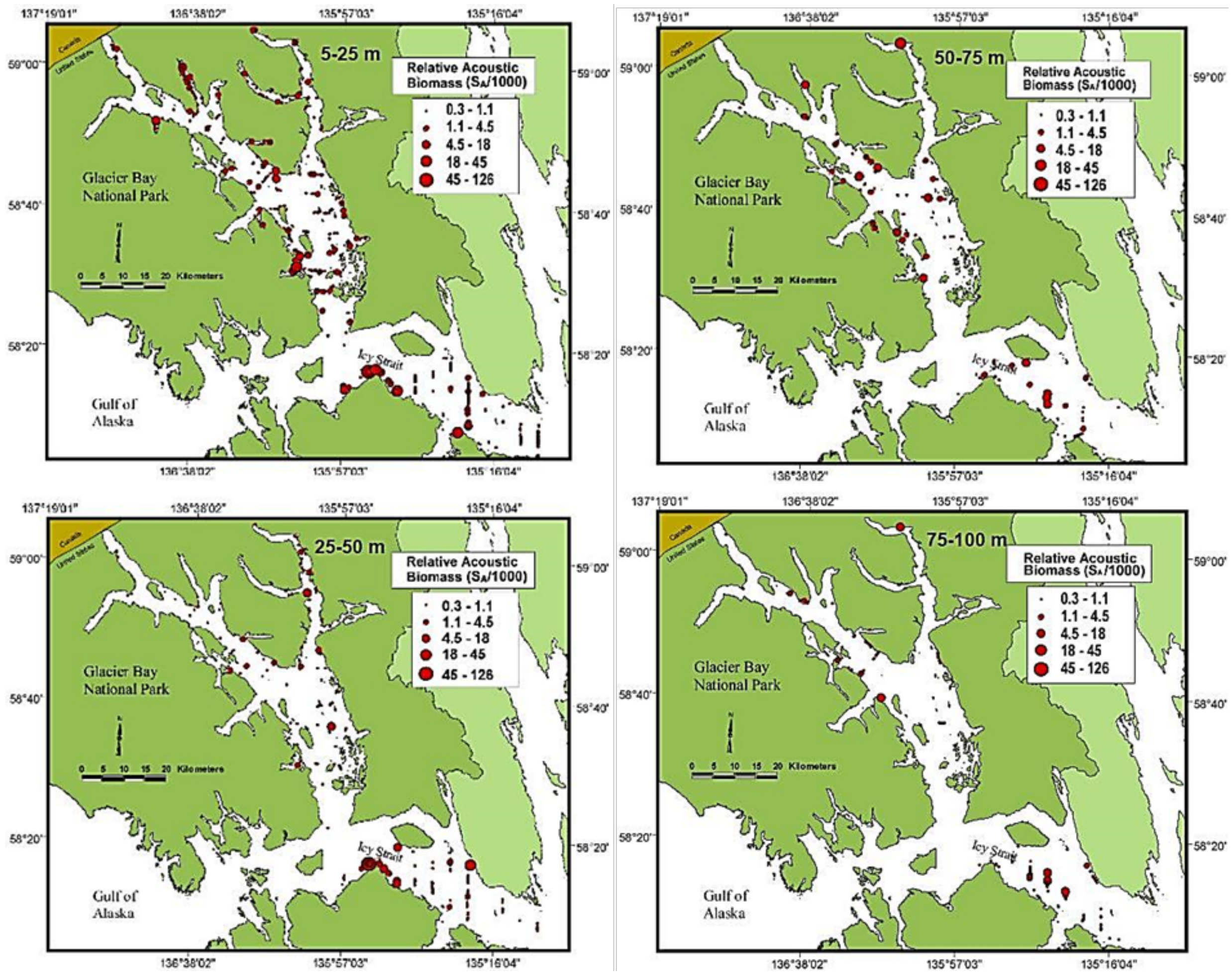

Figure 130. Distribution of fish and zooplankton biomass at various depths, as determined by hydroacoustic survey. Top left, 5-25 m depth; bottom left, 25-50 m; top right, 50-75 m; bottom right, 75-100 m (Robards et al. 2002).

Despite being a highly productive bay, biomass concentrations capable of supporting apex predators (e.g., whales) were found in very few places within GBP, and Robards et al. (2002) noted large expanses of the Bay where concentrations of forage fish are scarce or nonexistent. The majority of high-density aggregations of SSF, suitable for larger mammals, were primarily composed of capelin, pollock, and herring (Robards et al. 2002).

<u>Depth of Forage Species</u>

In June 1999, Robards et al. (2002) hydroacoustically surveyed plankton and fish and found that 50% of the acoustic biomass (primarily small schooling fish and euphausiids) were found at depths of <35 m (115 ft); 80% of the biomass was found at depths of <80 m (262 ft) and 90% of the biomass was found at depths of <100 m (328 ft) (Figure 131).

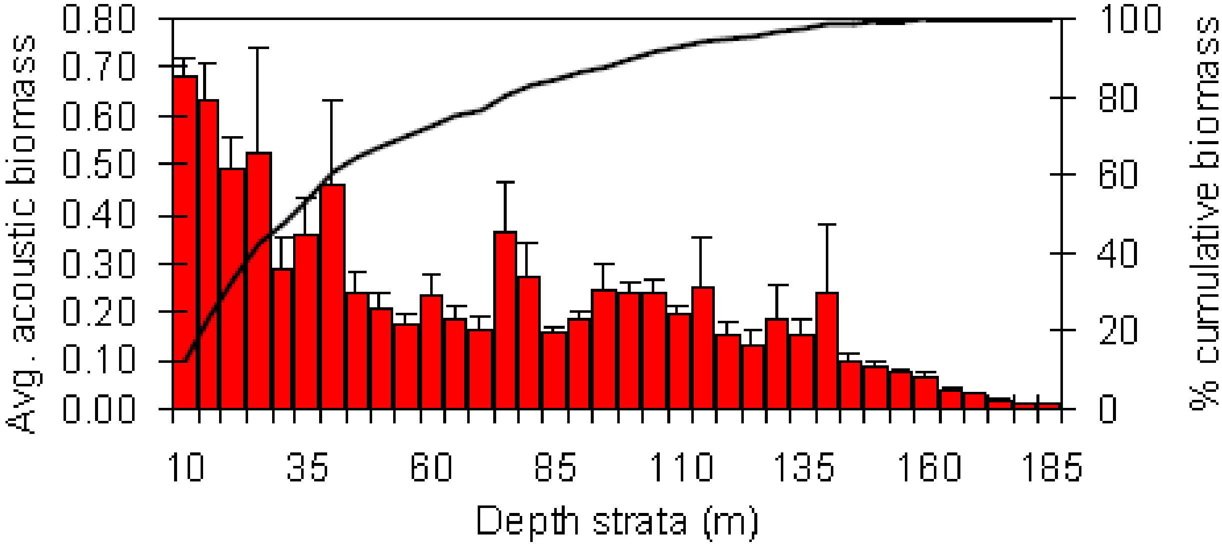

Figure 131. Average (red bars) and cumulative (line) forage fish biomass distribution by depth (Robards et al. 2002).

When depth was grouped into four categories (5-25 m [16-82 ft], 25-50 m [82-164 ft], 50-75 m [164-246 ft], 75-100 m [246-328 ft]), Robards et al. (2002) determined that the greatest biomass was located in the shallowest water layer, regardless of overall water depth. Also, when the mean densities of forage fish were calculated, Robards et al. (2002) found fish densities two to three times higher in waters <50 m (164 ft) in depth.

Changes in Species Composition Over Time

Changes in the mid-trophic level forage community's species composition need to be carefully examined by GLBA staff. Species composition shifts may indicate an individual underlying condition or a complex interaction of conditions. Identifying and understanding the direct effects of the change, as well as the underlying cause(s) are critical to management and mitigation of such changes. No long-term studies have been undertaken to document any trends in mid-trophic level forage community species richness.

Threats and Stressor Factors

GLBA staff identified several threats and stressors that are likely to affect the mid-trophic level forage community in the park: climate or ocean regime shifts (e.g., PDO), ocean acidification, increased predation, interspecific competition for habitat and food, diseases, harvest (primarily outside of GLBA), acoustic/underwater noise and availability of spawning habitat.

Marine ecosystems have been profoundly affected by climate change, with impacts at all trophic levels from plankton to apex predators (Lauria et al. 2012). Climate change is one of the major forces affecting ecosystems across the globe; this threat is becoming better understood as research and data continue to become available. Changes in the normal temperature and precipitation within the park could have both direct and indirect effects on the mid-trophic level forage community of GLBA. Examples of direct impacts could include shifts in the timing and magnitude of freshwater input into

GBP due to alterations in the timing and magnitude of snow, sea ice, and glacial melt. Change in the timing and magnitude of freshwater input and subsequent changes in circulation patterns within the Bay (Etherington et al. 2007, Hill et al. 2009) would likely alter the timing, magnitude, and spatial patterns of primary production, thus altering all marine trophic levels in GBP (Robards 2014).

The PDO is an oscillatory pattern of climate variability similar to El Niño (Mantua 2002). These variations are most apparent in the boreal winter and spring and can have significant impacts on sensitive natural resources (Mantua 2002). However, unlike El Niño events that persist for only 6-18 months, PDO events persist for 20-30 years. The effect produced by this mechanism can particularly impact the production of phytoplankton and zooplankton (Mantua 2002). Any change in production at these lower trophic levels will immediately impact the mid-trophic level forage community and ultimately impact all marine predators from seabirds to whales (Mantua et al. 1997).

Ocean acidification is a direct risk to Alaska's food chain, as well as to Alaska's commercial fisheries industry (Mathis et al. 2014). The uptake of CO_2 (into the ocean waters) reacts with calcium carbonate ions and consequently reduces those ions available in the ocean waters. Calcium carbonate ions are the same ions that some zooplankton, mollusks, shellfish, and corals use to build and maintain their shells and exoskeletons (Reisdorph and Mathis 2014). These animals play a critical role in the marine ecosystem as key prey species for the mid-trophic level community, providers of habitat for animals (corals), as well as a form of natural erosion control for shorelines and beaches (OCB and EPOCA 2010).

Increases in disease, predation, or harvest outside of GLBA would all likely lead to a decrease in the forage community population. Diseases that could impact GLBA's forage fish include viral hemorrhagic septicemia and *Icthyophonus hoferi* infection. These two pathogens are thought to be limiting the recovery of the herring population of Prince William Sound in southcentral Alaska (EVOS Trustee Council 2015). Predation may also have increased in recent decades, as the populations of two key predator species (humpback whales and Steller sea lions) have increased (Mathews et al. 2011, Neilson et al. 2014). Any decrease in the mid-trophic level forage community population would presumably lead to a decrease in the food supply to the upper trophic levels. Unless alternative sources of food or prey species were available, this impact to the upper-trophic level communities would likely result in changed animal behaviors and/or populations within GLBA (Springer and Speckman 1997). Increased competition between species within the mid-level trophic level forage community could lead to a shift in species composition within the community. This change may subsequently alter the species composition of the upper-level trophic community by altering prey availability (Renner et al. 2012).

<u>Data Needs/Gaps</u>

No long-term studies have been undertaken to document any trends in the mid-trophic level forage community. As key prey species for the large number of top-level predators in GLBA and general indicators of marine ecosystem health, changes in any of the mid-trophic level forage community's density, biomass, and distribution could be cause for concern within GLBA. In the most in-depth and direct look at the community, Robards et al. (2002) documented the community's status and distribution in 1999 and 2000. While Robards et al. (2002) is an excellent resource in assessing the mid-trophic level community, the data collected are now in excess of 15 years old and need to be utilized cautiously.

Overall Condition

Species-Specific Spatial Distribution of Forage Species

The project team defined the *Significance Level* for species-specific spatial distribution as a 3. While Robards et al. (2002) and Arimitsu et al. (2007) identified high-density locations for the community, they are dated studies. The data in Robards et al. (2002) and Arimitsu et al. (2007) are 10-15 years old and no long-term monitoring or studies have since been undertaken. This does not allow a *Condition Level* to be determined at this time.

Density of Forage Species

A *Significance Level* of 3 was assigned by the park for density. According to Robards et al.'s (2002) surveys, <8% of the total area surveyed exceeded forage fish densities suitable for seabirds (0.01 fish/m^3) and <1% contained densities suitable for whales (0.1 fish/m^3). However, the density-specific data available (in Robards et al. 2002 and Arimitsu et al. 2007) are 10-15 years old and no more recent density data have been reported. Therefore, a *Condition Level* cannot be determined at this time.

Biomass of Forage Species

A *Significance Level* of 3 was assigned by the park for biomass. Robards et al. (2002) hydroacoustically estimated biomass in many locations, but this is only a single study. Since the data in Robards et al. (2002) are now more than 15 years old and no long-term monitoring or studies have been undertaken, a *Condition Level* cannot be assigned.

Depth of Forage Species

The project team defined the *Significance Level* for depth as a 3. Robards et al. (2002) and Hooge and Hooge (2002) detail the morphology or physical structure of much of GBP. Robards et al. (2002) then identifies locations of high concentrations of phytoplankton, zooplankton, and SSF as well as the most common water depths where these concentrations are found. Although Robards et al. (2002) documented that 50% of the acoustic biomass (primarily small schooling fish and euphausiids) surveyed were at depths of <35 m (115 ft) and that 80% of the biomass was found at depths of <80 m (262 ft), it must be recognized that these data are 10-15 years old and may or may not be representative of current conditions. The lack of multiple studies along with the age of the available data (more than 15 years old) does not allow a *Condition Level* to be determined at this time.

Changes in Species Composition Over Time

Changes in species composition over time were also deemed important to GLBA staff and was given a *Significance Level* of 3. However, since no historical or current data have been collected in a manner allowing for temporal analysis so a *Condition Level* cannot be determined at this time.

Weighted Condition Score

The mid-trophic level marine forage community was not assigned a *Weighted Condition Score*, due to a lack of recent (within 15 years) data for several of the measures. Until a survey or study collects new data relating to the measures identified in this component, condition assessment would be speculative at best.

Mid-Trophic Level Marine Forage Community			
Measures	Significance Level	Condition Level	WCS = N/A
Spatial Distribution (species)	3	n/a	
Density	3	n/a	
Biomass	3	n/a	
Depth	3	n/a	
Change in Species Composition	3	n/a	

4.17.6. Sources of Expertise

- Jamie Womble, NPS Wildlife Biologist
- Chris Sergeant, SEAN Ecologist

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4.18. Pacific Halibut

4.18.1. Description

Pacific halibut are a large predatory flatfish species that are found on the continental shelf of the northern Pacific Ocean and the Bering Sea (IPHC 2014a). The geographic range of the Pacific halibut includes areas as far south as Santa Barbara, California and as far north as Nome, Alaska. The Asiatic coastline of Russia and Japan are also included in their range (IPHC 2014a). Flatfish are unique in that both eyes are located on one side of the head (Photo 44) and their skull is asymmetrical (Kramer et al. 1995).

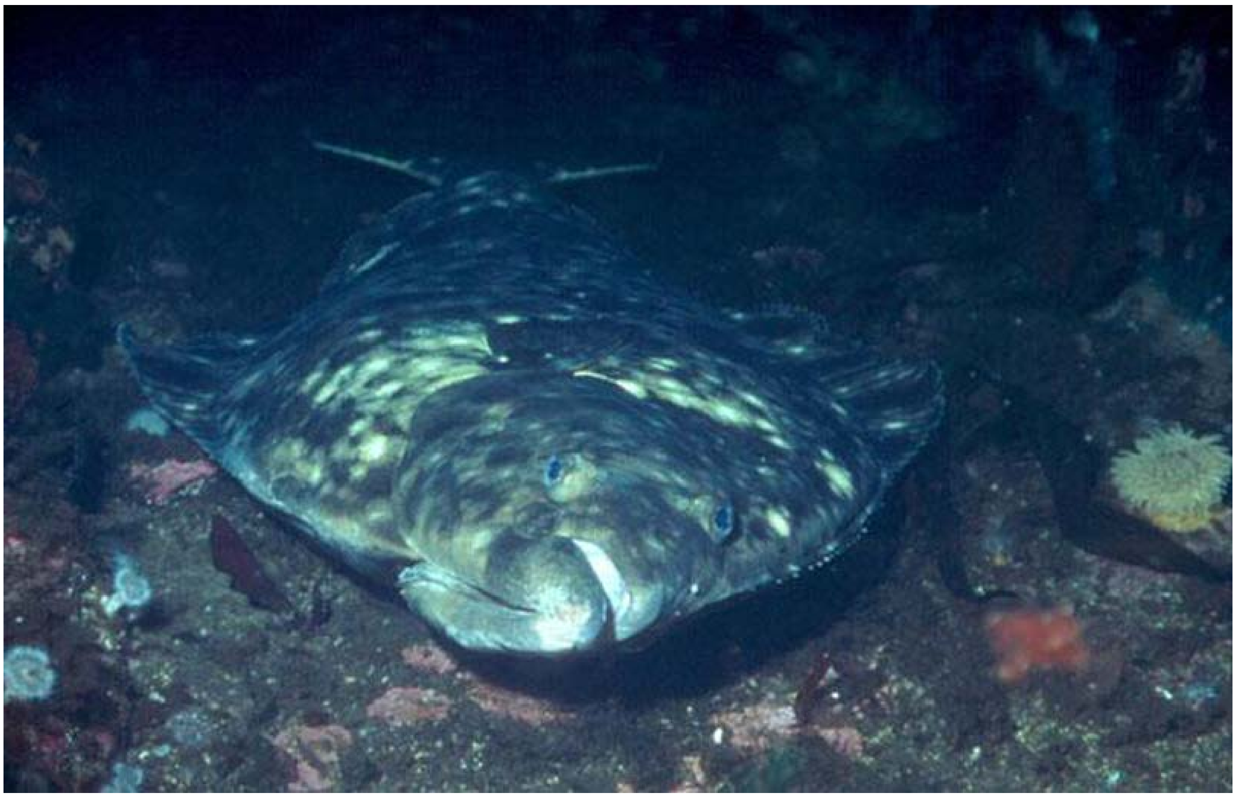

Photo 44. Pacific halibut (photo by Guy Becken, IPHC, courtesy of ADFG).

Halibut are considered a demersal (i.e., bottom-feeding) species (Kramer et al. 1995). They spend the bulk of their life in water depths between 27-274 m (90-900 ft) but have been known to be at depths as great as 1,219 m (4,000 ft) (IPHC 2014a). Halibut have a laterally compressed body and are symmetrical with light or white colored undersides and heavily pigmented (camouflaged) topsides (Photo 45) (IPHC 1987). Halibut are generally dextral, having both eyes on the right side of their body (IPHC 1987).

Photo 45. Examples of halibut coloring/camouflage (NPS photo).

Halibut feeding habits vary with different depths, areas, and seasons (Chilton et al. 1993). During the summer months, halibut are usually found in shallow waters and will move to deeper waters during the winter. According to Chilton et al. (1993), non-commercial crab and codfish (*Gadidae* spp.) are a major part of halibut diet. Other prey species include octopus, small forage fish and crustaceans. Changes in halibut diet also occur with age, as halibut tend to eat fewer crustaceans and small forage fish and more codfish and sculpin as they mature (Chilton et al. 1993).

Pacific halibut are pursued as a major commercial and sport fish fishery in Alaska. Multiple entities are intricately involved in the complex management and regulation of the halibut fishery within and surrounding GLBA. The managing entities include the International Pacific Halibut Commission (IPHC), the ADFG, the NPFMC, and the NMFS.

The commercial halibut fishery in the North Pacific Ocean has been studied and regulated by the IPHC since 1923. The IPHC estimates all removals (mortality) from the halibut population and annually produces a stock assessment (IPHC 2015a). This assessment is of the halibut in the northeastern Pacific Ocean and includes territorial waters of both the U.S. and Canada (Stewart and Martell 2014). All the halibut in those waters are treated and modelled as one population (Stewart and Martell 2014).

The IPHC divides this enormous area up into 10 different regulatory areas for management purposes. GLBA falls within Regulatory Area 3A and 2C (Figure 132, Figure 133). These IPHC regulatory areas are then further divided into many smaller areas referred to as statistical areas. In GLBA, there are five IPHC statistical areas (190, 185, 184, 182, 181; Figure 132, Figure 133). Areas 190 and 185 fall within Regulatory Area 3A; Statistical Area 185 includes all waters between the 2C/3A dividing line, which is why it is included on both Figure 132 and Figure 133. Statistical areas 184, 182, and 181 fall within Regulatory 2C. The division of statistical areas allows for more detailed analyses of halibut data. All management or regulatory decisions (i.e., catch limits, seasons, and restrictions) are implemented at the larger regulatory level (e.g., 2C, 3A) (Kong et al. 2004).

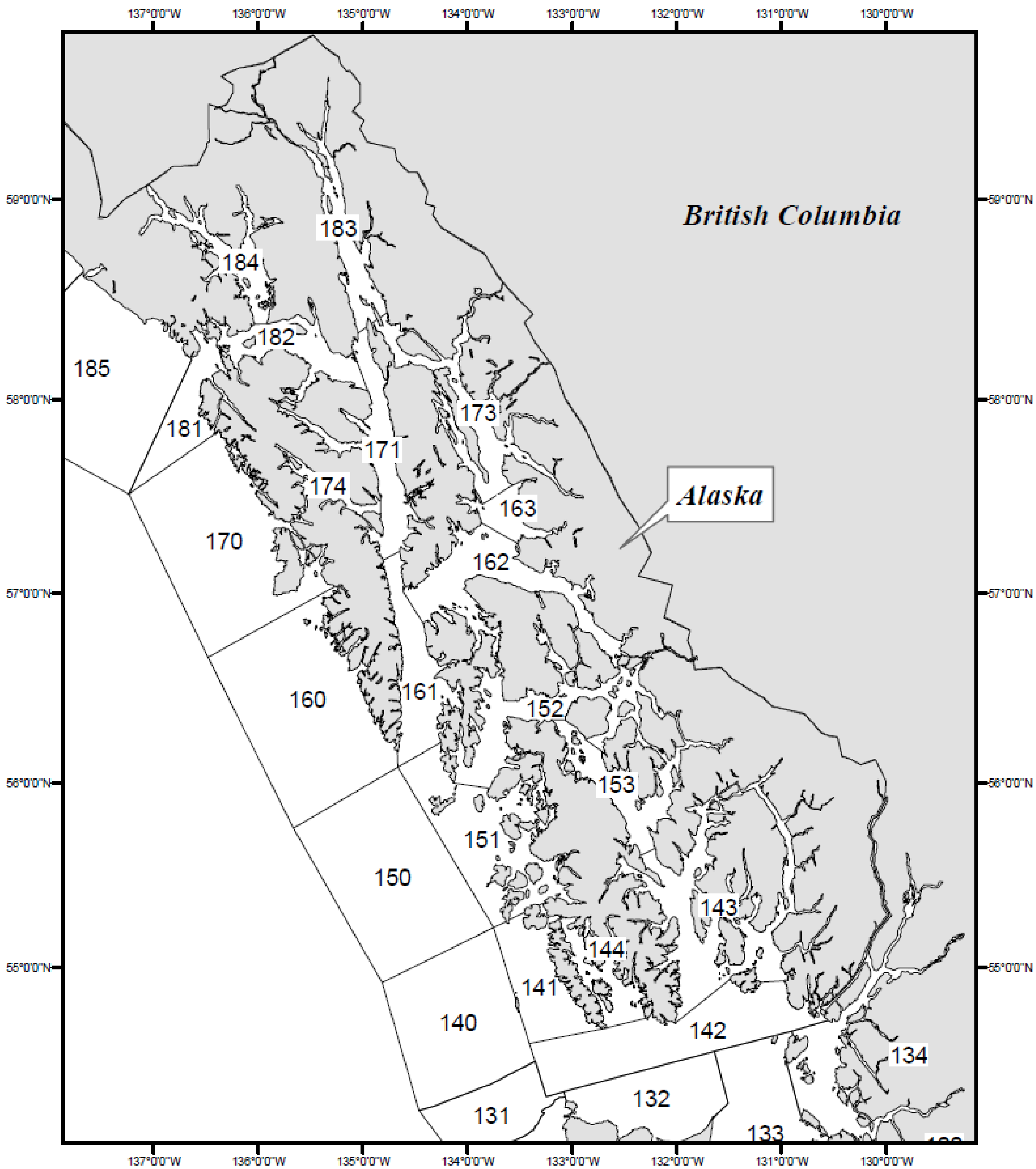

Figure 132. IPHC statistical areas of Regulatory Area 2C (includes Statistical Areas 140-184), southeast Alaska. GLBA falls within IPHC statistical areas 184, 182, and 181 in this Regulatory Area. Offshore boundaries of outer water areas typically extend 322 km (200 mi) (Kong et al. 2004).

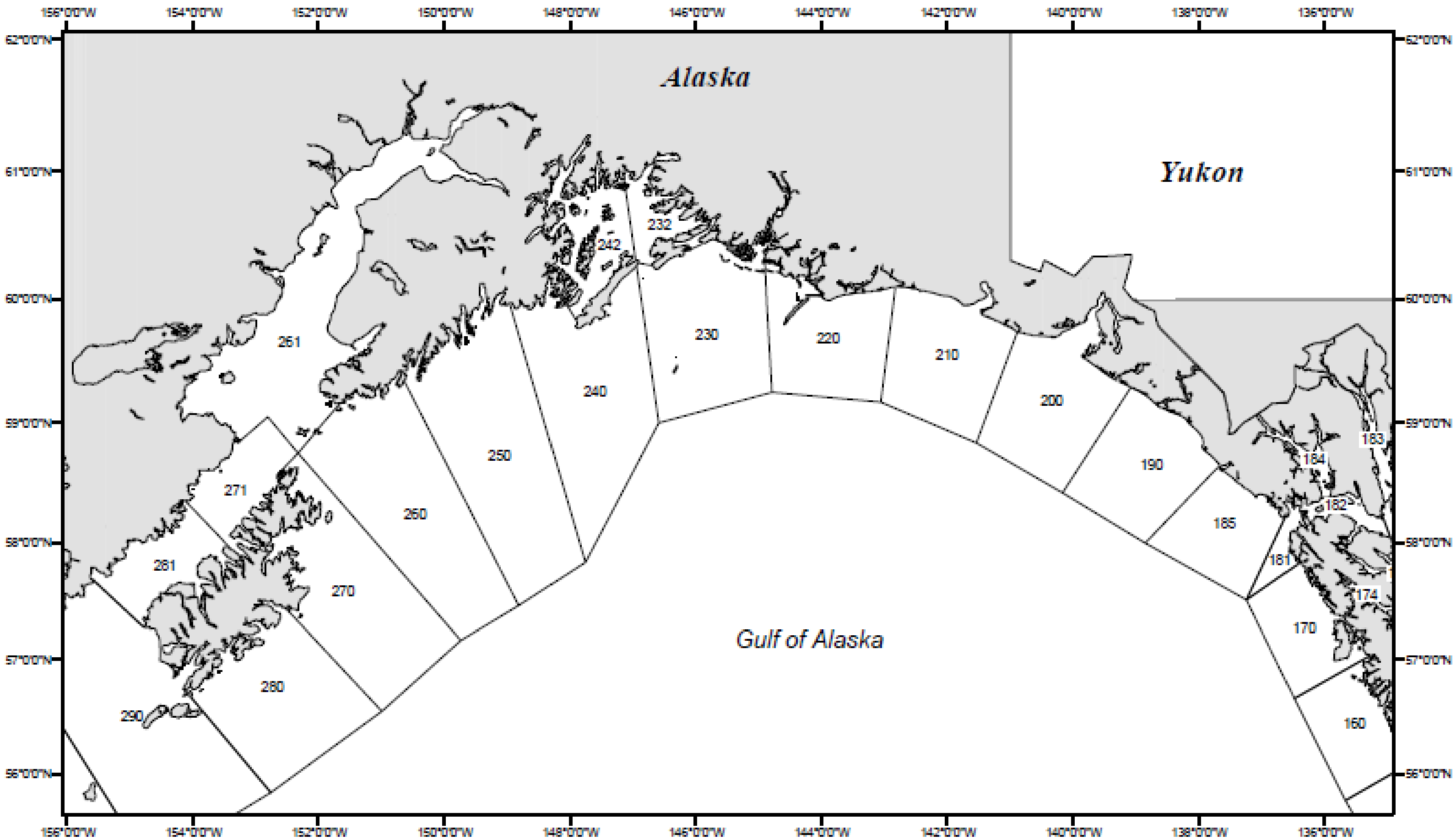

Figure 133. IPHC statistical areas of Regulatory Area 3A (includes Statistical Areas 185-281), which extends from Cape Spencer to the western end of Kodiak Island. GLBA falls within Statistical Areas 185 and 190 in this Regulatory Area. Offshore boundaries of outer water areas typically extend 322 km (200 mi) (Kong et al. 2004).

The IPHC compiles all relevant commercial halibut harvest data in addition to conducting an annual apportionment analyses. This apportionment analysis is used to distribute portions of the stock biomass estimate (calculated in the annual stock assessment) to the various IPHC regulatory areas (e.g., 2C or 3A). This appropriation then drives the catch limits assigned to the various regulatory areas (IPHC 2014a).

The ADFG compiles halibut harvest/removal data as they pertain to sport harvest. ADFG collects data via mail-in surveys and charter logbooks (guided fishing only) and reports sport harvest data by ADFG specific management area (Area G encompasses GLBA). Commercial harvest may also be reported by unique ADFG commercial groundfish statistical areas (365830, 355801, 365804, 365803, 375802, 375832), in lieu of the IPHC statistical areas (Soiseth, written communication, 2 November 2015). These ADFG groundfish statistical areas are not coextensive with the IPHC statistical areas (Figure 132, Figure 133, Figure 134).

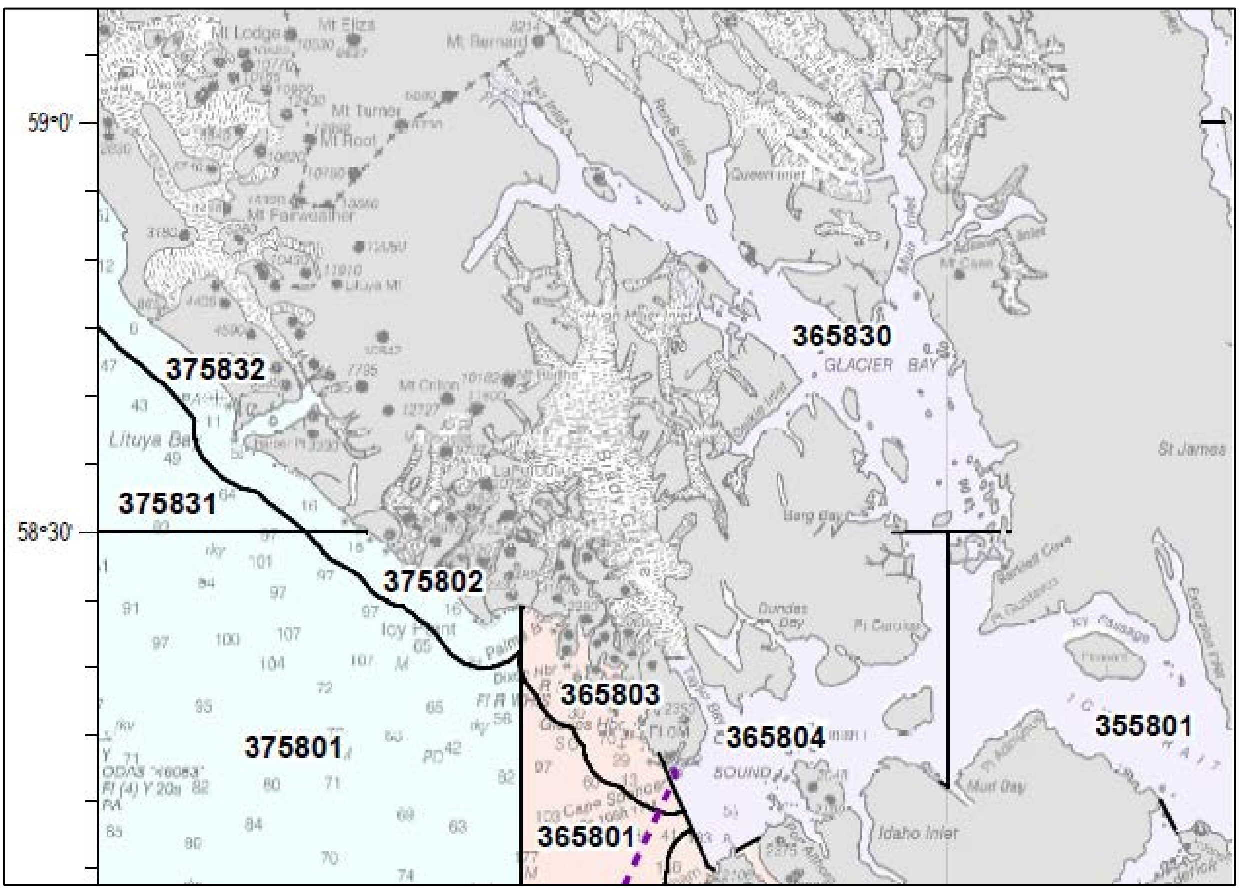

Figure 134. ADFG groundfish statistical areas in Southeast Alaska (ADFG 2011).

The spatial differences between the statistical and management areas used by the IPHC and the ADFG are magnified when attempting to analyze halibut data within the GLBA park boundary, which corresponds with neither of the agencies regulatory areas. Because of this difficulty, much of the data discussed in this document pertains to Pacific halibut as a fishery or single management unit.

When possible, information is presented at the IPHC statistical area (190,185,184,182,181) or the ADFG groundfish statistical areas (365830, 355801, 365804, 365803, 365802, 375832) to present the most spatially relevant data to GLBA.

4.18.2. Measures

- Spawning stock biomass
- Removals
- Size-at-age

Spawning Stock Biomass
Spawning stock biomass (SSB) is defined by NOAA (2006) as the total weight of all fish in the population that contribute to reproduction. SSB generally includes both males and females of the species and is often used to estimate egg production (NOAA 2006). Additionally, the IPHC defines female spawning biomass (Sbio) as an estimate in weight of sexually mature female halibut. Female sexual maturity begins at age 8, reaches 50% between ages 11-12, and 100% at age 20.

Removals
The term "removals" is defined by NOAA (2006) as the total number or weight of fish removed from a population by fishing. This includes the actual catch or harvest of fish as well as all fish mortality incurred during fishing activities. Removals in the halibut fishery can be categorized into five distinct areas: commercial fishery landings, fishery waste (mortality of fish captured but not landed including undersized fish), sport (guided and unguided), subsistence (federal and state authorized personal use), and bycatch (halibut caught by fisheries targeting other species) (Gilroy 2014).

Size-at-Age
Size-at-age is a common measure of many fish species and is representative of growth rates. NOAA (2006) describes this measure simply as the length or weight of a species at a particular age. The relationship between size-at-age, stock levels, and SSB is poorly understood (Stewart and Martell 2014).

4.18.3. Reference Conditions/Values

Spawning Stock Biomass
A defined reference condition for SSB does not exist for this component. The IPHC produces estimates of the SSB annually using setline surveys, but these estimates have varied greatly over the past 100 years. Generally, a commonly used reference for SSB can be estimated from the start of commercial harvest under the assumption that the stock is at carrying capacity. When the stock is 20-30% of unfished levels a fishery can occur, but when it is ≤20% no fishery can occur.

Removals
The defined reference condition for removals coincides with the Total Constant Exploitation Yield (TCEY) for a particular calendar year. The IPHC first calculates the Exploitable Biomass (ebio), which represents the abundance of halibut in that year, then modifies this by applying a harvest rate

(exploitation rate) from that ebio, which results in the TCEY for a given year. Because of this annual process, the reference condition for removals will vary from year to year.

Size-at-Age
The defined reference condition for size-at-age is undetermined. Due to an unknown number of factors, the size-at-age for Pacific halibut has increased and decreased over time (Figure 135) and a “norm” has not been established. This relationship only gets more complex when various spatial areas and age classes are analyzed.

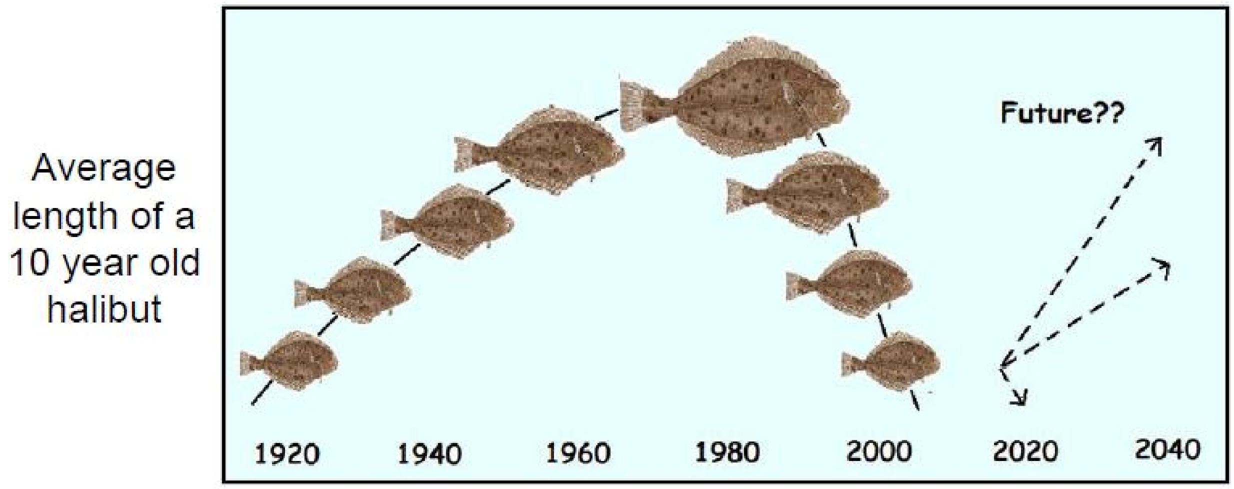

Figure 135. Size-at-age of a 10-year-old Pacific halibut over time (Sullivan et al. 2015).

It may not be the best practice to use a specific size or year as a reference condition, as most organisms experience natural variation in response to their environment or other naturally occurring variables. To what degree size-at-age fluctuates in response to these influences may not ever be fully understood. What may be more important is to observe to what degree a biological response is caused by anthropogenic influences and how much that response/change reduces a population’s resilience (Murdoch, written communication, 5 November 2015).

4.18.4. Data and Methods
The IPHC produces many research publications annually. Every year at the IPHC annual meeting, the organization releases a new summary publication of research and management activities known as the “bluebook” (e.g., IPHC 2014, IPHC 2015). This is a publication containing multiple research articles by multiple authors, as well as the annual stock assessment. Much of the information in this NRCA component is from these annual bluebooks.

In addition to the IPHC bluebook, the IPHC has released numerous technical reports. Publication of these reports began in 1969 and continues through the present. While each technical report is specific to a topic or small group of topics, together they cover a broad range of topics ranging from basic halibut biology, geographically specific research, and discussions of new research methodology. A number of IPHC technical reports (e.g., IPHC 2006, IPHC 2014a) have been cited in this document.

The IPHC also annually produces a *Report of Assessment and Research Activities* for the given year. This large report also contains many research articles and reports from different authors and coauthors. It contains research and data on the halibut fishery as a whole for the particular year. This report duplicates some articles from the other two IPHC publications and has also been cited within this document.

Bishop et al. (1993) conducted research specific to halibut in GLBA. It covers Pacific halibut and other groundfish habitat preferences in relation to oceanographic features. Bishop et al. (1993) used longline sets of constant length and hook number in a randomly stratified fashion. Fish size and species were recorded along with a grab sample of sediment from the longline set in an effort to establish a species to substrate to depth correlation. These data were collected from June to November of 1992 from the mouth of GBP. This study was useful in the identification of typical halibut habitat and distribution. Chilton et al. (1993) conducted diet studies of sport-caught halibut in GLBA to identify the prey preferences of Pacific halibut.

Clark et al. (1999) examined the long-term trends in halibut stock, size-at-age, and recruitment in relation to the PDO phenomenon and found strong correlation. The effect of the PDO was analyzed at a fishery-wide (North Pacific halibut) scale and included many sources (e.g., longline surveys, research charters) of halibut size and harvest information in relation to regime shifts in the PDO. Clark et al. (1999) identified the PDO as a contributor to the dramatic changes in growth and recruitment of the Pacific halibut.

Kaimmer (1997, 2001) conducted research on a condition referred to as chalky halibut. Chalky halibut is a condition that results in a “chalky” texture in the muscle tissue of recently caught halibut. Research appears to suggest that when a fish dies from exhaustion (or was exhausted at the time of capture) there is an accumulation of lactic acid in the muscle tissue of the fish which may contribute to the chalky texture (Kaimmer 1997). Kaimmer (1997, 2001) sought to correlate the condition with physical or environmental factors, as well as a method to predict which fish would develop the condition. Kaimmer (1997) conducted a series of surveys spanning 1996 and 1997 in an attempt to more fully document and understand the condition. These surveys solicited information from fishermen, fish buyers, fish brokers, fish processors, and retail users. Later, Kaimmer (2001) explored the effectiveness of using pH meters to predict the occurrence of chalky halibut. For this study, 32 halibut were collected near Homer, AK. These fish were then monitored frequently over a 6-day period for pH in several locations on the fish. Kaimmer (2001) attempted to establish a correlation between pH and the chalky condition.

Stewart (2015) provided an overview of the data sources for the IPHC annual stock review and discussions of all known sources of halibut removal, as well as examinations of biases inherent within certain data sources and their implications. This report reviewed data as it pertained to the North Pacific halibut fishery as a whole and is updated annually.

ADFG recreational halibut removal data were provided in spreadsheets by GLBA Fisheries Biologist Chad Soiseth. These spreadsheets contained summary data of ADFG logbook and mail-in-survey data that were compiled from various ADFG management reports and databases. These data were the most specific quantifications of data to GLBA and were utilized frequently in this document. These data used the ADFG groundfish statistical areas and were often summarized as “inside waters” (areas: 365830, 355801, 365804) and “outside waters” (the Gulf of Alaska coastline – areas-365803, 375802, 375832) (Figure 134).

Some IPHC commercial halibut removal data were provided in spreadsheets by C. Soiseth. These spreadsheets contained summary data of IPHC management area (primarily area 184) data that were compiled from various IPHC sources. These data were among the most specific to GLBA and were utilized when possible in this document. These data used the IPHC management areas and were often summarized as “inside waters” (GBP – area-184).

4.18.5. Current Condition and Trend

Spawning Stock Biomass

Accurately determining SSB is critical in the management and survival of the halibut species and is a complex task that is undertaken each year by the IPHC. Figure 136 represents the SSB estimates for the North Pacific halibut fishery as a whole. SSB is a necessary piece in the puzzle for determining the recommended season catch limits for halibut (IPHC 2014a). The IPHC produces an annual stock assessment from the available commercial fishery data, a setline survey, and other sources of halibut harvest. These data are then inputs for models that combine them with the current understanding of halibut maturity, natural mortality and growth (IPHC 2014a). An ensemble of four models is then used to estimate the current stock and the current spawning stock biomass (Stewart and Martell 2015). This is done each year in an effort to estimate both the trend direction and actual abundance of the resource.

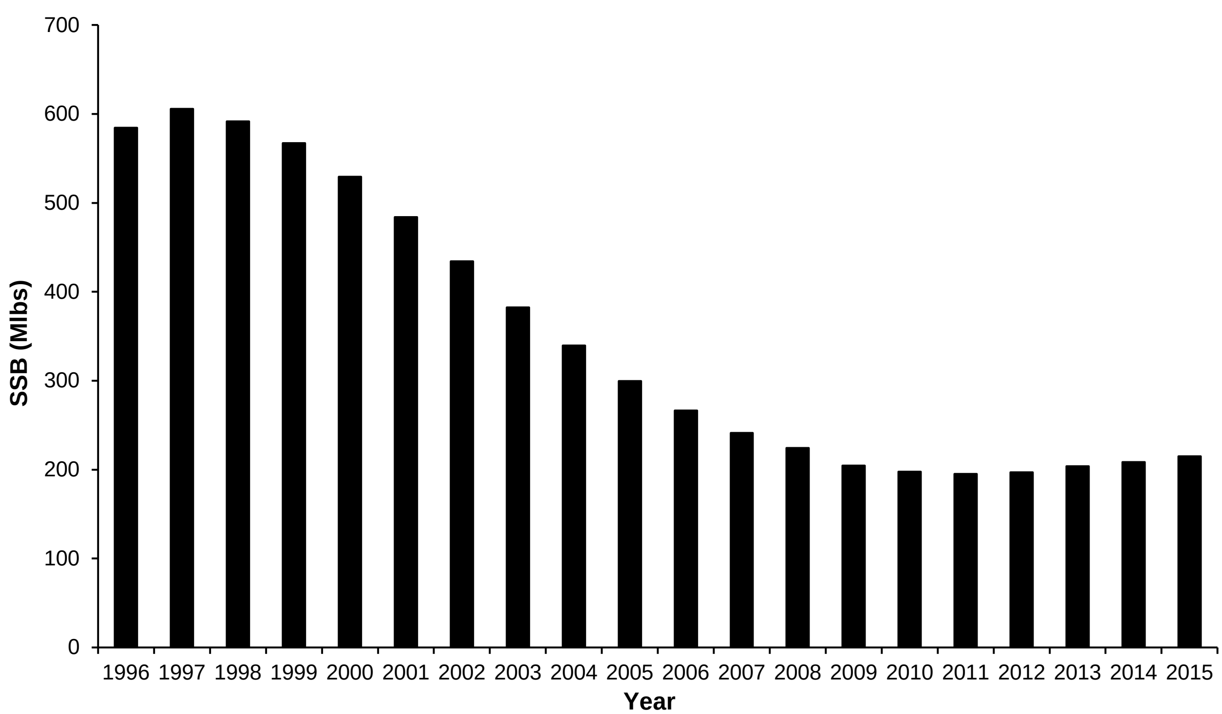

Figure 136. The 2015 Pacific halibut SSB estimate (Stewart and Martell 2015).

The SSB is also critical in the annual IPHC harvest determination (Figure 137). The IPHC sets a threshold based on SSB to regulate final harvest rates. This rule uses SSB ratio (compared to the estimated unfished spawning biomass) to begin to limit the proposed harvest when the ratio falls below 30% (of unfished biomass). The proposed harvest decreases linearly to 0% (no harvest) if the SSB ratio reaches 20% (of unfished biomass) (IPHC 2012).

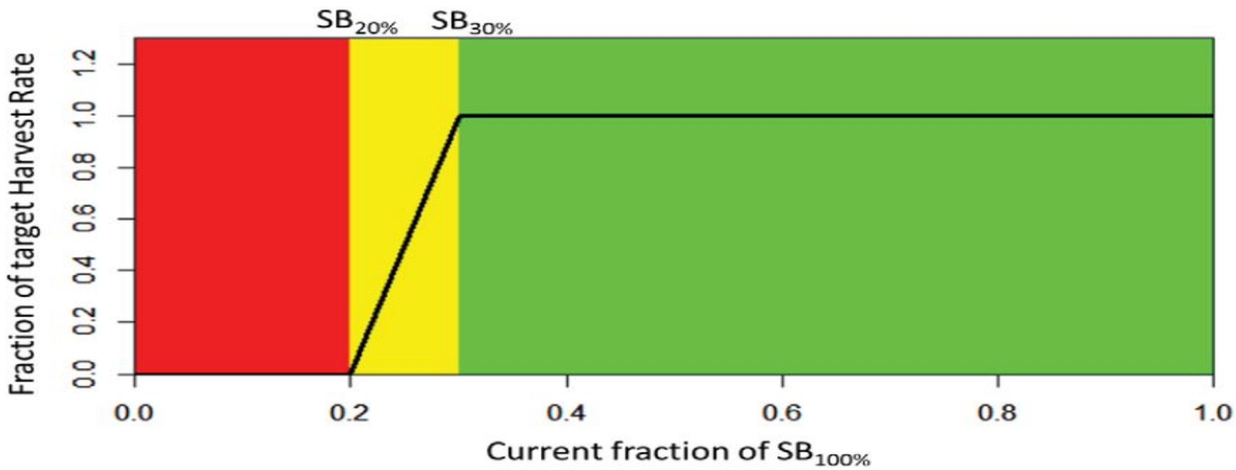

Figure 137. IPHC harvest control rule relative to target harvest rates (IPHC 2015a).

The halibut population for the North Pacific is modeled as a continuous population from the waters off the west coasts of California, Oregon, Washington, and British Columbia all the way to the Gulf of Alaska and the Bering Sea (IPHC 2014a). The population is then distributed among 10 regulatory areas for management implementation and enforcement. GLBA is split between IPHC areas 2C and 3A (Figure 138).

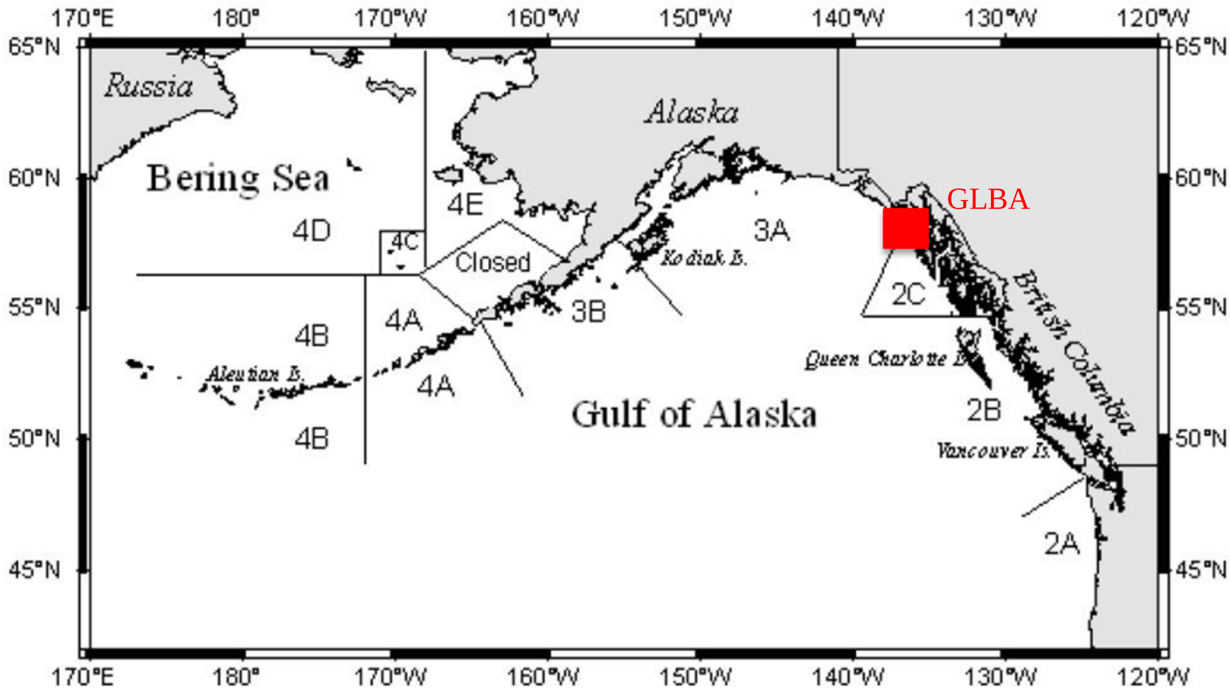

Figure 138. IPHC regulatory areas (IPHC 2014a).

In recent years, the trend in halibut population abundance has been relatively variable; however, an overall decline has been documented over the last decade (IPHC 2014a). During the 1970s, the stock estimates of abundance (and catch) declined to very low levels. This decline was believed to be the result of nearly three decades of poor halibut recruitment (IPHC 2014a). During the late 1990s and early 2000s this trend was reversed and high levels of abundance (and catch) were experienced. Since the height of this last peak stock abundance has once again been in decline (Figure 139) and allowed removals have decreased accordingly (Stewart and Martell 2014). Poor recruitment, limited understanding of halibut biology and life cycle, as well as catch/size limits are all thought to contribute to these declines (IPHC 2014a).

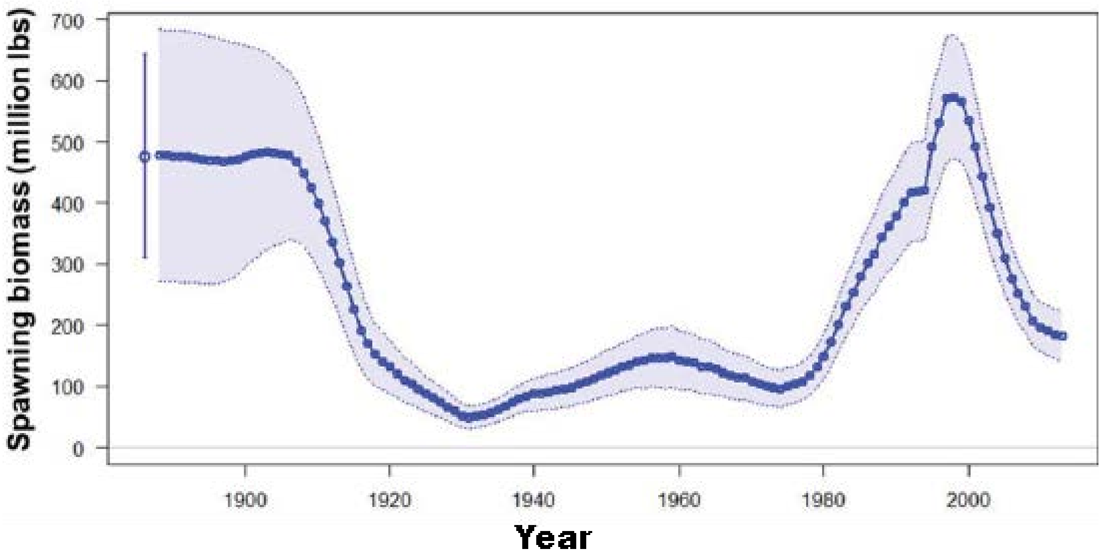

Figure 139. Spawning biomass estimates for Pacific halibut from the long time-series model (IPHC 2014c).

Sbio is currently considered stable and estimated at roughly 200 millions of pounds, net weight (Mlbs) fishery-wide with a plausible range of 150–250 Mlbs. (Stewart and Martell 2014). The current level of spawning biomass is estimated to be 42% of the equilibrium condition in the absence of fishing, with a 10 out of 100 chance that the stock is below the 30% relative spawning biomass harvest policy threshold (Figure 140) (Stewart and Martell 2014).

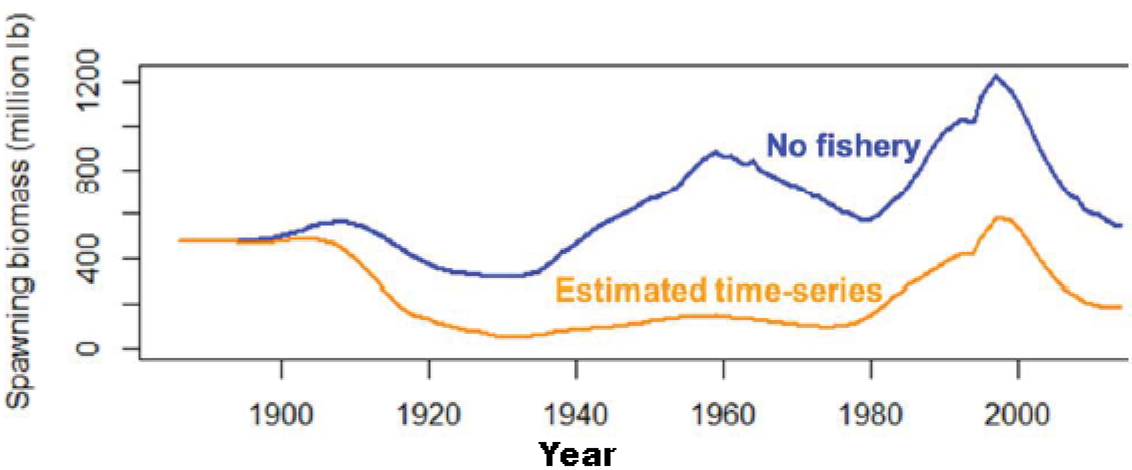

Figure 140. Estimated spawning Pacific halibut biomass time-series from the long time-series model (lower, orange line) and recreated time-series in the absence of fishery removals (upper, blue line) (IPHC 2014c).

Modelling the halibut fisheries stock and the SSB is complex and the individual models produce varying results (Figure 141). One of the models the IPHC utilizes is the Areas-As-Fleets (AAF) model. This model estimates current SSB at 133% of the minimum SSB values estimated in the 1970s, whereas the coast-wide model estimates that SSB is at 211% of the minimum values estimated for the 1970s (IPHC 2015a). Until modeling of halibut recruitment, estimates of size-at-age, and other biological indicators improve, the accuracy of stock forecasting and modelling stock biomass will remain less than desirable (IPHC 2015a). In 2015, the IPHC staff proposed studies to determine the best method of estimating the SSB (IPHC 2015a).

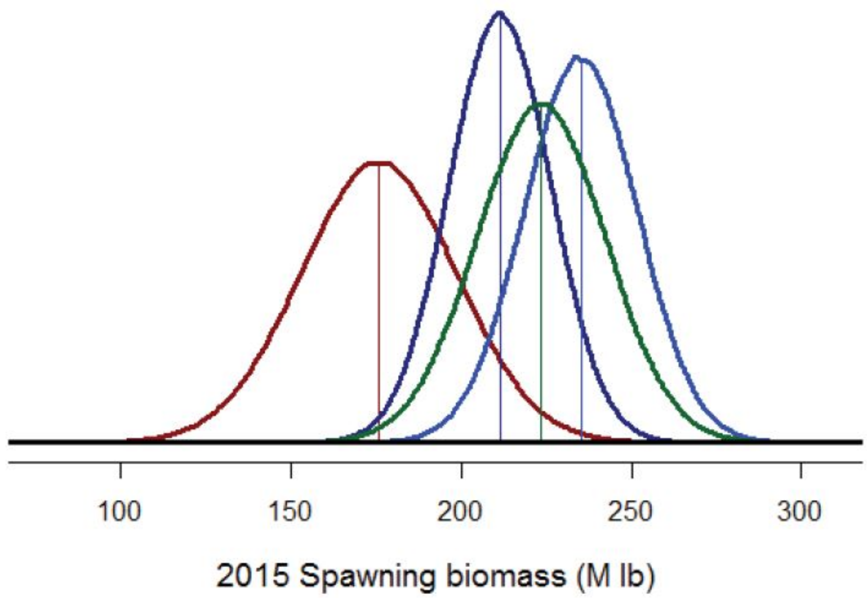

Figure 141. Comparison of the differing outputs of the four models used in the 2015 stock assessment estimating Pacific halibut SSB. Vertical lines indicate median SSB values (Stewart and Martell 2015).

SSB data specific to GLBA do not exist and cannot be estimated since the Pacific halibut stock is modelled as a single population. Pacific halibut migrate and freely move among the many IPHC regulatory areas, which necessitate this approach.

<u>Removals</u>

The total poundage of North Pacific halibut removed from the fishery in 2014 is estimated at 42.5 Mlbs (IPHC 2014a) down from the 2013 removal figure of 46.2 Mlbs. Over the past century halibut removals from the northeastern Pacific have varied from a low of 34 Mlbs (1977) to a high of 100 Mlbs (2004, Stewart 2015). The average halibut removal is estimated at 64 Mlbs (Stewart 2015). An estimate of total fishery-wide halibut removals (total halibut mortality) is a critical measure in halibut analysis and assessments. The total removal is currently estimated annually using five sub-components: commercial landings, commercial wastage, sport harvest, personal use and subsistence harvest, and bycatch (Stewart 2015) (Figure 142).

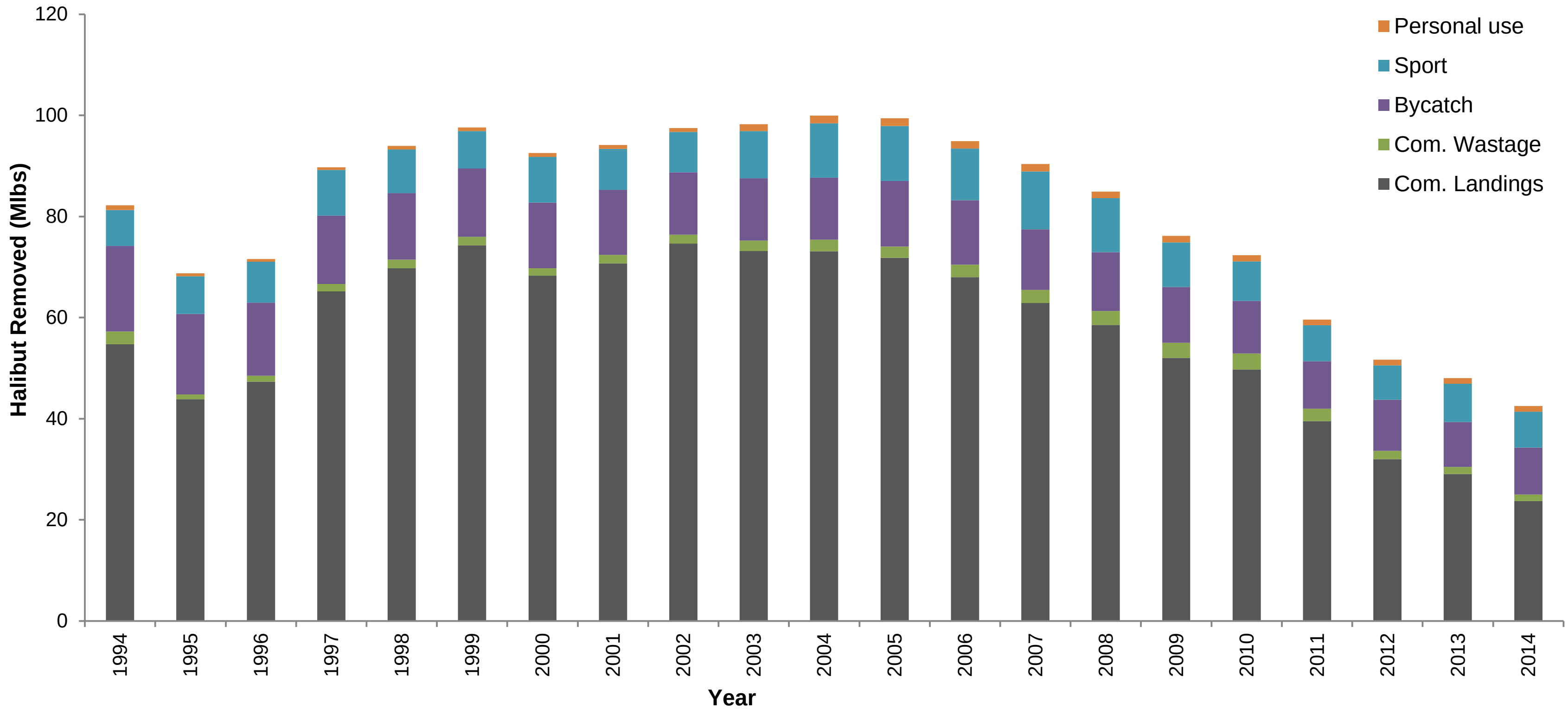

Figure 142. Historical Pacific halibut removals by type from 1994 to 2014 in all IPHC regulatory zones (2A, 2B, 2C, 3A, 3B, and 4) (IPHC 2014a).

Commercial and sport fishery halibut removals within GBP are limited. Especially because existing NPS regulations (36 CFR 13.1130-1146) limited the number of LAP holders that can continue to fish within the Bay Proper in 1999 and LAPs cannot be passed on (Soiseth, written communication, 13 July 2015). Thus, participation in the commercial halibut fishery will become extremely limited as anglers age and drop out of the fishery. Based on the original age distribution of qualifying participants, very few LAP holders will exist beyond 2050. Moreover, NPS restricts the number of vessel that can enter during the summer (June-August) visitor use season. Up to six charter and 25 private vessels are authorized to enter the Bay Proper on a daily basis during this period. Thus guided and unguided sport halibut fishing effort is limited during the summer period. When analyzing halibut data specific to GLBA, it is important to keep in mind the geographic area that is being represented by the data; with the exception of IPHC area 184 representing GBP, no halibut harvest or effort data align very well with the actual park boundaries but some approximate comparisons may be made.

IPHC area 184 coincides with the NPS boundary delineating GBP. Figure 143 represents the net weight of halibut harvest and effort from 1992-2013. During the past 10 years, effort and harvest have declined in this statistical area.

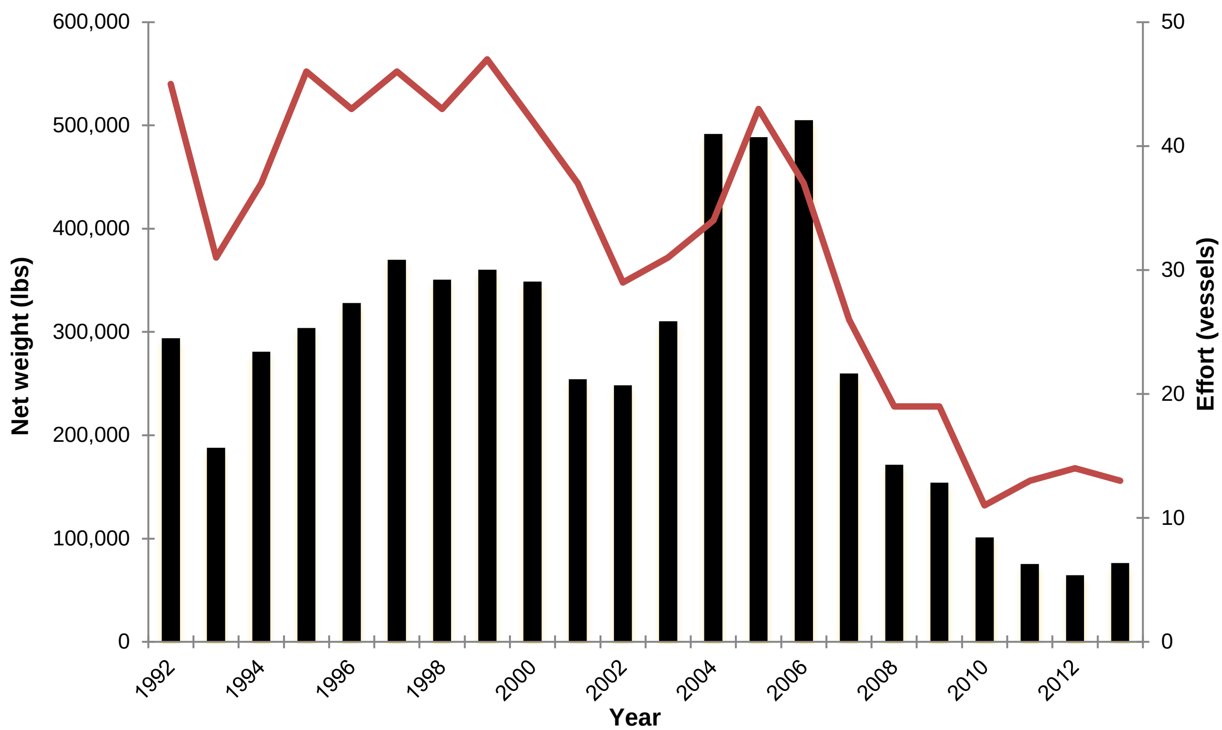

Figure 143. Net weight of halibut harvest (black bars) and number of unique commercial fishing vessels (red line) inside IPHC area 184 (Glacier Bay Proper) (IPHC 2015b). Note that LAP regulations were established in GBP in 1999/2000.

The ADFG groundfish statistical areas 355801, 365830, and 365804 may be loosely associated with GBP (Figure 134). Two of these statistical areas (365804 and 355801) encompass additional areas to the south and east of GLBA within Cross Sound and Icy Strait but are used for a rough estimation of trend for the purpose of halibut harvest and effort statistics (Figure 144). Likewise, the ADFG groundfish statistical areas 365803, 375802, 375832 can be associated with GLBA's outer Gulf of Alaska boundary. With the exception of statistical area 375832 which fails to align closely along its northerly limit with the park's outer waters, these three areas conform relatively closely to the Gulf of Alaska waters within GLBA 4.8 km (3 mi offshore). Thus, they can be used as a good spatial approximation of GLBA's outer coast waters along the eastern Gulf of Alaska (Figure 145).

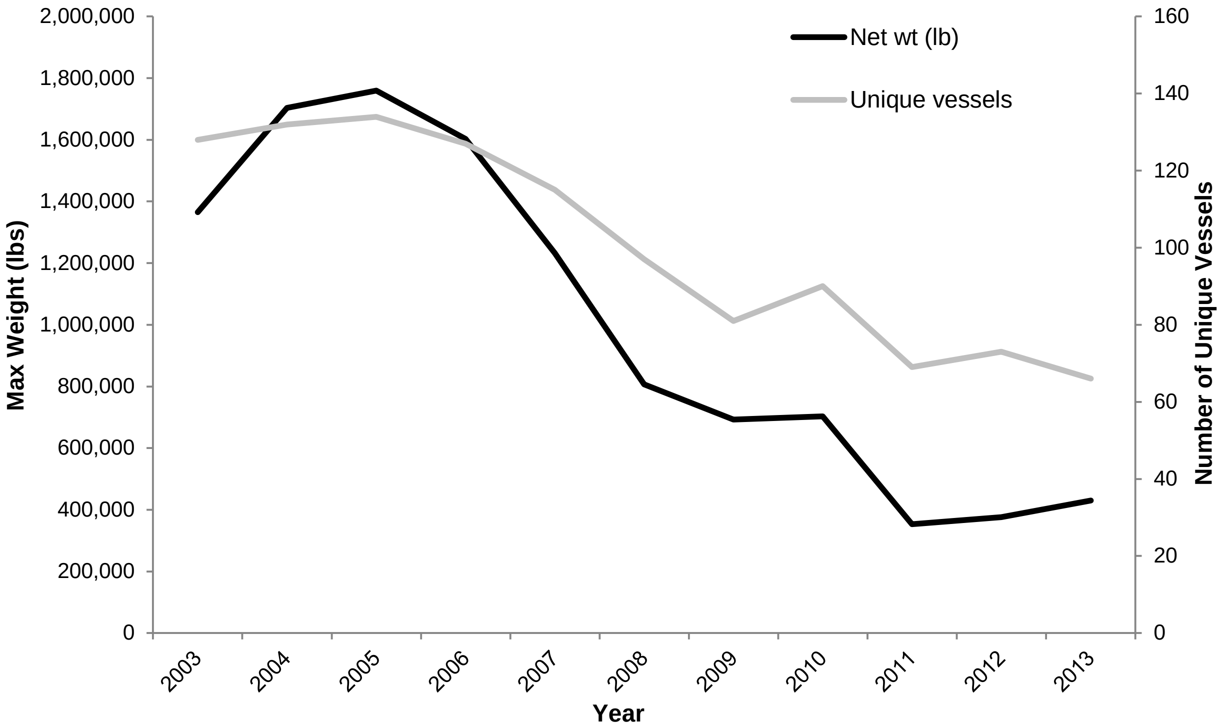

Figure 144. Approximation of commercial halibut harvest and effort (unique commercial fishing vessels) within national park waters of GBP, Cross Sound, and Icy Strait, as well as state waters outside the park (ADFG groundfish statistical areas 355801, 365830, 365804) along with the unique number of fishing vessels over an 11-year period (IPHC 2015b). These statistical areas encompass considerably more area than defined by the park's political boundary and therefore overestimate commercial halibut removal from park waters in this area.

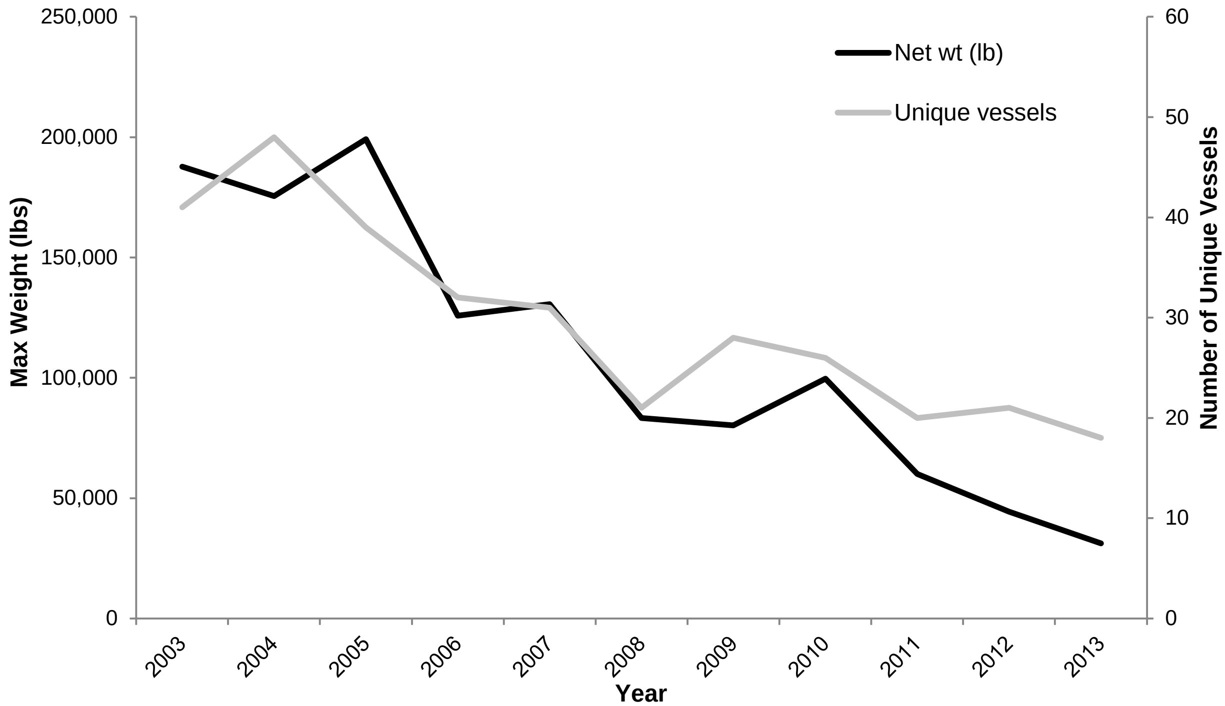

Figure 145. Approximation of commercial halibut harvest and effort within GLBA "outside waters" (ADFG groundfish statistical areas 365803, 375802, 375832) along with the unique number of fishing vessels over an 11-year period (IPHC 2015b). These statistical areas correspond fairly well with the park's outer coast boundary and offer a reasonable approximation of area harvest and effort.

A second category of removal of halibut is termed "fishery wastage." This includes halibut that are removed from the population through the commercial fishing industry, but do not become part of the landed catch (Stewart 2015). The main sources of wastage include: fish that were captured but lost when the fishing gear was lost, fish that were discarded for regulatory reasons such as over limit catches, and fish that are caught but discarded because they were below the legal size limit (Stewart 2014). This category was added in the 1980s and was not counted as a part of removals prior to that time. Fishery-wide wastage decreased from an estimated high of more than 4.5 Mlbs in the mid-1980s to a low of the current (2014) estimate at approximately 1.3 Mlbs (IPHC 2014a, Stewart 2015). Wastage was estimated to be highest in the 1980s, decreasing in the early 1990s and then rising again with the trend of decreasing size-at-age of halibut (Stewart 2015).

A third category of removals, sport removals, was also not considered a part of overall removals prior to the late 1970s (Gilroy 2015). Considered insignificant prior to 1973, the sport removals were regulated by the commercial fishing season duration and were not considered as an appreciable impact to the halibut population (IPHC 2014). However, since that time this type of removal has shown rapid growth in Alaska; an estimated 289,000 lbs were removed in 1977, while more than 11 Mlbs were removed by sport fishing in 2007 fishery-wide (Stewart 2015), this represents an increase of more than 40 fold in just three decades. Since 2007, charter removals have been actively managed using bag and fish size retention limitations due to increasing harvest by this sector and a corresponding allocation concern between commercial and guided charter removals (IPHC 1998, Stewart 2015). About half of these removals come from Area 3A each year, which includes the Gulf of Alaska coastal waters of GLBA. But Area 2C generally comprises the next largest coast-wide component of sport removals. The Catch Sharing Plan (CSP), implemented by the NOAA in 2014, established allocations between the sport charter and commercial sectors in Areas 2C and 3A that fluctuates annually with stock abundance (78 FR 239).

Figure 146 illustrates annual sport removals from GBP (IPHC area 184) as net weight using angler-reported ADFG mail survey information on the number of fish harvested factored by mean net halibut weight as determined from ADFG creel surveys . Figure 147 represents these same data as fish numbers rather than by weight, and presents charter (guided) logbook data in addition to mail survey data. The combined commercial and sport harvests are shown in Figure 148 for the same time period and area of GBP (IPHC area 184). Note that not only has overall harvest declined since 2006 as a consequence of decreasing stock abundance, but that the proportion of total harvest attributed to the recreational sector has increased. Most significantly, recreational harvest exceeded commercial harvest in 2013 by over 21,000 lbs. The NOAA CSP will address and rectify the allocation issue in coming years.

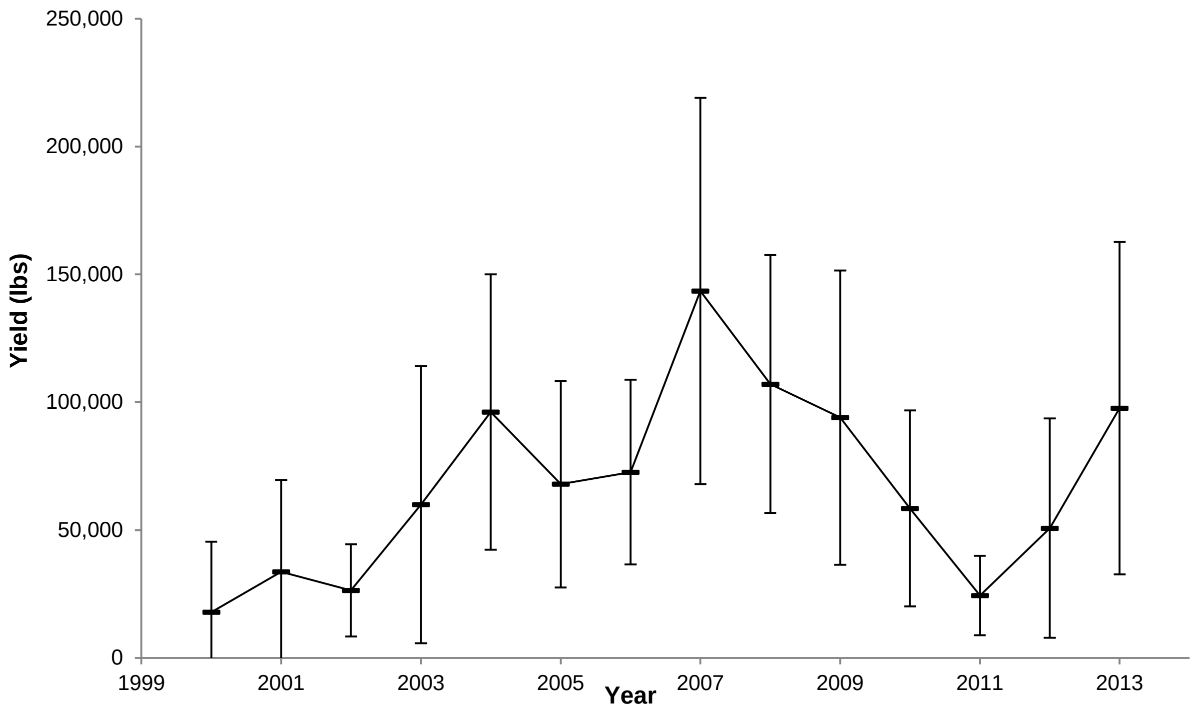

Figure 146. Sport halibut yield (net lbs) in GBP (ADFG 2015b).

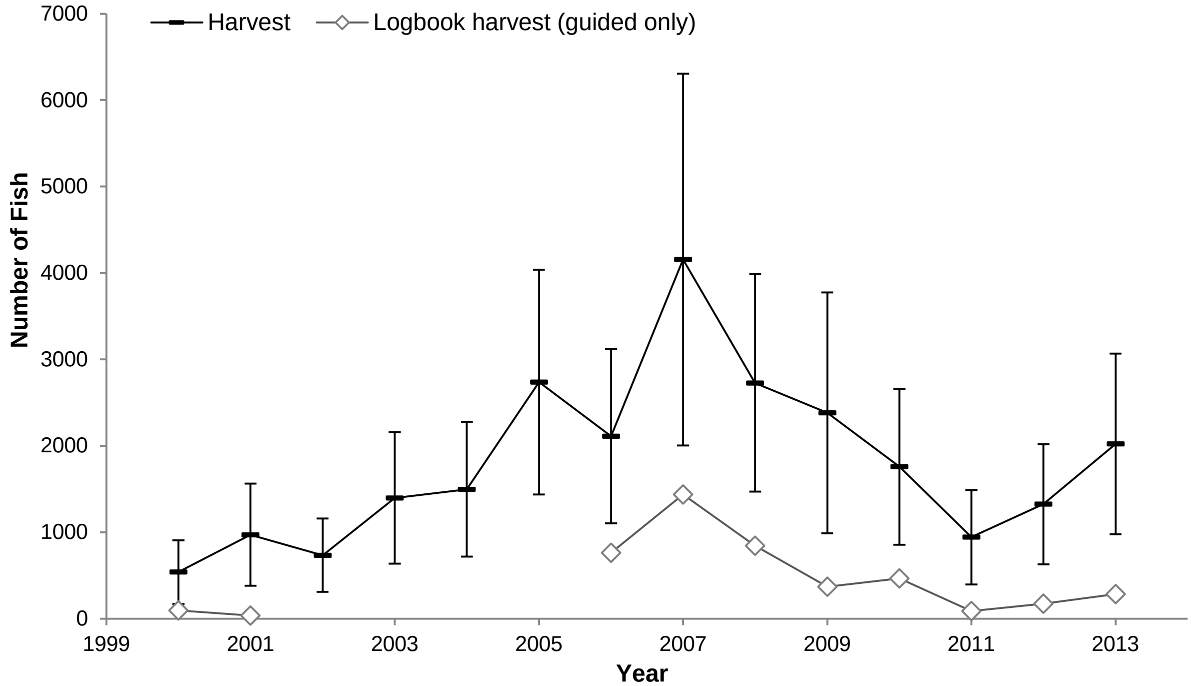

Figure 147. Sport halibut harvest (# of fish) in Glacier Bay (proper) (ADF&G 2015b). Note that harvest values (represented by the rectangular boxes) include combined harvest from both the guided and unguided sectors of the sport fishery.

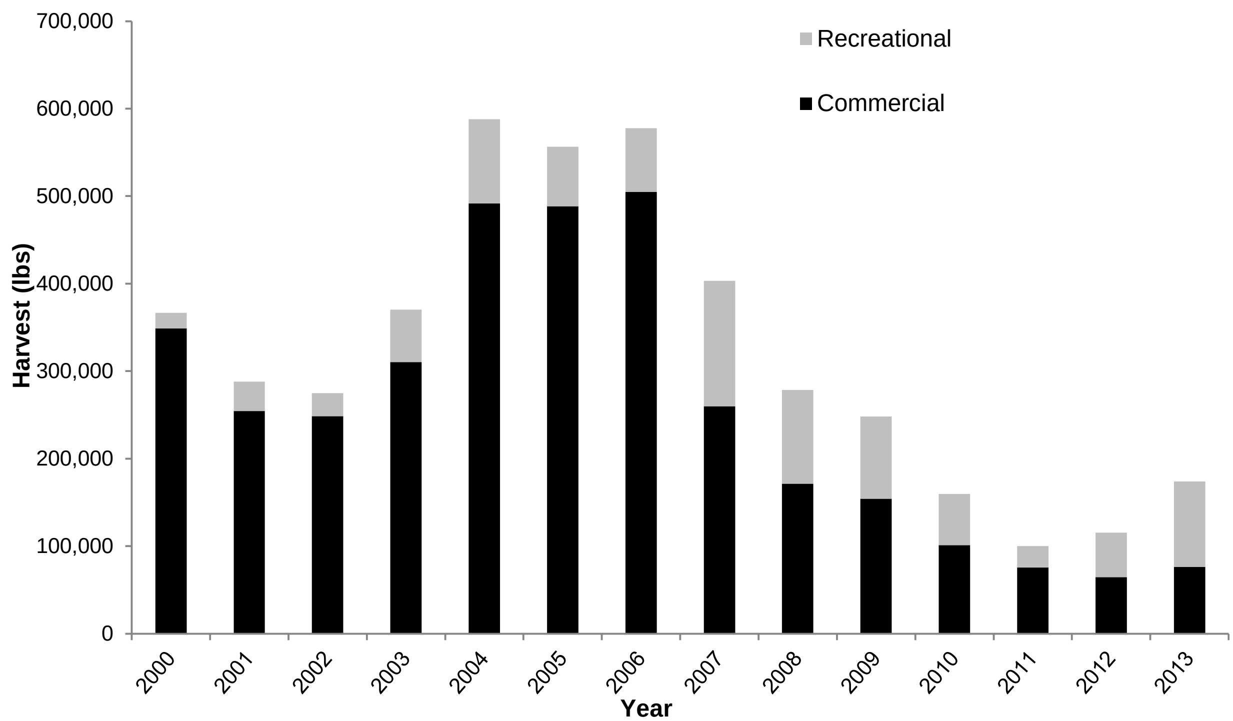

Figure 148. Total commercial and sport harvest (net lbs) of Pacific halibut from GBP (IPHC 2015b and Meyer 2015).

Subsistence harvest of halibut by rural Alaskan residents or members of Alaskan Native tribes is a fourth category of removal in the coast-wide fishery. Estimates of this harvest or removal are not carefully monitored and are often not available every year (Stewart 2015). While the data are not precise, the removal is estimated at less than 2 Mlbs annually fishery-wide, and is not considered a significant removal factor (Stewart 2015).

Federal subsistence harvest under ANILCA is not allowed in the park portion of GLBA, yet some illegal (federal) subsistence halibut harvest has reportedly occurred within park waters. However, these harvest amounts are likely relatively inconsequential to the halibut fishery (Soiseth, written communication, 6 May 2015).

Bycatch is the final category of fishery-wide removals and is defined as halibut that are caught or removed by the fishing industry without intent. This occurs when fishing for other species and incidentally removing halibut in addition to the desired species. Commercial salmon trollers occasionally catch halibut as do commercial groundfish longline fishers (Soiseth, written communication, 4 November 2015). The quality and regularity of the reporting of these numbers are highly variable (Stewart 2015). Currently (2014), an estimated 9.3 Mlbs of halibut are removed annually through bycatch fishery-wide (Stewart 2015). Bycatch has been decreasing steadily since 2006.

Neither wastage nor bycatch data specific to GLBA exist and these cannot be estimated with high confidence due to the geography of the statistical areas used by both the IPHC and ADFG.

<u>Size at Age</u>
The individual size-at-age of halibut has been decreasing for decades (Stewart 2015). Data indicate that the period from the mid-1970s to the mid-1990s experienced the greatest period of decrease in size-at-age (Stewart 2015). Currently, the halibut population size-at-age is still considered low, relative to the past several decades, but comparable to the early 20th century (Stewart and Martell 2015). Decreased size-at-age increases the number of fish below the minimum size threshold (81.3 cm [32 in]), which is the size at which fish are considered to be recruited into the commercial fishery (Stewart 2015) and has been an important contributor to recent overall stock declines (Stewart and Martell 2014). Historical size-at-age is displayed for all female halibut as well as by IPHC regulatory area in Figure 149 below.

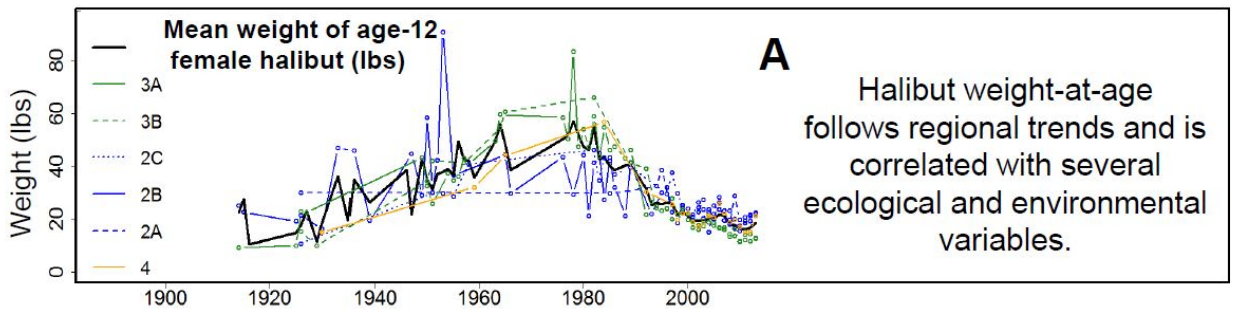

Figure 149. Historical weight at age of 12-year-old female halibut (Sullivan et al. 2015).

Several factors may be influencing the current decrease in size-at-age including the PDO, increased competition (intra- and interspecific), and change in prey quality and/or abundance (Clark et al. 1999, Sullivan et al. 2015). The potential effects from the PDO as well as competition are described in the threats and stressors section below. A change in prey quality or abundance may be an influence on the halibut's decreased weight-at-age, as a strong biological response seen post-1977 is consistent with a reduction in carrying capacity (Clark et al. 1999).

This trend of decreasing size is a priority for IPHC research as well as other organizations (Stewart et al. 2015). IPHC staff, the NMFS, the University of Washington (UW), and the ADFG are all cooperating in an effort to examine the multiple factors influencing this phenomenon (Stewart et al. 2015). Interestingly, the long time-series model the IPHC utilizes in modelling the halibut population predicts this decrease in size-at-age and subsequent decrease in overall stock even in the absence of fishery removals (Stewart and Martell 2014).

Threats and Stressor Factors

GLBA staff identified many threats and stressors that likely affect halibut in the park. The list of threats and stressors include halibut harvest, changes in climate or regime shifts (e.g., PDO), ocean acidification, increased competition, frequency of chalky and mushy halibut, contaminants, and parasites (*Ichthyophonus*).

The IPHC has monitored the change in the Pacific halibut stock since the 1930s (IPHC 1998). During the early years of monitoring, the halibut stock was thought to be primarily influenced by commercial fishery removals and abundance was measured by the amount of fish caught per unit of standard fishing gear (IPHC 1998). During the 1970s, when the halibut stock hit an all-time low, other sources potentially impacting the halibut stock were examined. Today, better understanding of the removal sources of halibut, as well as improved data and modelling, have allowed IPHC staff to better forecast the current and future halibut stocks (IPHC 2015a). While halibut population dynamics are not completely understood, better research and modelling have lessened the threat of overharvest of the halibut stock considerably.

The PDO is an oscillatory pattern of climate variability similar to El Niño (Mantua 2002). These variations are most apparent in the boreal winter and spring and can have significant impacts on sensitive natural resources (Mantua 2002). However, unlike El Niño events that persist for only 6 to 18 months, PDO events persist for 20 to 30 years. The effect produced by this mechanism can particularly impact the production of phytoplankton and zooplankton (Mantua 2002). Any change in the production of phytoplankton and zooplankton will have an impact on the higher trophic levels (including halibut) by altering food supply (Mantua et al. 1997).

From 1925-1946, and again from 1977 until the late 1990s, the PDO was considered to be in a positive "phase" (Clark and Hare 2002). During this positive phase, the IPHC's long time-series stock assessment model estimates halibut recruitment to be 37% higher on average (Stewart and Martell 2014). This estimate was confirmed by Clark and Hare (2002) and is illustrated in Figure 150. The opposite of this, decreased recruitment in negative phases, may also be seen.

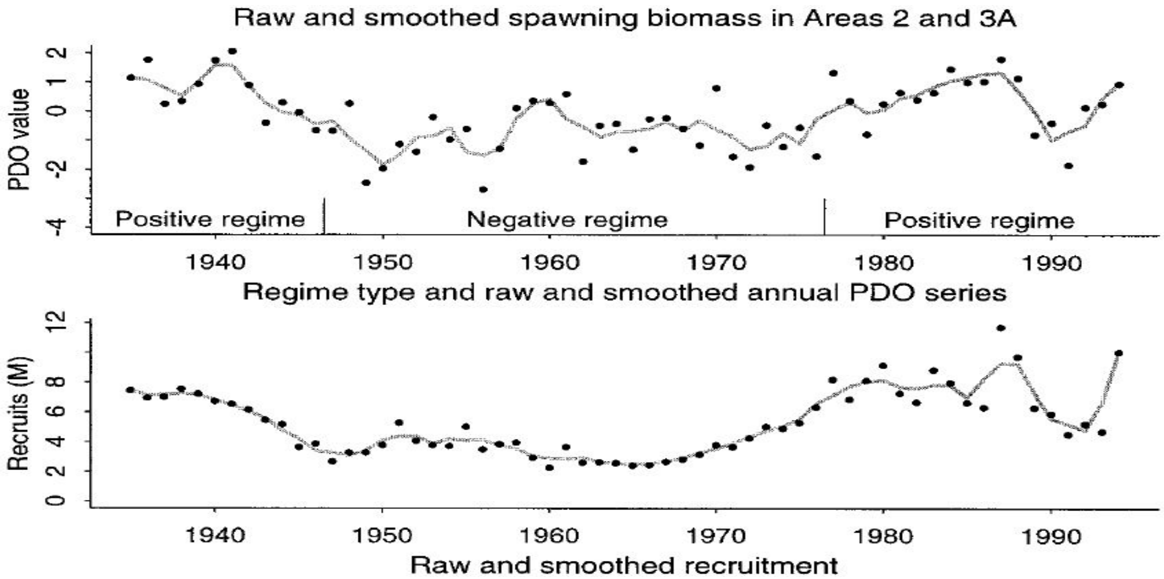

Figure 150. PDO regimes compared to halibut recruitment (Clark and Hare 2002).

Marine ecosystems have been profoundly affected by climate change, with impacts at all trophic levels from plankton to apex predators (Lauria et al. 2012). Climate change is one of the major forces affecting ecosystems across the globe; this threat is becoming better understood as research and data continue to become available. Changes in the normal temperature and precipitation within GLBA could have both direct and indirect effects on the halibut population. Examples of direct impacts could include shifts in the timing and magnitude of freshwater input into GBP due to alterations in the timing and magnitude of snow, sea ice, and glacial melt. Change in the timing and magnitude of freshwater input and subsequent changes in circulation patterns within the Bay (Etherington et al. 2007, Hill et al. 2009) would likely alter the timing, magnitude, and spatial patterns of primary production, thus altering all marine trophic levels in Glacier Bay (Robards 2014).

Ocean acidification is a direct risk to Alaska's food chain, as well as to Alaska's commercial fisheries industry (Mathis et al. 2014). Carbon dioxide absorbed by ocean waters reacts with calcium carbonate ions and consequently reduces those ions available in the ocean waters. Calcium carbonate ions are the same ions that some zooplankton, mollusks, shellfish, and corals use to build and maintain their shells and exoskeletons (Reisdorph and Mathis 2014). These animals play a critical role in the marine ecosystem as key prey species for the mid-trophic level community, providers of habitat for animals (corals), and as a form of natural erosion control for shorelines and beaches (OCB and EPOCA 2010).

Competition for territory and prey species may also be affecting halibut stock and size-at-age. Several species including yellowfin sole (*Limanda aspera*), arrowtooth flounder (*Atheresthes stomias*), rock sole, and flathead sole, may be potential competitors to the halibut (Williams 2015). Competitors such as arrowtooth flounder may be increasing competition for food, leading to nutritional deficiencies in halibut, and may potentially be a cause or contributor to mushy halibut syndrome, although little is currently known about that syndrome and its causes. The relationship between halibut and competing species is poorly understood and more research is needed. Concern for this competition may be warranted as arrowtooth flounder, although a smaller fish species, have close to ten times more female spawning biomass than halibut in the North Pacific (Spies and Turnock 2014).

Halibut that have been harvested occasionally display one of two undesirable conditions. In a small portion of the total halibut harvest, mushiness or chalkiness of the flesh can occur (IPHC 2014b). While fish with these conditions appears safe for human consumption, they do reduce the desirability and overall quality of the product. After harvest, a small number of halibut exhibit “flaccid” or “jelly-like” flesh over large areas of body tissue (IPHC 2014b), a condition known as mushy halibut syndrome (Meyers et al. 2008). Little is known about the occurrence of mushy or “jelly-like” flesh in halibut, but reports from recreational halibut fishers in parts of Southcentral Alaska are becoming more common (IPHC 2014b). The cause is believed to be due to nutritional deficiencies (IPHC 2014b) and shows evidence of concentrating in certain locales and during certain years (1998, 2005, 2011, 2012; ADFG 2015a). Because the apparent cause is nutritional/environmental rather than a biological vector of disease, concern over an outbreak is minimal.

Chalkiness in halibut also occurs infrequently but can affect as much as 3% of the commercial harvest, resulting in a multi-million dollar loss to the industry (IPHC 2014b). This condition has been known to exist within the halibut industry for more than 40 years but is still poorly understood (Kaimmer 1997). The chalkiness has been shown to be related to a lactic acid buildup in the muscle tissue of halibut, likely caused by the stress of being captured (Kaimmer 1997). Some evidence suggests that male halibut are more susceptible to becoming chalky than females (IPHC 2014b). The condition is reversible in living fish and may not occur if harvested fish are allowed a resting period prior to processing (IPHC 2014b). Similar to mushy halibut, because the potential causes include physiological stress and environmental conditions rather than a biological vector of disease, concern over an outbreak is also minimal.

The protozoan parasite *Ichthyophonus* is wide-spread among fish species and has been detected in more than 80 species worldwide (McVicar 1999, Mendoza et al. 2002). The effects of this parasite vary greatly by host species and even among individuals within species. The parasite’s greatest effect has been seen in Atlantic herring (*Clupea harengus*), where massive fish kills have been observed (Dykstra et al. 2013). Its effects on Pacific halibut seem to be less pronounced. First discovered in 2011 by testing halibut heart tissue collected from disparate locations, the parasite is now believed to be wide-spread among the halibut population (Dykstra et al. 2013). In an expanded 2012 study, the parasite was found in 33.7 percent of all halibut tested (Dykstra et al. 2013). It is unknown whether *Ichthyophonus* infection in halibut is a new phenomenon or a recently discovered long-term infection (Dykstra et al. 2013). Even with profound effects from the parasite documented in other fish species (e.g., Pacific herring, chinook salmon) (Kocan et al. McVicar 1999, Kocan et al. 2004), to date it is unknown what effect, if any, this parasite is having on the halibut population (Dykstra et al. 2013).

<u>Data Needs/Gaps</u>

Data concerning halibut harvest (removals), size-at-age, and spawning stock biomass, as they pertain to GLBA waters, are scarce. Enormous amounts of halibut data are available through the IPHC and

ADFG, as well as other agencies, but these data do not coincide well with GLBA boundaries. Adapting these data specifically to the GLBA park boundary is not possible and only general conclusions may be drawn from these data. Several studies and data sources are available that are specific to the halibut within GLBA but do not address the measures mentioned in this document.

A more complete understanding regarding halibut movement and use of Glacier Bay is needed as well. Recent work (Nielsen et al. 2014) has been done on halibut movement by doctoral student Julie Nielsen, associated with the School of Fisheries and Ocean Sciences at UAF. Past work characterized small-scale halibut movements using acoustic tags on adults and determined that non-dispersive movement patterns, relatively small home ranges and site fidelity exhibited by adult females suggest Glacier Bay could prove valuable for conserving halibut broodstock. Results of more recent ongoing work suggest that the majority of pop-up satellite archival tagged (PSAT) halibut for which movement data (i.e., evidenced by depth, temperature and tides) were recovered (14 of 20) showed no evidence of winter spawning migrations outside of GBP. The significance of this finding is that as commercial fishing is phased out in GBP (i.e., by 2050 or soon thereafter) adult fish are likely to be better protected because fishery mortality will be considerably reduced. Additional and continued research related to this topic is needed, especially moving closer towards the culmination of the commercial fishing in GBP. Research into what kind of ecological shifts may occur following the cessation of commercial fishing in GBP will likely be needed in the future.

Studies regarding site fidelity in halibut in GBP would also be beneficial, especially in regards to more fully understanding the species' diet in the bay and what their influence is on the benthic community of the bay. Understanding this complex relationship may help managers more accurately predict how the benthic community and halibut in general may shift with increasing threats of ocean acidification and climate change.

<u>Overall Condition</u>

Spawning Stock Biomass

The staff at GLBA determined the SSB measure had a *Significance Level* of 3. The data available concerning SSB are fishery-wide; no SSB has been estimated or is applicable specifically to GLBA or GBP. The fishery-wide spawning stock biomass (female) is currently considered stable and estimated at roughly 200 Mlbs with a plausible range of 150–250 Mlbs. (Stewart and Martell 2014). The current level of spawning biomass is estimated to be 42% of the equilibrium condition in the absence of fishing, with a 10 out of 100 chance that the stock is below the 30% relative spawning biomass harvest policy threshold (Stewart and Martell 2014).

While the overall SSB is considered low when compared to recent decades, the downward trend seems to have stopped. Therefore, this measure is assigned a *Condition Level* of 1, indicating low concern.

Removals

A *Significance Level* of 2 has been assigned by GLBA staff to this measure. It is important to recognize that most of the data available concerning removals are fishery-wide. Over the past century, halibut removals from the Northeast Pacific have varied from a low of 34 Mlbs (1977) to a high of 100 Mlbs (2004; Stewart 2015). The average halibut removal over this time is estimated at 64 Mlbs (Stewart and Martell 2014). Total removals were recorded at the highest levels (i.e. generally from 80-100 Mlbs.) from the late 1980s to late 2000s (i.e., 1986-2008), but have been in general decline since that time to their current level of 42.5 Mlbs in 2014 (Stewart et al. 2014). Fluctuations in overall removals have been the norm over the last century, and methodologies for estimating the stock levels and for setting the harvest levels have been steadily improving.

Commercial harvest for GBP was evaluated using data from IPHC statistical area 184. Over the past decade these data indicate a commercial harvest high of over 500,000 lbs (2006) and a commercial harvest low of 64,512 (2012). It should be noted, however, that the decline in commercial harvest numbers within GBP do not indicate a decline in local halibut abundance, but rather likely reflect the result of declining coast-wide SSB, decreased size-at-age, a reduced total allowable catch, and the establishment of the LAP fishery since 1999/2000. Declines in harvest more directly reflect the management decisions to reduce harvest that are put into place by the many agencies that oversee the commercial fishing regulations for halibut in Alaska and are not necessarily reflections of halibut abundance. Typically, management decisions to reduce harvest are based on population estimates and are adjusted based on estimated halibut abundance. Between 2007 and 2013 the commercial harvest quota was reduced by approximately 75% based on population estimates that showed the halibut population was in decline.

While the Pacific halibut is one of the best managed fish species in the world, realized harvest still regularly exceeds the recommended harvest rate (Stewart 2015). In the past decade, the realized harvest rate has exceeded the blue line – which represents the annual harvest target – and in some year’s harvest has reached 60-80% when it was intended to be only 20%. Further compounding issues is that the final annual harvest quota has not yet been fully established by backed science. For these reasons, and despite the fact that recent halibut removals are considered low relative to past fishery performance, this measure was assigned a *Condition Level* of 2, indicating moderate concern.

Size at Age

GLBA staff has assigned this measure a *Significance Level* of 3. The data available concerning size-at-age are fishery-wide; no size-at-age data is available specifically for GLBA or GBP. The individual size-at-age of halibut has been decreasing for decades (Stewart 2015). Data indicate the period from the mid-1970s to the mid-1990s showed the greatest period of decrease in size-at-age (Stewart 2015). Currently, the halibut population size-at-age (fishery-wide) is still considered low, relative to the past several decades, but comparable to the early 20th century (Stewart and Martell 2015). Several factors may be influencing the current decrease in size-at-age including PDO or other environmental effects, increased competition (intra- and interspecific), and change in prey quality and/or abundance (Clark et al. 1999). A change in prey quality or abundance may influence halibut size-at-age, as the strong biological response seen post-1977 is consistent with a reduction in carrying capacity (Dorn et al. 2014).

Because of the multiple factors and potentially complex interactions between them, little certainty exists regarding the actual causes of the current small size-at-age exhibited by halibut. Moderate concern is therefore warranted, and this measure has been assigned a *Condition Level* of 2.

Weighted Condition Score

The *Weighted Condition Score* for Pacific halibut in GLBA is 0.54, indicating moderate concern. This level of concern is due to the overall decreases in SSB and size-at-age. The decrease in removals is from management action taken in response to the two other measures. The overall trend for the resource currently appears to be stable.

Pacific Halibut			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.54**
Spawning Stock Biomass	3	1	
Removals	2	2	
Size-at-Age	3	2	

4.18.6. Sources of Expertise

- Chad Soiseth, GLBA Fisheries Biologist
- Craig Murdoch, GLBA Fisheries Biologist

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4.19. Plants

4.19.1. Description

GLBA contains a complex and dynamic geological landscape that supports a diversity of vegetation characteristic of the northern Pacific coastal biome (Carlson et al. 2004). Land surface ages range from thousands of years old in areas that escaped the most recent glaciation to zero years where glaciers have recently receded. This results in a wide range of plant successional communities, ranging from sparsely vegetated barrens to mature spruce-hemlock forests and peat bogs (Carlson et al. 2004). GLBA is widely recognized as one of the best documented examples of terrestrial primary succession after deglaciation (Chapin et al. 1993). Four primary stages of vegetation succession are generally recognized within the park (Chapin et al. 1993). The first is a “pioneer community” of blue-green algae that form a crust on the newly exposed surface. The second stage consists of a mat of yellow avens (*Dryas drummondii*) with scattered alders, willows, and cottonwood. This is followed by a stage with dense thickets of alder, and finally a mature spruce forest dominated by Sitka spruce (Photo 46) (Cooper 1923, Chapin et al. 1993). Current spruce forests are projected to trend increasingly toward hemlock-dominated hemlock-spruce stands in the coming decades, ultimately to include areas of patchy bogs on poorly-drained sites in a few centuries (Sharman, written communication, 29 April 2015). A landcover map completed by Boggs et al. (2008) shows the general locations of various plant communities throughout the park (Figure 151).

Photo 46. Conifers in Bartlett Cove (NPS photo).

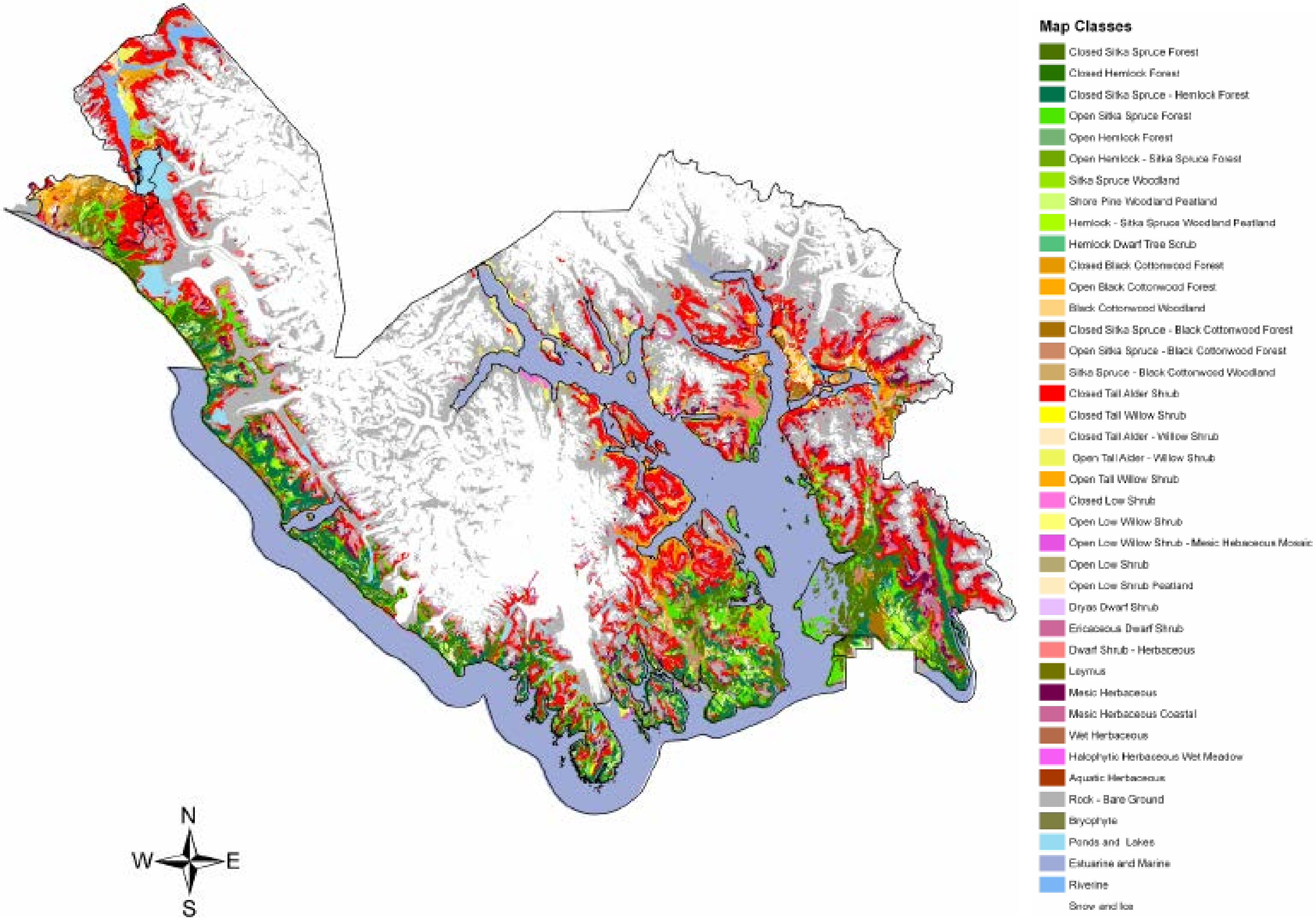

Figure 151. Landcover map of GLBA (Boggs et al. 2008).

4.19.2. Measures

- Species richness
- Number of non-native invasive species
- Distribution of non-native invasive species
- Sitka spruce mortality distribution
- Yellow-cedar (*Callitropsis nootkatensis*) distribution
- Yellow-cedar stand density
- Yellow-cedar age structure
- Tree density in shore pine (*Pinus contorta* var. *contorta*) stand

4.19.3. Reference Conditions/Values

The reference condition for the species richness measure will be the NPS Certified Species Lists for vascular and non-vascular plants in GLBA (NPS 2015). For the non-native invasive species measures, the reference condition is no invasive species present in the park. Sitka spruce mortality is a natural part of succession in Alaska's forested ecosystems and, as such, it is difficult to set a reference condition. The highest stand mortality experienced in GLBA during a 1980s spruce beetle (a type of bark beetle) outbreak, which was natural and not human-induced (Eglitis 1982), was approximately 70% in some areas. Any future mortality above this level or any mortality that is induced by human activity would be a significant cause for concern.

The reference condition for yellow-cedar measures will be determined by research recently completed in the park (NPS 2013b, Oakes et al. 2014, 2015). A reference condition could not be determined for tree density in shore pine stands due to a lack of information and data on the species.

4.19.4. Data and Methods

A list of vascular and non-vascular plant species found in GLBA was obtained from the NPSpecies database (NPS 2015; https://irma.nps.gov/App/Species/Search). Information regarding non-native invasive plant species in the park was found in Alaska Exotic Plant Management Team (EPMT) reports (Fisk 2011) and in ThemeManager, the NPS Alaska Region's repository for GIS data (NPS 2013a). ThemeManger includes data on the taxa found during each annual survey (2004-2012 for GLBA) and their distributions.

Sitka spruce mortality was studied extensively in the park during a spruce beetle outbreak in the 1980s. Eglitis (1982, 1986, 1988) reported on annual monitoring of spruce stands by the U.S. Forest Service (USFS) from 1982-1987. Schultz and Hennon (2007) continued this research with surveys in 1998 and 2004, documenting both spruce mortality and how the forests are subsequently responding. More recent information on spruce mortality was found in Alaska Forest Health Condition Reports (USFS 2012, 2013) and a map of spruce beetle damage in the GLBA area from 2011-2014 was provided by the USFS (USFS 2015b). This map was created from aerial survey data; these surveys cover large areas in a limited amount of time and may miss some small disturbance areas.

The majority of yellow-cedar research has occurred south of GLBA (Beier et al. 2008, Hennon et al. 2012). Wiles et al. (2012) included one site in GLBA (on Excursion Ridge) in a study on the impacts of climate on yellow-cedar growth rates. In 2011 and 2012, aerial surveys were flown over the park to map the distribution of yellow-cedar within GLBA, and researchers conducted ground surveys in some of the yellow-cedar stands identified to examine the health status of yellow-cedar trees inside the park (NPS 2013b). Plots sampled within GLBA were along the coastline of Graves Harbor and Dick's Arm (Figure 152; Oakes et al. 2014).

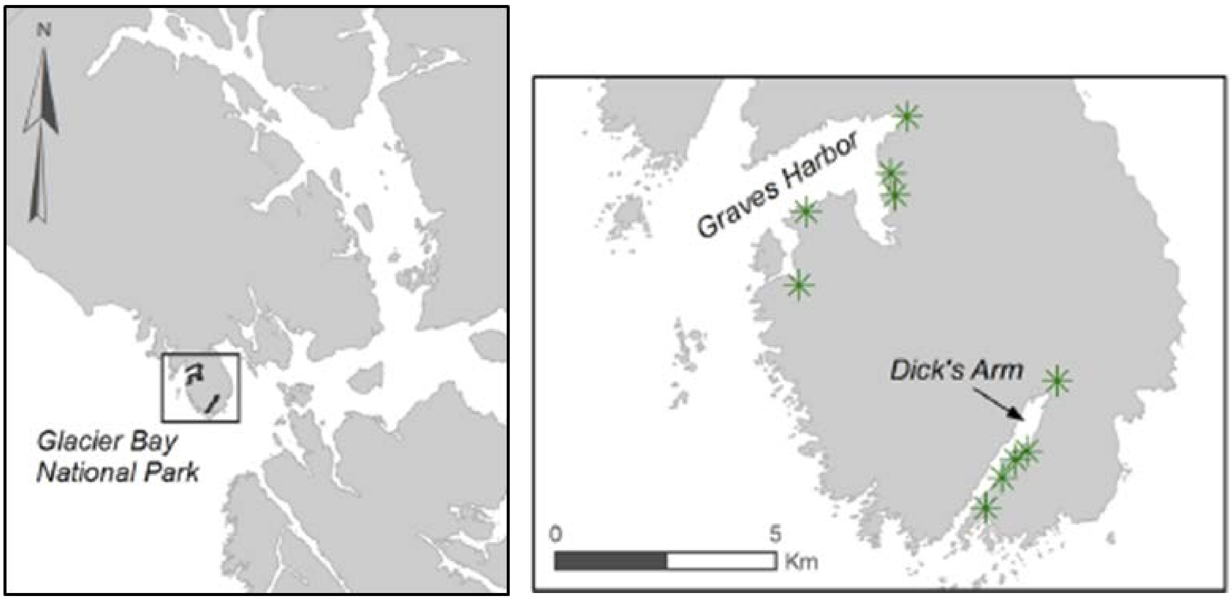

Figure 152. Location of yellow-cedar research plots within GLBA (Oakes et al. 2014).

Shore pine has recently been a subject of study in the GLBA area due to observed mortality and poor tree crown condition around Gustavus that began in 2010 (Mulvey and Cleaver 2014). The primary cause has been identified as a localized foliage disease epidemic of Dothistroma needle blight, caused by the native foliar fungal pathogen *Dothistroma septosporum* (NPS 2013c; Robin Mulvey, USFS Forest Pathologist, written communication, 7 May 2015). The USFS installed four transects around Gustavus to monitor shore pine health and survival (Mulvey and Cleaver 2014). Overall, 56% of pines in plots were dead, and 46% of severely diseased pines specifically tagged for monitoring died between 2013 and 2014 (Mulvey, written communication, 7 May 2015). The USFS mapped approximately 1,820 ha (4,500 ac) of damage near Gustavus and within GLBA, during 2012 and 2013 aerial forest health surveys (USFS 2012, 2013).

4.19.5. Current Condition and Trend

Species Richness

The NPS certified species list for GLBA's vascular plants (NPS 2015) includes 594 taxa (not counting species with a status of "unconfirmed"). The majority (530, or 89%) have been documented as present while the remaining 64 are considered "probably present." Over 330 of these were identified during the park's vascular plant inventory (Carlson et al. 2004). Prior to this inventory, around 260 species had been documented within the park (Carlson et al. 2004). The park's certified species list for non-vascular plants includes 98 taxa (NPS 2015). However, only 49 of these have been confirmed as present in the park; the remaining species are considered probably present (18 taxa) or unconfirmed (23 taxa). Given that the remoteness of some areas of GLBA have limited access for research and inventories, it is possible that the known species richness for GLBA could increase with further survey efforts.

Although not technically plants, lichens are an important component of GLBA's floral diversity (Sharman, written communication, 29 April 2015). Lichens are a symbiotic life form with algal and fungal components (Photo 47; USFS 2015a). A 2011-2012 comprehensive inventory of select park areas documented 878 species, at least 30 of which are thought to be new to science (plus one genus new to science) (Spribille and Fryday 2012, Spribille *In prep*).

Photo 47. *Platismatia glauca*, a lichen known to occur in GLBA (NPS photo).

<u>Number of Non-native Invasive Species</u>

The number of non-native plant species documented within GLBA has increased over time. During the first EPMT survey of the park in 2004, only 14 non-native species were detected. By the end of the 2010 EPMT season, 46 species had been confirmed within the park (NPS 2013a). As of 2013, 49 different non-native plant taxa had been documented within GLBA boundaries by EPMT surveys; an additional 13 species have been found in Gustavus, which is just outside park boundaries (NPS 2013a). These plant species and the years they have been observed are listed in Appendix J. The time spent on these efforts by the EPMT, along with the spatial extent of surveys, varied between years, so it would likely not be appropriate to compare results between years. Any apparent trends in number of species could simply be due to differences in survey effort or location as opposed to actual change over time. Two species found in the park (*Elymus repens*, *Sonchus arvensis*) and three found in Gustavus (*Cirsium arvense*, *Galeopsis tetrahit*, *Hieracium aurantiacum*) are classified as prohibited noxious weeds by the state of Alaska (AK DNR 2010).

<u>Distribution of Non-native Invasive Species</u>

The locations of all AK EPMT invasive species observations in GLBA from 2004-2012 are shown in Figure 153. The majority of invasive species have been found in developed areas of the park, particularly in Bartlett Cove and Dry Bay; common dandelion (*Taraxacum officinale* ssp. *officinale*) appears to be the only species that has become established in the backcountry (Fisk 2011). The areas surveyed and time spent on these efforts by the EPMT varied between years, so it would likely not be appropriate to compare results between years. Any apparent trends in distribution could simply be due to differences in survey effort or location as opposed to actual change over time.

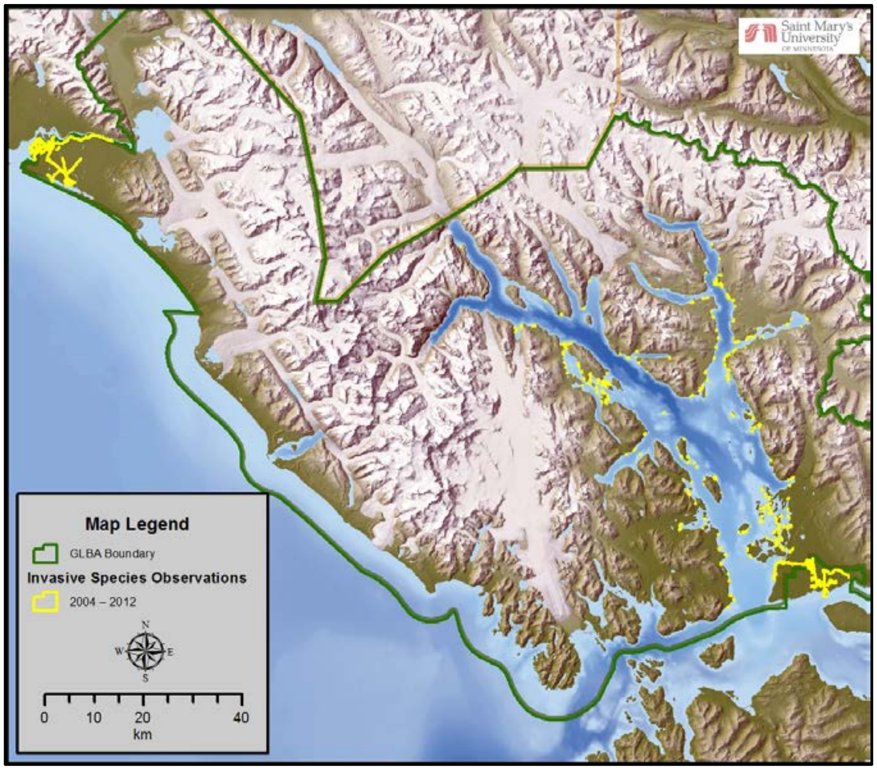

Figure 153. Locations of all invasive plant observations in GLBA by the AK EPMT, 2004-2012 (NPS 2013a).

The most widely documented invasive species in the park is the common dandelion (Figure 154). Common mouse-ear chickweed (*Cerastium fontanum*) has also been documented in a few places outside the developed areas of the park (Figure 154). Common plantain (*Plantago major*) and several invasive grasses (*Phleum pratense*, *Triticum aestivum*, and *Phalaris arundinacea*) are also frequently observed between the Bartlett Cove developed area and Gustavus (Figure 155 and Figure 156).

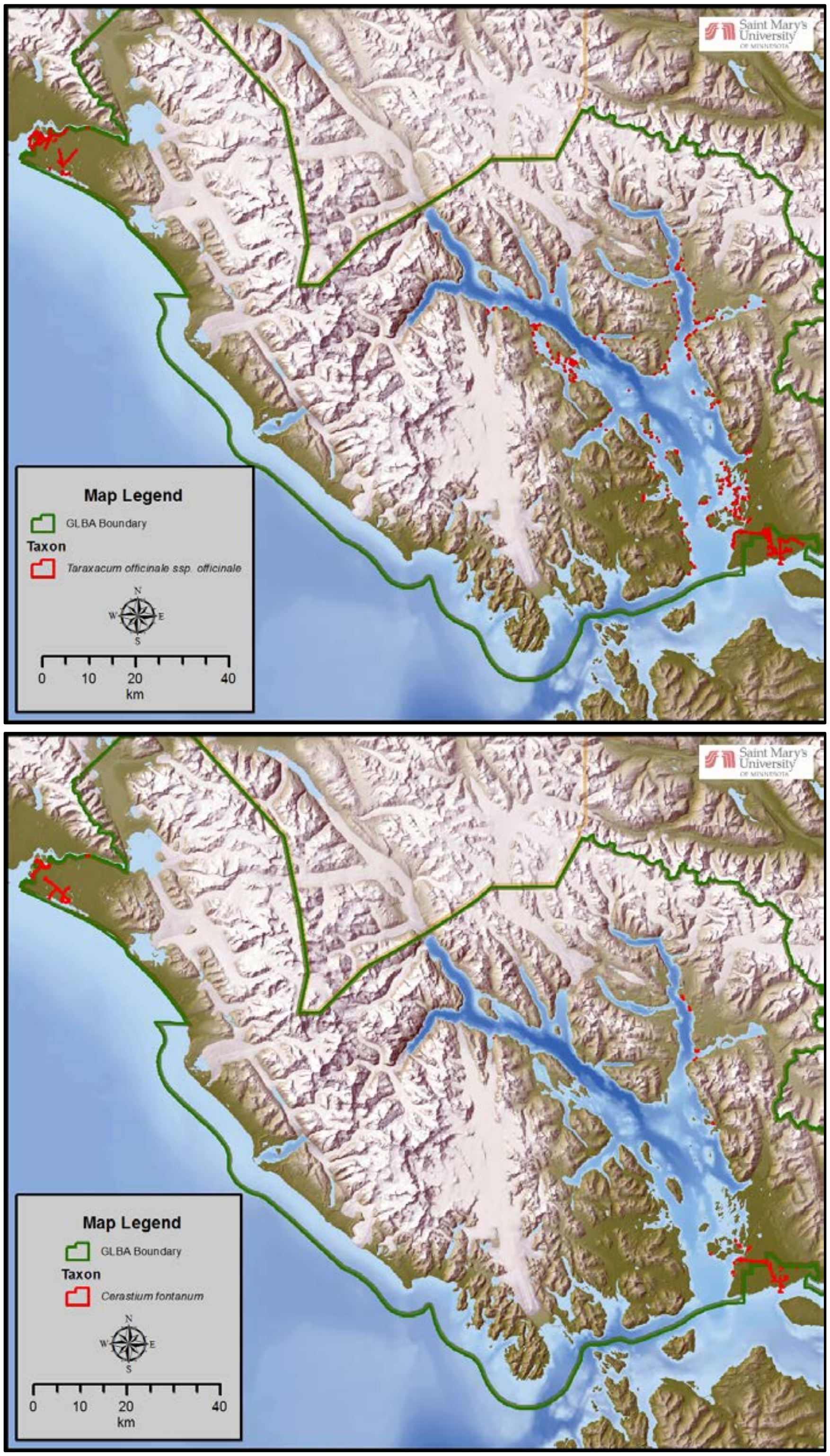

Figure 154. Confirmed locations of common dandelion (top) and common mouse-ear chickweed (bottom) within GLBA, 2004-2012 (NPS 2013a).

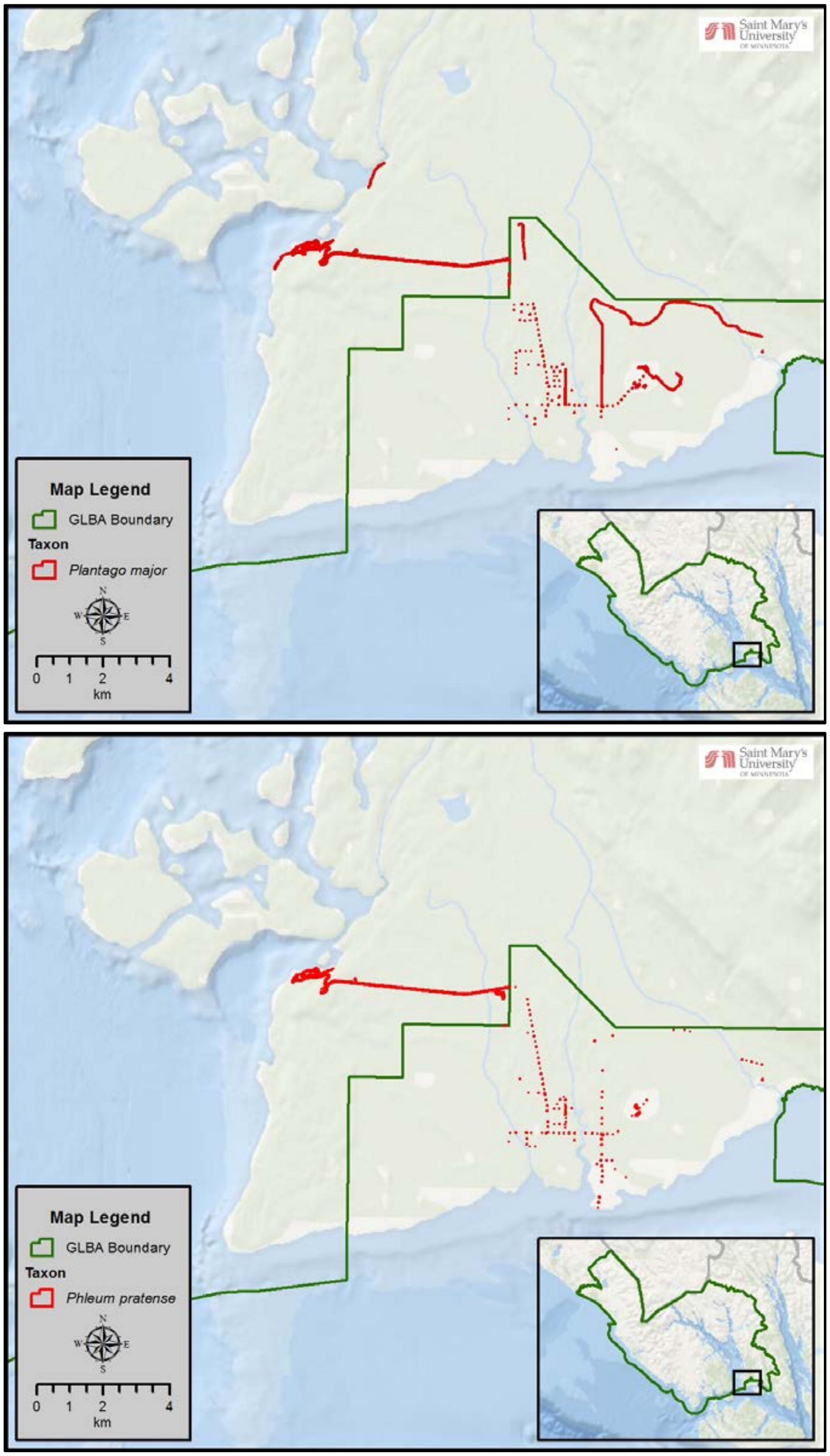

Figure 155. Distribution of common plantain (top) and timothy (bottom) around Gustavus and Bartlett Cove (NPS 2013a).

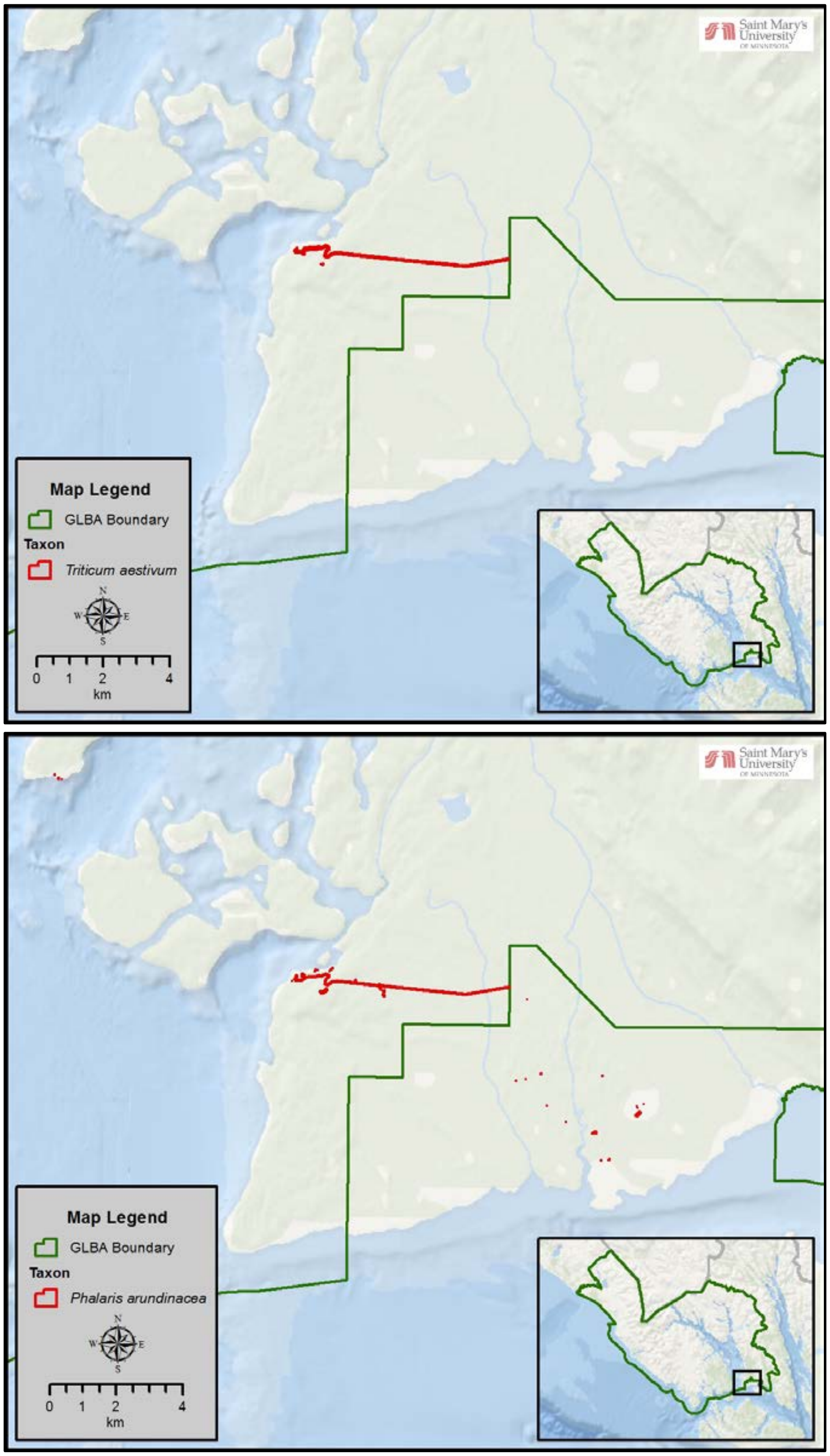

Figure 156. Distribution of common wheat (top) and reed canary grass (bottom) around Gustavus and Bartlett Cove (NPS 2013a).

Sitka Spruce Mortality Distribution

Sitka spruce is one of the most dominant tree species in the forests surrounding Glacier Bay (Moynahan et al. 2008). The leading cause of spruce mortality in southeastern Alaska is spruce beetle damage (Photo 48). Additional causes of mortality include stem decay pathogens (e.g., fungal infections) and windstorm damage, which are often associated with beetle outbreaks (Eglitis 1982, USFS 2013). The two main stem decay (heart rot) pathogens of Sitka spruce are *Phaeolus schweinitzii* and *Fomitopsis pinicola* (Mulvey, written communication, 7 May 2015). Both are brown rots commonly associated with bole and root wounds, and infection typically increases with tree diameter (Kimmey 1956). Spruce mortality from bark beetle and stem decay pathogens can create hazardous trees in developed recreation areas, requiring regular monitoring and occasional treatment (Mulvey, written communication, 7 May 2015).

Photo 48. Spruce beetle adult and larva (USFS photo).

The spruce beetle outbreak that peaked in the 1980s impacted nearly 14,000 ha (34,595 ac) of forest in lower Glacier Bay around the Sitakaday Narrows and into the Excursion Ridge area by 1996 (Schultz and Hennon 2007). Researchers believe the outbreak began in the late 1970s; at that time, the lower bay area consisted of relatively dense, even-aged (120-140 years) stands dominated by Sitka spruce (Eglitis 1982, Schultz and Hennon 2007). These spruce trees were likely some of the first trees to colonize land exposed by glacial recession. At that age and stage in the successional process, spruce growth often slows and tree vigor is reduced as soil nutrient availability is depleted (Schultz and Hennon 2007). This reduced vigor, resulting from nutrient limitation exacerbated by intraspecific competition, likely increased the susceptibility of these stands to bark beetle infestation. In addition, favorable conditions for bark beetles were created by extensive blow-downs from wind in the late 1970s and dry years in the early 1980s (Schultz and Hennon 2007). These conditions are ideal for beetle development and survival, allowing populations to build up to an outbreak level (Elizabeth Graham, USFS Entomologist, written communication, 15 May 2015). When the USFS first surveyed the outbreak area in 1982, they found 1,068 ha (2,640 ac) of extensive spruce mortality on the west side of the Sitakaday Narrows (Eglitis 1982). Beetle damage was also observed on approximately 324 ha (800 ac) on western Young Island, 69 ha (170 ac) on Strawberry Island, and 24 ha (60 ac) on western Lester Island, for a total infested area over 1,485 ha (3,670 ac) (Eglitis 1982). Eglitis (1982) noted that beetles did not preferentially attack the largest-diameter trees, as had been observed elsewhere in previous outbreaks, and that infestation levels were not correlated with stand density or individual tree growth. The extent of the outbreak according to1984 aerial photography is shown in Figure 157.

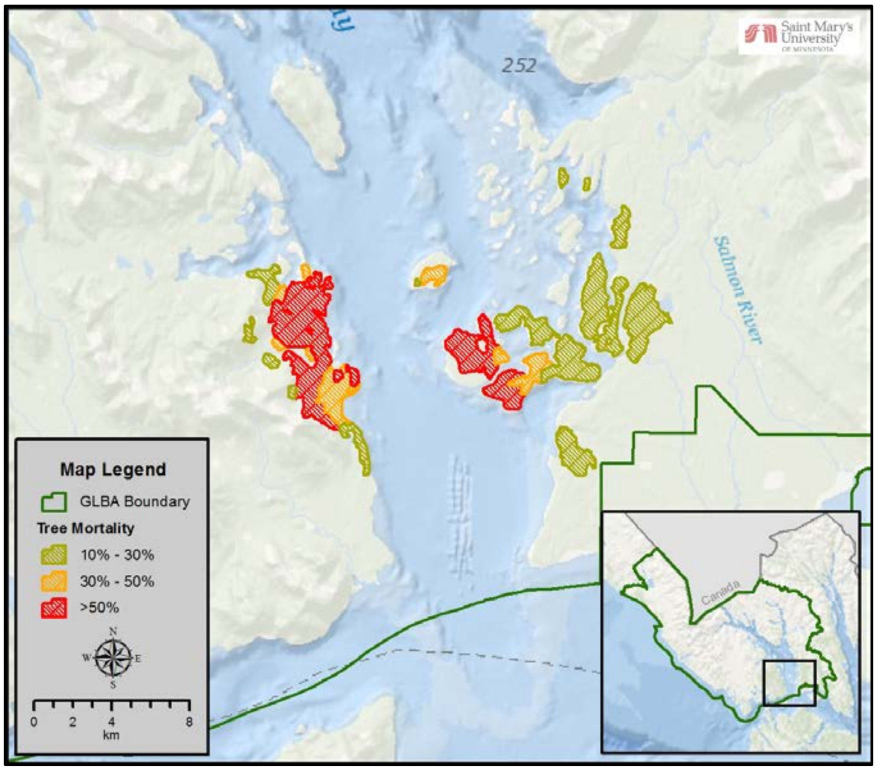

Figure 157. Spruce mortality extent in lower Glacier Bay in 1984 (Eglitis 1986).

By 1985, the outbreak had expanded to nearly 5,000 ha (12,300 ac) and included areas south of Bartlett Cove and near the Bartlett River, although the rate of spread seemed to be declining (Eglitis 1986). In 1987, aerial surveys indicated that 6,070 ha (15,000 ac) were impacted, with spruce mortality in the area ranging from 5-70% (Eglitis 1987). Eglitis (1987, 1988) summarized the total affected area in 1986 and 1987, broken out by percent mortality (Table 57). During both years, over 40% of the affected area was experiencing at least 50% spruce mortality. The distribution of spruce mortality within GLBA in 1987 is shown in Figure 158.

Table 57. Area (ha) within GLBA experiencing spruce mortality in 1987 and 1988, by percent mortality (Eglitis 1987, 1988).

Year	Area Affected (ha), by Percent of Spruce Mortality											Total
	70%	60%	50%	40%	35%	30%	25%	20%	15%	10%	5%	
1986	724	494	1,194	239	490	198	--	1,117	1,072	312	53	**5,892**
1987	716	700	1,056	231	490	36	571	1,214	567	130	77	**5,787**

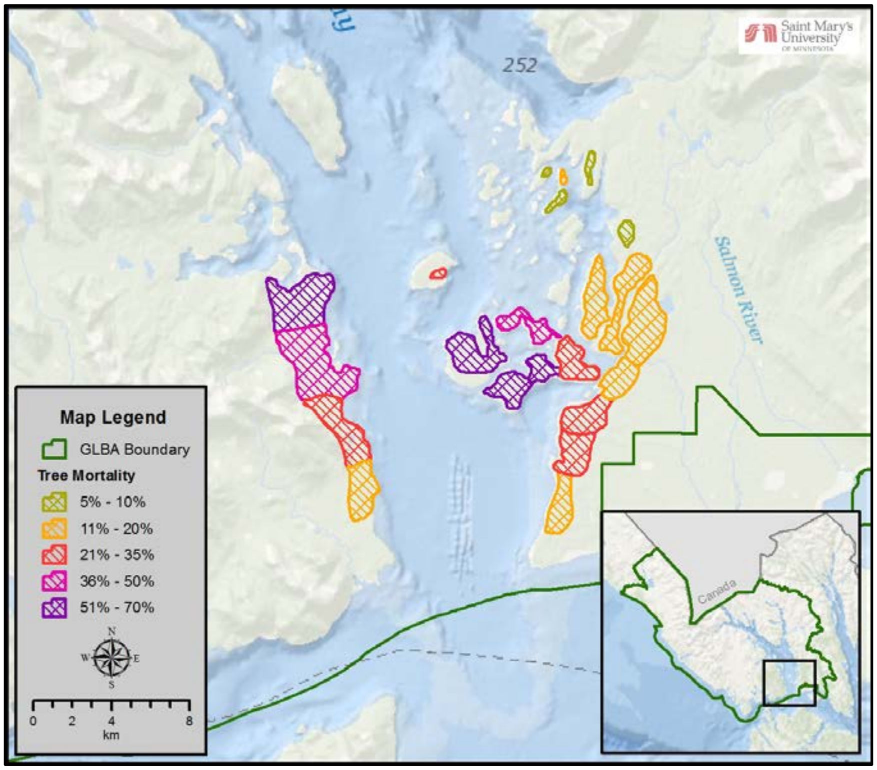

Figure 158. Spruce mortality extent in lower Glacier Bay in 1987 (Eglitis 1988).

Eglitis (1988) also summarized the condition of spruce in 45 monitoring plots within the outbreak as of 1987 (Table 58). Mortality in these plots ranged from 33% in Ripple Cove to 67% on Young Island, with an average mortality of 52% (Eglitis 1988). Although not documented within monitoring plots, Eglitis (1988) reported that mortality exceeded 70% in some stands near Berg Bay.

Table 58. Condition of spruce trees in 1987 within 45 GLBA monitoring plots (Eglitis 1988).

Area	Live Spruce			Dead	Windthrown[B]	% Spruce Dead	Total Spruce
	Unattacked	Pitchout[A]	Dead Strip				
Berg Bay	27	49	6	42	0	34	**124**
Ripple Cove	101	88	8	94	1	33	**292**

[A] "pitchout" is when beetles initially attack a tree but abandon it before any portion of the tree dies.

[B] Trees that were alive the previous year but were windthrown in the winter of 1986-1987.

Table 58 (continued). Condition of spruce trees in 1987 within 45 GLBA monitoring plots (Eglitis 1988).

Area	Live Spruce			Dead	Windthrown[B]	% Spruce Dead	Total Spruce
	Unattacked	Pitchout[A]	Dead Strip				
Lester Island	115	258	39	572	53	60	**1,037**
Bartlett Lake Trail	15	18	4	48	0	48	**71**
S. Bartlett Cove	52	67	13	36	1	36	**206**
Young Island	13	41	10	67	12	67	**193**
Total	**323**	**521**	**80**	**52**	**67**	**52**	**1,923**

[A] "pitchout" is when beetles initially attack a tree but abandon it before any portion of the tree dies.

[B] Trees that were alive the previous year but were windthrown in the winter of 1986-1987.

While the beetle infestation slowed during the 1990s, spruce mortality continued to expand through 1996 (Schultz 2004). Schultz and Hennon (2007) noted that spruce beetles may have served as a vector for the decay fungus *Fomitopsis pinicola*. Although this fungus is ubiquitous on wounded and dead conifers throughout southeastern Alaska, spruce trees that were attacked during the outbreak developed *F. pinicola* fruiting bodies and decayed faster than trees that died before or after the outbreak (Schultz and Hennon 2007). The beetle outbreak subsided in the park in the early 2000s, but the mortality it caused initiated a rapid transition from forests that had been homogenous (dominated by even-aged Sitka spruce) to forests that are more structurally and biologically complex, and more similar to old-growth stands along the Alaskan coast (Schultz and Hennon 2007).

Spruce beetle damage and Sitka spruce mortality has been low throughout southeastern Alaska in recent years (USFS 2012, 2013). In 2012, USFS surveys recorded the lowest annual spruce beetle-caused mortality since surveys began in the 1970s (USFS 2013). No beetle activity was detected within GLBA boundaries by aerial surveys in 2011, 2012, or 2014, and only one small area of damage was noted along the Pacific coast in 2013 (USFS 2015b; Figure 159). The small 2013-2014 outbreak towards the top of the map in Figure 159 is north of the park.

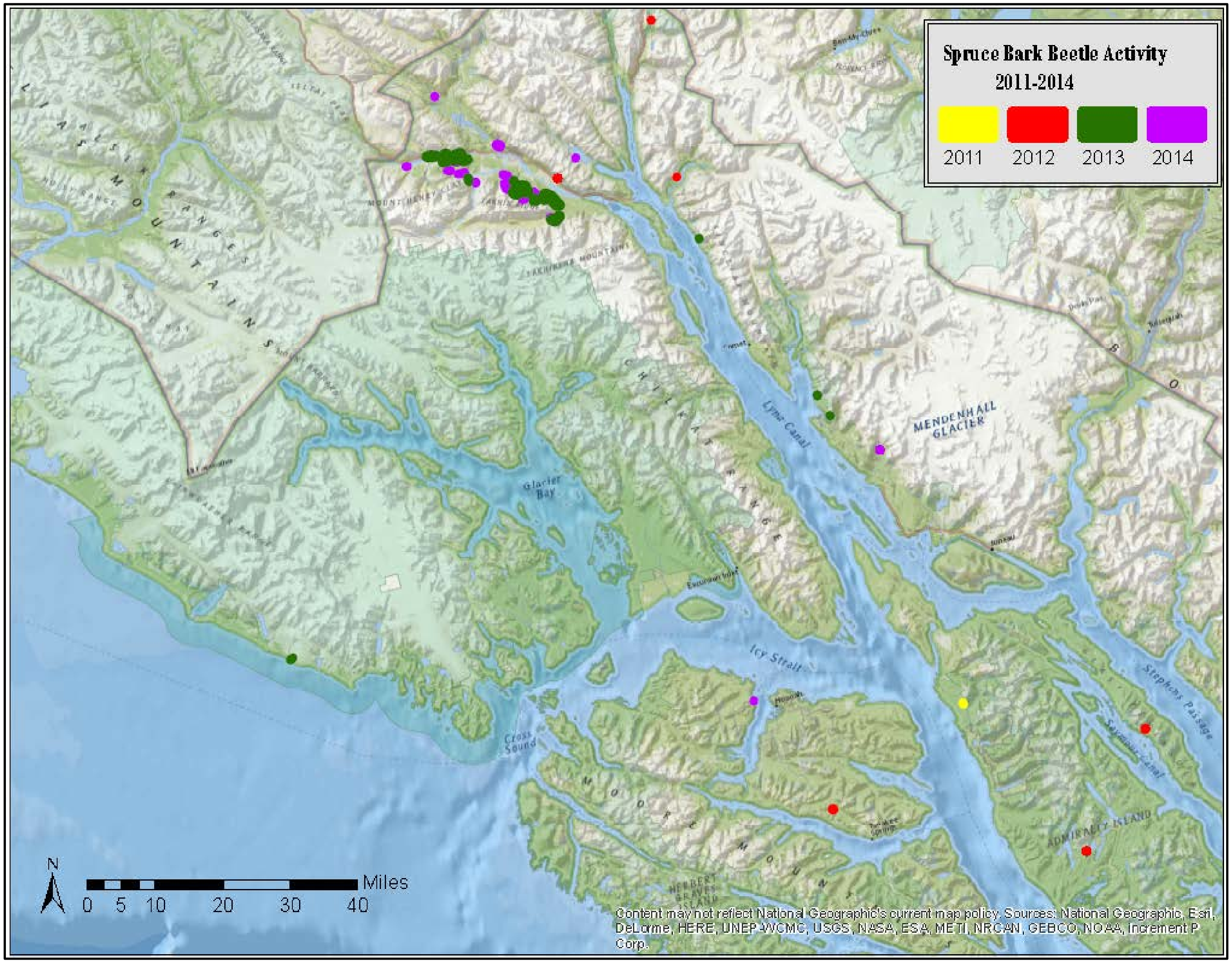

Figure 159. Spruce bark beetle damage detected in the GLBA area by USFS aerial surveys, 2011-2014 (map provided by the USFS Alaska Region [USFS 2015b]).

Threats and Stressor Factors

The primary threats to GLBA's plant communities are climate change and vectors that introduce and/or spread invasive plants. Air pollution from cruise ships associated with increased tourism throughout Southeast Alaska may also be a concern. Although plumes of exhaust are often observed from ships travelling through the fjords of GLBA, emissions can become trapped by inversions that create haze and affect visibility. These impacts tend to be temporary, lasting for several hours to a day (Molders et al. 2013, Pirhalla et al. 2014). Nitrogen and sulfur deposition are known to be elevated at Sitka National Historical Park south of GLBA, another popular cruise ship destination (Geiser et al. 2010).

Climate is a primary driver of vegetation distribution and abundance (Hennon et al. 2012). GLBA's climate is predicted to become warmer and drier during this century (SNAP et al. 2009). Temperatures are expected to increase, with the most dramatic change occurring in the winter. These increased temperatures are likely to lengthen the growing season, impact plant phenology, and influence soil water availability (SNAP et al. 2009). While precipitation is projected to increase, evapotranspiration will also likely increase due to warmer temperatures and a longer growing season. This means water will be used by plants or will evaporate back into the atmosphere faster and will not be stored in the soil or on its surface as long. As a result, the area will seem drier, particularly in summer and fall. This could increase wildfire risk and contribute to wetland drying (SNAP et al. 2009). Changes in vegetation as a result of these climate shifts will also impact wildlife distribution and habitat use (Moynahan et al. 2008).

Climate also influences the diseases and insect pests that impact many tree species in southeastern Alaska. For example, weather plays a key role in bark beetle population dynamics, which favor warm, dry weather, particularly in spring (Mulvey, written communication, 7 May 2015). The life cycle of the spruce beetle is typically 2 years, but warmer and longer seasons can allow the beetles to complete their life cycles in 1 year (Holsten 1999). Projected increased temperature or other changes that prolong needle wetness during the growing season could also favor pathogens that affect shore pine, such as western gall rust and Dothistroma needle blight (VanLeuven 2014; Mulvey, written communication, 7 May 2015). Caused by a fungus, Dothistroma needle blight affects a wide variety of pines and is considered one of the most important pine diseases worldwide (Woods et al. 2005, USFS 2013). Infected trees exhibit needle discoloration and may have sparse crowns and reduced growth due to premature needle shed (Photo 49; USFS 2013). Successive years of severe foliage disease, as have recently been observed within localized areas within the park, can directly kill trees (Mulvey, written communication, 7 May 2015). This stress can also increase a tree's vulnerability to the secondary bark beetle (*Pseudips mexicanus*), which only successfully attacks trees weakened by other factors (Mulvey and Cleaver 2014; Mulvey, written communication, 7 May 2015).

Photo 49. Shore pine affected by Dothistroma needle blight (left) and a close-up of one-year-old shore pine needles with *Dothistroma septosporum* fruiting bodies (small black dots) (right) (USFS photos).

An extended Dothistroma needle blight outbreak that began near Gustavus in 2010 has caused shore pine mortality (USFS 2013, Mulvey and Cleaver 2014). In study plots established by Mulvey and Cleaver (2014) around Gustavus, 115 of 204 shore pine (56%) have died due to foliage disease, and additional mortality is expected. According to estimates from 2013-2014 aerial surveys of the area, 1,700-1,900 ha (4,200-4,700 ac) are affected by severe Dothistroma needle blight (Figure 160; Mulvey and Cleaver 2014). Damage has been observed in the park along Rink Creek Road and the park entrance road (NPS 2013c). Studies show that the disease is driven by temperature and moisture. Severe outbreaks in British Columbia have been linked to local increases in summer precipitation by one study (Woods et al. 2005) and to increased August minimum temperatures and increased spring precipitation in another (Welsh et al. 2014).

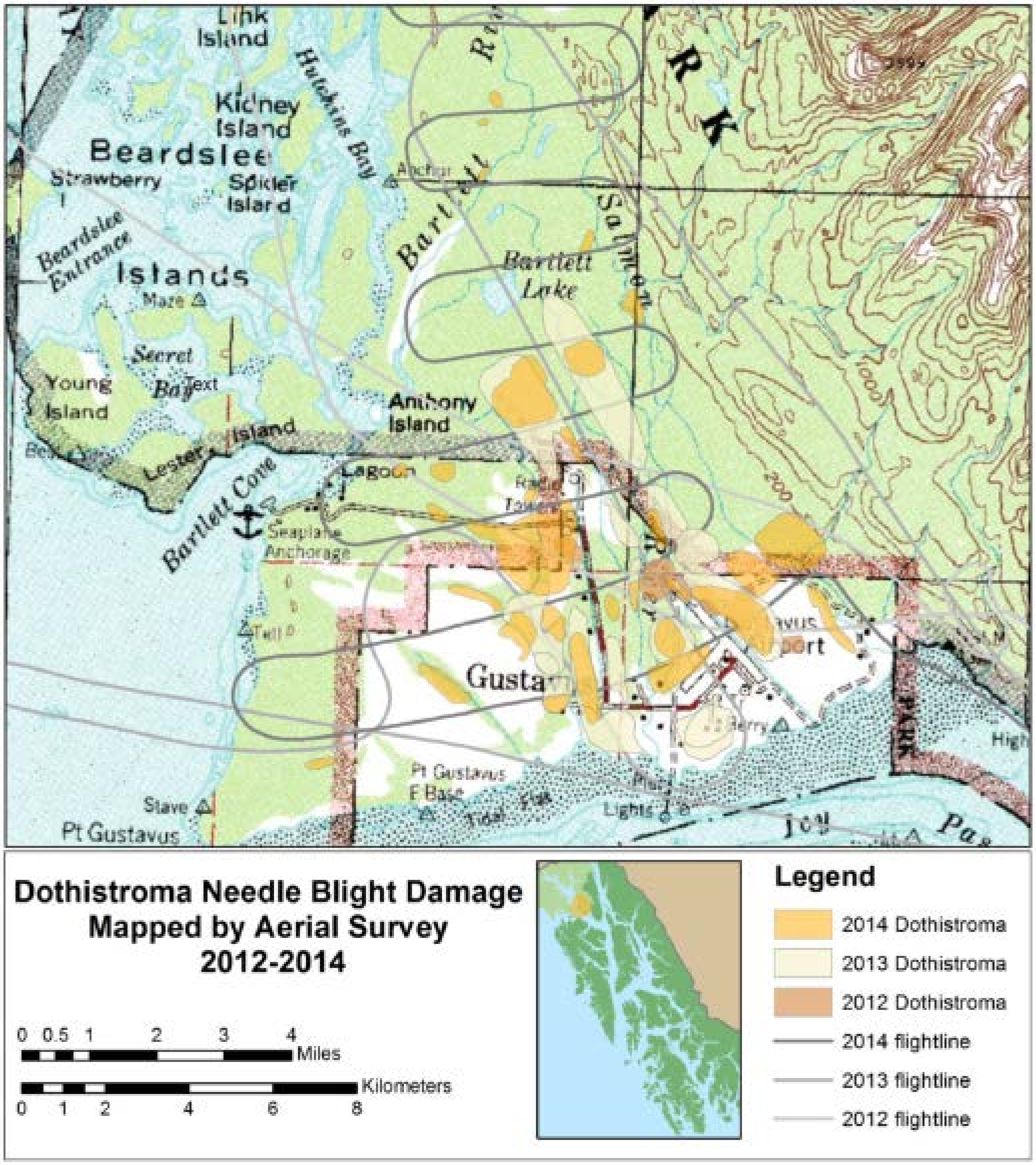

Figure 160. Shore pine stands impacted by Dothistroma needle blight; the shaded red line represents the GLBA boundary (Mulvey and Cleaver 2014).

Yellow-cedar has been experiencing widespread mortality in parts of Southeast Alaska and British Columbia. The dieback, commonly referred to as yellow-cedar decline, has been linked to climate change (Beier et al. 2008, Hennon et al. 2012, NPS 2013b). Yellow-cedar is an ecologically, economically, and culturally important species of Alaska's coastal forests (Hennon et al. 2012). It has few natural pests, can live for more than 1,000 years, and produces particularly durable wood (Harris 1990, Hennon et al. 2012). This decline has impacted 200,000 ha (494,209 ac) of temperate forest in Southeast Alaska, primarily at low elevations (Beier et al. 2008). Symptoms begin with foliage color changes (yellow, then brown), and tree mortality typically occurs within 3-15 years (Beier et al. 2008). The decline began in the 1880s and experienced a peak in the 1970s and 1980s during a warm period in the PDO, a regional climate pattern similar to El Niño (Hennon et al. 2012). Research to date indicates a complex mechanistic pathway to decline, where fine root damage occurs from soil freezing and lack of snow cover increases risk to sudden cold events (Hennon et al. 2012; Lauren Oakes, Stanford PhD candidate, written communication, 7 May 2015). While tree roots normally "harden" to protect against freezing temperatures in the winter, yellow-cedar roots appear to "deharden" earlier than other conifer species, making them more vulnerable to freezing injury in late winter/early spring (Hennon et al. 2012). Snow typically insulates soils and plant roots, but climate shifts over the past century have decreased snowpack at lower elevations in southeastern Alaska (Beier et al. 2008). When snowpack was removed from potted yellow-cedar saplings grown outdoors in Juneau during a 2006 thaw-freeze cycle, the saplings experienced 100% fine root damage and mortality by the end of spring (Schaberg et al. 2008). At low elevations, yellow-cedar often grows in poorly-drained soils, where shallow rooting depth and relatively saturated soils make the roots even more vulnerable to freezing (Beier et al. 2008). The shallow rooting zone of soils experiences greater temperature variability, including diurnal fluctuations and frequent below freezing temperatures, particularly when snow is not present in the winter (Hennon et al. 2012). Yellow-cedar roots are protected from freezing injury on sites with deeper, more productive soils, where roots are sufficiently deep to avoid extreme temperature fluctuations of the near-surface soil horizons (Mulvey, written communication, 7 May 2015). Canopy cover may reduce vulnerability to root damage by protecting snowpack from solar radiation and preventing early snow melt (Hennon et al. 2012, NPS 2013b). Yellow-cedar stands may be unlikely to recover from this decline, as Oakes et al. (2014) found that the species is significantly less likely to regenerate in stands that experienced extensive mortality.

Currently, the latitudinal limit of yellow-cedar decline is located just south of GLBA (Oakes et al. 2014) and stands within the park appear healthy at this time (Figure 161; Oakes et al. 2015). However, coastal Alaskan forests are predicted to experience the largest increase in frost-free days this century of any location in North America (Meehl et al. 2004). Winter temperatures will likely increase from an average just below freezing during the twentieth century to an average above freezing, which would reduce snow accumulation (Hennon et al. 2012). As a result, researchers believe that yellow-cedar decline will expand into new areas where snow cover is insufficient and may eventually impact stands further upslope (Beier et al. 2008). If yellow-cedar stands are impacted in GLBA, this may also influence wildlife due to changes in forage and habitat (NPS 2013b, Oakes et al. 2015). A vulnerability assessment was conducted to assess the future vulnerability of yellow-cedar inside GLBA to decline (Oakes, written communication, 7 May 2015). The model considered well-documented risk factors for mortality (soil drainage, snow cover) under climate scenarios (present to 2099). Results indicate an expected increase of yellow-cedar forests highly vulnerable to decline from 3% in the 2020s to 27% by the 2080s, developing from east to west along the outer coast and upward in elevation (Oakes et al. 2015).

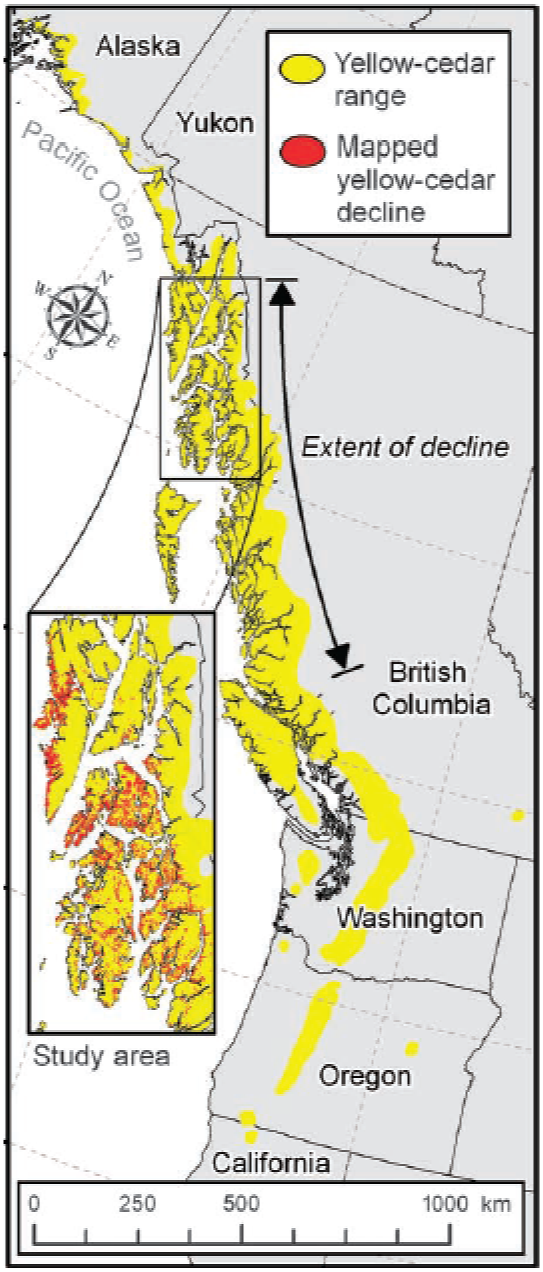

Figure 161. Distribution of yellow-cedar and extent of mapped decline (Hennon et al. 2012). GLBA lies just outside the northern edge of the smaller inset map.

Invasive plants are a threat to native vegetation communities because of their potential to displace native species and to alter ecological processes (NPS 1996). GLBA has been largely protected from non-native plant invasions by its isolation from typical human-related vectors such as roads, trails, and development. These activities will increase the risk of invasive species introduction. Vulnerability to invasive species is not limited to areas impacted by humans. Glacial recession can also expose open ground that could be colonized by opportunistic invasive plants (Fisk 2011). To date, most invasive plants documented inside park boundaries have been in developed areas around Bartlett Cove (Fisk 2011) and the Dry Bay Ranger Station (Carlson et al. 2004). Until 2011, all invasive species control efforts in and around GLBA had been limited to manual removal (e.g., pulling and cutting) by AK EPMT teams; beginning in 2011, several infestations were targeted for herbicide treatments (Fisk 2011). In 2013, the park shifted its focus from mapping and manual control by seasonal EPMT teams toward more serious efforts to eradicate several problem species with herbicide treatments in selected areas. These include perennial sow thistle (*Sonchus arvensis*; Photo 50) on Strawberry Island, and tall buttercup (*Ranunculus acris*), creeping buttercup (*R. repens*), and reed canary grass in the Bartlett Cove Developed Area (Chris Overbaugh, Alaska Exotic Plant Management Team Liaison, phone communication, 22 January 2015). In just one year, herbicide treatments have reduced patches of these species by over 50%, and by as much as 95% in some areas (Overbaugh, phone communication, 22 January 2015).

Photo 50. Perennial sow thistle (NPS photo).

<u>Data Needs/Gaps</u>

Additional surveys could expand knowledge regarding species richness and invasive species distribution within the park. Carlson et al. (2004) recommended inventories in the following locations and/or habitats to achieve a more complete picture of species richness: high-elevation calcareous regions of Excursion Ridge/Chilkat Range, Cape Spencer fjords, high elevations of the Dry Bay forelands and Alsek River corridor, and nunataks in the Fairweather Range.

Schultz and Hennon (2007) noted that additional spruce forests within GLBA will soon be reaching the age class at which the lower bay stands experienced the 1980s beetle outbreak. These forests should be monitored to see if they experience a similar disturbance from spruce beetles at the same successional stage (Schultz and Hennon 2007). Soil surveys could also test whether nitrogen deficiency, which contributed to the 1980s outbreak, is potentially a predisposing factor in these forests (Mulvey, written communication, 7 May 2015).

Knowledge regarding shore pine and biological threats to the species is limited, particularly information on foliar and canker pathogens, as well as pathogenic insects and fungi (Mulvey 2011). Research is also needed into the causes of bole wounding in shore pine; likely sources include animal damage, breakage from snow loading, and a possible bole canker pathogen (Photo 51; USFS 2013). An investigation into whether shore pine and these biological threats are influenced by climate change would be useful as well (Mulvey and Cleaver 2014, VanLeuven 2014). Continued monitoring of shore pine in the area of the recent Dothistroma outbreak is also recommended (Mulvey, written communication, 7 May 2015).

Photo 51. Bole wounds on a shore pine thought to be caused by a canker pathogen (USFS photo).

While yellow-cedar stands within the park are known to be relatively healthy at present compared to those farther south (Oakes et al. 2014, Oakes et al. 2015) this species should be monitored for any sign that the decline experienced in the south is spreading northward.

Overall Condition

Species Richness

The project team assigned the species richness measure a *Significance Level* of 3. The NPS certified species list for vascular plants includes 594 taxa (NPS 2015). Prior to a vascular plant inventory in the early 2000s, around 260 plant species had been documented within the park (Carlson et al. 2004). Forty-nine non-vascular plants have been confirmed present in the park (NPS 2015). Considering the remoteness of some areas of GLBA, it is possible that further survey efforts could increase the known species richness. The *Condition Level* for this measure is a 0, indicating no concern at this time.

Number of Non-native Invasive Species

This measure was assigned a *Significance Level* of 3. As of 2013, 49 non-native taxa had been documented within GLBA, with an additional 13 species found in Gustavus, just outside the park (NPS 2013a). The number of invasive species within the park has increased since the first AK EPMT survey in 2004 which identified only 14 species and the number could continue to climb if on-the-ground visitor use increases (Fisk 2011). As a result, this measure is assigned a *Condition Level* of 2 for moderate concern.

Distribution of Non-native Invasive Species

The distribution of invasive species measure was also assigned a *Significance Level* of 3. The majority of invasive plants are limited to the developed areas of GLBA, primarily around Bartlett Cove and Dry Bay. However, given their potential to alter the park's native vegetation communities, these species are seen as a significant threat. Therefore, this measure is assigned a *Condition Level* of 2, indicating moderate concern.

Sitka Spruce Mortality Distribution

A *Significance Level* of 2 was assigned for this measure. The lower bay area within GLBA experienced significant Sitka spruce mortality in the 1980s and 1990s due to a severe spruce beetle outbreak. However, this disturbance appears to be a natural part of forest succession, as the forests in impacted areas are now more complex and similar to old-growth stands in other parts of Alaska (Eglitis 1982, Schultz and Hennon 2007). Spruce beetle-related damage has been very low in recent years (USFS 2012, 2013). Between 2011 and 2014, only one small area of spruce damage was detected within GLBA in one year (USFS 2015b, Figure 159). Because of this, Sitka spruce mortality is currently of low concern (*Condition Level* = 1).

Yellow-cedar Distribution

Yellow-cedar distribution was assigned a *Significance Level* of 1. Measures with a *Significance Level* of 1 are not discussed in depth in the current condition section of this assessment, but available information is summarized here in the overall condition section. In Alaska, yellow-cedar occurs near the coast, from sea level nearly up to the timberline (Beier et al. 2008). GLBA is at the northern extent of the species range (see Figure 161). Modeling from recent aerial surveys estimate that yellow-cedar occurs on 23,968 ha (approximately 60,000 ac) within the park (Oakes et al. 2015). Maps of their locations were still in development by the USFS during the writing of this assessment. Given that the yellow-cedar stands within the park all appear healthy at present (NPS 2013b), there is currently low concern regarding this measure (*Condition Level* = 1). However, future monitoring is critical given the potential increasing vulnerability of yellow-cedar to dieback (Oakes, written communication, 7 May 2015).

Yellow-cedar Stand Density

The project team also assigned a *Significance Level* of 1 for this measure. Very little is known about density within GLBA's yellow-cedar stands. Oakes et al. (2014) collected density data from 18 healthy yellow-cedar stands (i.e., not impacted by decline). Ten of these stands were within GLBA (see Figure 152) and eight were on Chichagof Island in the Tongass National Forest, south of GLBA. The densities of various species within these stands are provided in Table 59; note that these values are means for all 18 stands, including those on Chichagof Island, and are not exclusive to GLBA stands. These data may serve as a baseline for future studies of yellow-cedar density within the park, but do not provide enough information to assign a *Condition Level* at this time.

Table 59. Density (stems/ha) of live and standing dead trees in three size classes within forest stands showing no sign of yellow-cedar decline (Oakes et al. 2014). Ten of these stands were within GLBA and eight were on Chichagof Island, south of GLBA.

	Treelets[A]		Small Trees[B]		Big Trees[C]	
Common Name	**Live**	**Dead**	**Live**	**Dead**	**Live**	**Dead**
yellow-cedar	510	210	695	165	177	15
western hemlock	230	–	255	–	113	–
mountain hemlock	80	–	65	–	33	–
dead hemlock (western+mountain)	–	110	–	100	–	13
Sitka spruce	60	10	50	15	15	7
shore pine	5	0	25	5	13	3
Total	**885**	**335**	**1,090**	**290**	**352**	**40**

[A] Treelets = 2.5-9.9 cm dbh

[B] Small trees = 10-24.9 cm dbh

[C] Big trees ≥ 25.0 cm dbh

Yellow-cedar Age Structure

Yellow-cedar age structure was assigned a *Significance Level* of 1 as well. Forest age structure data can help identify causes for concern within ecosystems; for example, an absence of young age classes suggests the absence of regeneration, which is essential for forest sustainability. Yellow-cedar age structure has not been studied in GLBA; therefore, a *Condition Level* cannot be assigned for this measure.

Tree Density in Shore Pine Stands

This measure was assigned a *Significance Level* of 1. Shore pines play an important role in forested wetlands such as the muskegs of Southeast Alaska, providing habitat and a food source in harsh environmental conditions (e.g., high water table and high soil acidity) where few tree species can survive (Mulvey 2011). In the area around Gustavus, shore pine also occurs in mixed, even-aged stands with Sitka spruce and cottonwood; these stands occur in areas uplifted following glacial retreat in recent centuries. Overall tree density in the mixed stands is much higher than in the open muskegs where shore pine typically occurs (Mulvey and Cleaver 2014). While the recent

Dothistroma needle blight outbreak has affected shore pine in both the dense mixed stands and pure shore pine stands on more open muskeg sites, researchers predict that the ecological impacts will be more lasting in the dense mixed stands (Mulvey, email communication, 26 January 2015). Higher tree densities may contribute to an environment conducive to disease development, as moisture persists longer in stands with lower air circulation and sun exposure (Mulvey and Cleaver 2014). Shore pine require high light levels for regeneration, so the species is unlikely to recover in mixed stands where young pines are shaded by a canopy of other tree species; however, successful regeneration will likely continue on the open muskeg sites, allowing the species to remain dominant in those areas (USFS 2013, Mulvey and Cleaver 2014).

Due to this likely connection between stand density and shore pine condition, GLBA park managers are interested in monitoring tree density in shore pine stands. However, none of the shore pine research conducted thus far has focused on density. Based on tree counts in plots of known size from Mulvey and Cleaver's (2014) work just outside of Gustavus (but not within GLBA boundaries), density can be estimated for these sites (Table 60). All plots were located in mixed stands; no sampling was conducted in more open pine stands for comparison. Because these data are extremely limited and are from outside the park, a *Condition Level* could not be assigned for this measure at this time.

Table 60. Tree density and shore pine mortality data from four survey plots (100 x 15 ft) in shore pine-dominated stands near Gustavus (Mulvey and Cleaver 2014).

Plot	Total trees	Density (per 100 ft^2)	Total shore pine	Dead shore pine	% pine mortality
1	87	5.8	51	24	47.1
2	37	2.5	23	13	56.5
3	112	7.5	82	47	57.3
4	68	4.5	48	31	64.6

Weighted Condition Score

The *Weighted Condition Score* for plants in GLBA is 0.42, indicating moderate concern. This level of concern is primarily due to the threat posed by invasive, non-native plants. An overall trend could not be assigned as the trends for different measures vary. Some appear stable (e.g., species richness, Sitka spruce mortality) while others may be declining (e.g., invasive species) or are unknown (e.g., yellow-cedar density and age structure).

Plants			
Measures	Significance Level	Condition Level	WCS = 0.42
Species Richness	3	0	
# of Invasive Species	3	2	
Distribution of Inv. Species	3	2	
Spruce Mortality Dist.	2	1	
Yellow-cedar distribution	1	1	
Yellow-cedar density	1	n/a	
Yellow-cedar age structure	1	n/a	
Density in shore pine stands	1	n/a	

4.19.6. Sources of Expertise

- Lewis Sharman, GLBA Ecologist
- Chris Overbaugh, NPS Alaska Region, Exotic Plant Management Team Liason
- Robin Mulvey, USFS, Southeast Alaska Forest Pathologist
- Lauren Oakes, Stanford University PhD Candidate
- Elizabeth Graham, USFS Entomologist, Alaska Region

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4.20. Western Toad

4.20.1. Description

Amphibians play an important role in ecosystems, as they participate in nutrient recycling, insect control, and provide prey for several predatory species (BER and EWC 2004, NPS 2009). Due to their dependence on aquatic habitats, amphibians are also important indicators of ecosystem health (Anderson 2004). The western toad is an important amphibian found in GLBA, and has been identified as a Vital Sign in SEAN (Moynahan et al. 2008; Photo 52).

Photo 52. Western toad (NPS photo).

Western toad distribution extends from Southcentral and Southeast Alaska through western portions of Canada and the U.S. (Fairchild 2008, MacDonald 2010, ADFG 2014b) and south to Northern Baja California (IUCN 2015). Western toads, which are also known as boreal toads, have been found at elevations ranging from sea level to the high mountain elevations below 3,640 m above sea level (11,942 ft) (ADFG 2014b). Western toads utilize several habitat types (ADFG 2014b), and gather in shallow breeding ponds in the spring and later move to upland wetland sites. In winter the toads hibernate in terrestrial forests (NPS 2009).

Although an icy saltwater bay seems an unlikely place to observe a western toad, the species has been occasionally observed and reported under such conditions. Anderson (2004) recounted observations in Taylor's (1983) thesis in which he observed several apparently glacial stream-dispersed western toads swimming in marine waters; whether or not the saltwater dispersal mechanism was accidental or intentional could not be determined. Western toads in GLBA reportedly exhibited a critical thermal minimum of 1.7°C (n=10) and withstood saltwater immersion for up to several (<9; n=18 toads) hours (Taylor 1983; but see also MacDonald 2010). Moreover, this species was reportedly observed on the Burroughs Glacier in 1978 (Taylor 1983). This hardy amphibian seems capable of limited dispersal within the marine environment to other suitable breeding habitats in the park, including areas that were recently glaciated (Anderson 2004). Their presence on various islands within Glacier Bay and elsewhere throughout Alaska confirms this (MacDonald 2010).

The western toad has become the focus of national and regional monitoring efforts in recent years. Descriptive, anecdotal accounts from local Gustavus residents recounted having to watch where one stepped to avoid squishing western toads underfoot because of the amount of toads on the ground, especially during the fall rainy period (Anderson 2004). Local anecdotal reports also suggest a decline in western toad population over the last several decades, which parallels precipitous declines in large portions of their range as well as declines in the overall global amphibian population (Carstensen et al. 2003). The southern and eastern Rocky Mountain Distinct Population Segments (DPSs) of western toads in the lower 48 states are currently under review for listing under the USFWS Endangered Species Act (USFWS 2015). A lack of regional knowledge regarding western toad distribution, population abundance, and habitat range also prompted recent monitoring efforts in SEAN (within Klondike Goldrush National Historic Park) (Wetherbee 2009). The IUCN Red List of Threatened Species lists the western toad as near threatened with a decreasing population trend; the significant rate of decline is likely due to a myriad of threats, including habitat loss, disease, predation, and other synergistic effects (Hammerson et al. 2004).

4.20.2. Measures

- Distribution (for all life stages)
- Abundance (for all life stages)

4.20.3. Reference Conditions/Values

Due to the lack of survey data, historical anecdotes of distribution and abundance of the western toad are the only sources of reference condition.

4.20.4. Data and Methods

Anderson (2004) conducted an amphibian inventory in Alaska’s National Parks from 2001 to 2003. Surveys were conducted in an opportunistic fashion due to constraints in budget as well as the sheer geographic scale of Alaska’s park areas; the inventory was conducted with other fieldwork that was already funded. NPS staff defined a protocol to be used in all the Alaska parks for this multi-year, finite project with the caveat that it may lead to further longer-term monitoring (Anderson 2004). Data collection was executed using field-durable flashcards produced specifically for assisting un-trained persons in the field identify amphibian species and life stage; the cards were accompanied by datasheets designed for recording detailed amphibian observations to be collected and compiled in a database specific to the project (Anderson 2004). The project also integrated searches of previously compiled databases such as NPSpecies and University of Alaska online Arctos Database for historic amphibian records (Anderson 2004).

Following completion of Anderson (2004), GLBA staff continued to update the newly created amphibian sighting database, and added confirmed western toad observations to the database whenever an opportunistic observation was made (NPS 2015). Toad observations were also reported whenever they were observed during a 2004-2005 investigation of black and brown bear activity in the park (Partridge et al. 2009). The updated NPS observation database is current with observations through 2010.

Pyare et al. (2007) investigated western toads in GLBA to determine breeding-site encounter rates in lower GLBA, breeding site microhabitat characteristics, and appropriate spatial scales for future western toad monitoring in GLBA (Pyare et al. 2007). The investigations were conducted using satellite, orthophoto imagery, and color infra-red "coastwalker" imagery to identify four areas that were abundant in wetland landcover at GLBA, and also to generate walking-survey routes to maximize the number and diversity of potential breeding sites (Pyare et al. 2007). The surveys were conducted at wetlands by visually searching shorelines and shallower margins for signs of breeding activity, such as egg masses and/or larvae (tadpoles) (Pyare et al. 2007). Two additional locations where a number of wetlands occurred were visited (Pyare et al. 2007). At all sites where breeding activity was confirmed, and a few where breeding activity was absent, 10 microhabitat variables were measured (Pyare et al. 2007).

The Alaska Gap Analysis Project (AGAP) compiled a database of Alaskan biota spatial distributions during a 3 year project conducted from 2009 to 2011 (AGAP 2013). The effort utilized an acquisition, synthesis, and organization of georeferenced species occurrence data with the purpose of ensuring a continued upkeep of storing ongoing collections of species occurrence data from future projects that can then become the tools for identifying trends in populations of target species in Alaska. Gotthardt and Pyare (2013), which represents the Alaska Amphibian Database, was an addition to the AGAP (2013) project that contained all known records of western toad observations within GLBA and throughout Alaska from 1958-2004.

4.20.5. Current Condition and Trend

<u>Distribution (for all life stages)</u>

Anderson (2004)

Anderson (2004) documented many variables during the 2001-2003 opportunistic inventory in GLBA, with occurrence/distribution (Table 61) and life stage (Table 62) being among several of the variables recorded (Table 61). A significant breeding area near the park boundary was observed during the inventory, with over 1,000 tadpoles observed in a single borrow pond (an area dug out for sand and gravel ditches to promote drainage in the area) near the southeast end of the Gustavus Airport, just outside of the park (Anderson 2004).

Table 61. Western toad inventory results from Anderson (2004) organized by habitat type.

Habitat Type	Number of Observations
Forested Area	5
Freshwater Pond/Lake	6
Manmade*	4
River	2
Saltwater/Estuarine	7
Stream	11
Wetland/Bog	3

* Indicates a habitat type that includes the borrow ponds and ditches in the Gustavus area.

Table 62. Number of tadpoles observed from 2001-2003, year in parenthesis, by habitat type and time of year during Anderson (2004).

Habitat Type	June	July
Manmade*	900 (2002)	200 (2002)
Freshwater Pond/Lake	40 (2003)	125 (2002) 30 (2003)

* Indicates a habitat type that includes the borrow ponds and ditches in the Gustavus area.

NPS Amphibian Sighting Database (NPS 2015)

The NPS Amphibian Sighting Database (NPS 2015) includes observations of western toads from 1991-2010; most of the data from 2001-2003 are from Anderson (2004), although some additional observations during this time frame that are not present in Anderson (2004) are included in the NPS database. Entries in the database also include toads observed during chytrid fungus testing in 2007 and 2008 (NPS 2008, Olsen 2014), toads sampled for DNA testing in 2009, and toads documented on the GLBA "Add-A-Toad Amphibian Observation Tracking Sheet" from 2006-2010.

Among observations with habitat type designated, the freshwater pond or lake habitat type had the highest number of western toad observations in GLBA from 1991-2010 (23 observations/17% of all observations) (Table 63). Most of the observations in the NPS Amphibian Sighting Database post-2005 did not report habitat type (66 observations), as these observations were largely from the Add-A-Toad amphibian observation tracking sheets. Other habitat types frequently observed included streams, and saltwater and estuarine areas.

Table 63. Distribution of western toads from 1991-2010 in the GLBA area. There is some overlap with Anderson (2004) in this, as NPS (2015) includes data from 2001-2003 which was collected by Anderson (2004).

Habitat Type	Number of Observations
Forested Area	9
Freshwater Pond/Lake	23
Other*	11
River	3
Saltwater/Estuarine	11
Stream	12
Wetland/Bog	5
Not Classified*	66
Total	**138**

* Includes observations at man-made habitats (e.g., borrow ponds in Gustavus, along man-made roads/trails).

Distribution of western toad life forms by habitat type can also be summarized using the NPS Amphibian Sighting Database (NPS 2015; Appendix K). The majority of adult observations were not attributed to a habitat type (66 observations), but of the 89 adult western toad observations from 1991-2010 that did have a habitat type identified the freshwater pond/lake, saltwater/estuarine, and stream habitats all had the same number of observations (eight; each accounting for nine percent of all adult observations). The diversity of habitats that adults were reported in emphasizes the adaptability and wide-ranging distribution of this species in the park. This adaptability and dispersal is perhaps best demonstrated by the fact that adults have been observed in both the upper East and West Arms of Glacier Bay, have been observed on islands, and have been observed in close proximity to glaciers (see also: Barteit et al. 2010, Moore et al. 2011, and Long and Prepas 2012). Figure 162 displays the distribution of the western toad (all life forms) using all available distribution data. Examples of these datasets includes: museum and voucher specimens (1958, 1967-1969, 1971, as well as undated specimens), citizen science data, Anderson (2004), AGAP (2013), Gotthardt and Pyare (2013), and NPS (2015).

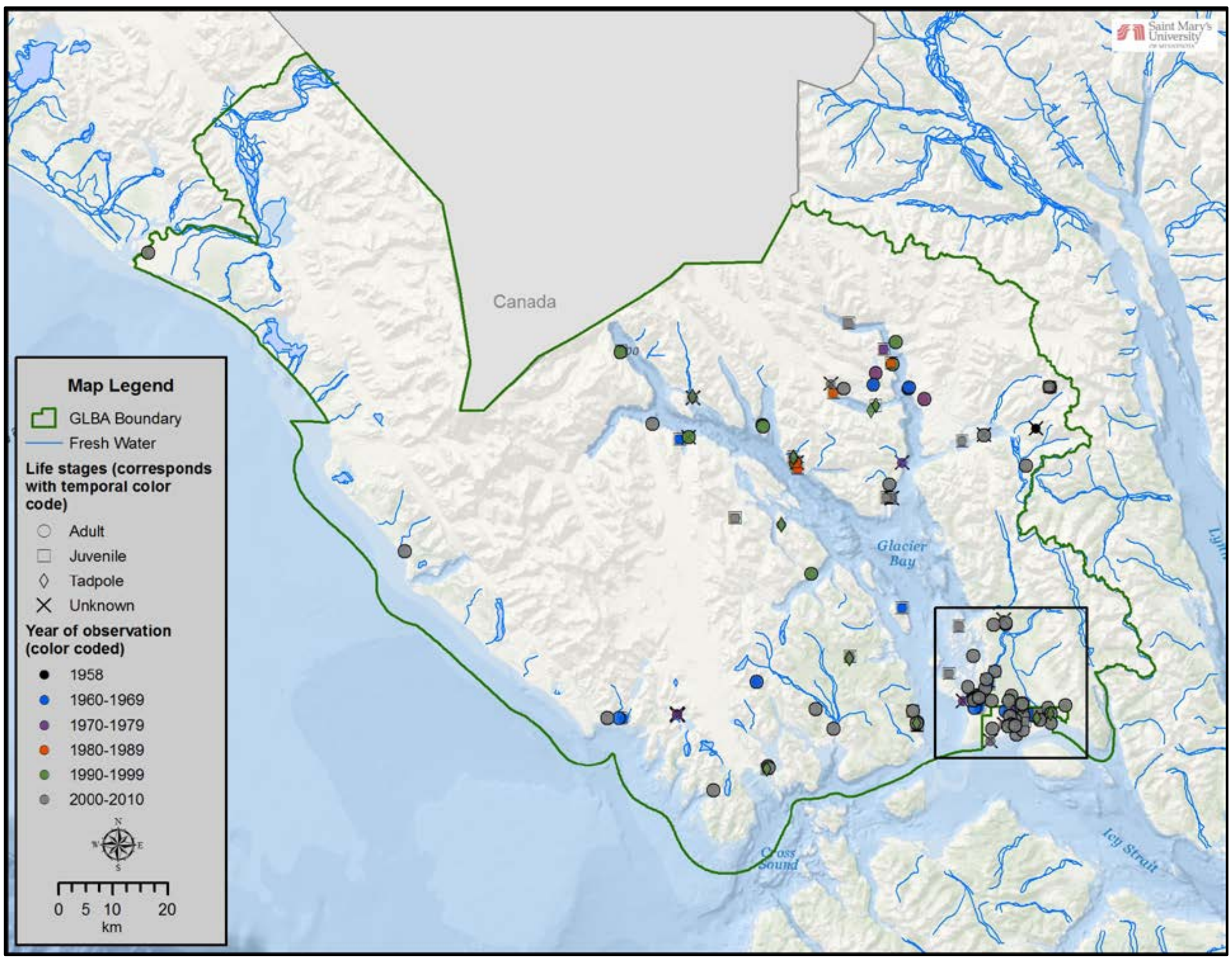


Figure 162. Western toads observed in GLBA and Gustavus, AK from 1958-2010. Observations are also separated by life stage when a stage was indicated in the observation record (NPS 2015). The black box around the Gustavus area is highlighted and expanded in Figure 163.

In NPS (2015), subadults were observed in highest numbers in the freshwater pond/lake habitat type (28 observations, 48% of all subadult observations), with no other habitat type contributing to more than seven percent of subadult observations (Appendix K). Outside of observations with no habitat type classified, tadpoles were observed in highest numbers in freshwater ponds and lakes (18% of all tadpole observations, or 68% of tadpole observations using only data with a habitat type identified) (Appendix K). These observations are typical of the species' reproductive behavior, as fresh water ponds or lakes are needed for successful egg laying. It is apparent based on NPS (2015) observations and AGAP (2013) and Gotthardt and Pyare (2013) data that there are several critical western toad spawning sites in and near GLBA. The Wilson Road gravel pit ponds in Gustavus, the south airport gravel pit pond in Gustavus, Vivid Lake in the West Arm of Glacier Bay, and the mainland east of Russell Island all appear to be important toad rearing and spawning sites, based on the distribution data for subadults and tadpoles (Figure 163; Fraley 2010). While limited, there are also reports of tadpoles in other areas of the park, as Carstenson et al. (2003) reported that there were observations of tadpoles in a small pond at 610 m (2,000 ft) on top of White Thunder Ridge in 1978, and AGAP (2013) shows tadpoles in the Bartlett Cove area.

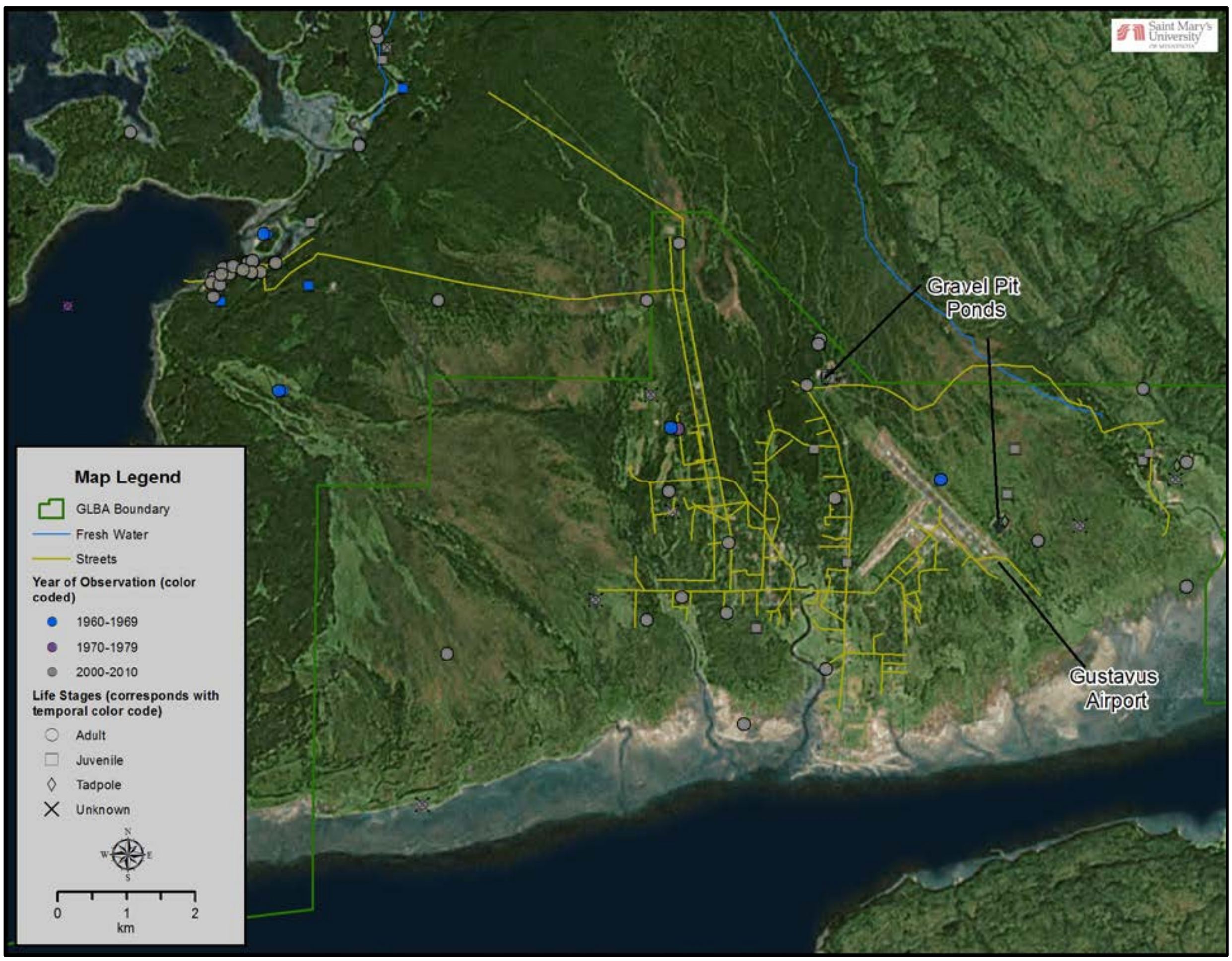


Figure 163. Western toads observed in the Gustavus, AK area from 1960-2010. Observations are also separated by life stage when a stage was indicated in the observation record (NPS 2015). Gravel pit ponds near the Gustavus airport are also indicated.

Pyare et al. (2007)

Pyare et al. (2007) identified four areas in the park where there were an abundance of high-density wetland clusters, these areas overlapped with wetland "hotspots" previously identified by Christensen et al. (2004). The methodology used for the selection of the potential breeding habitats is presented in Figure 164. During surveys at these 94 potential breeding spots, researchers identified only four ponds (<5%) that held either eggs or larval western toads. There were eight total sites (9%) that had general evidence of western toads but no breeding evidence was observed.

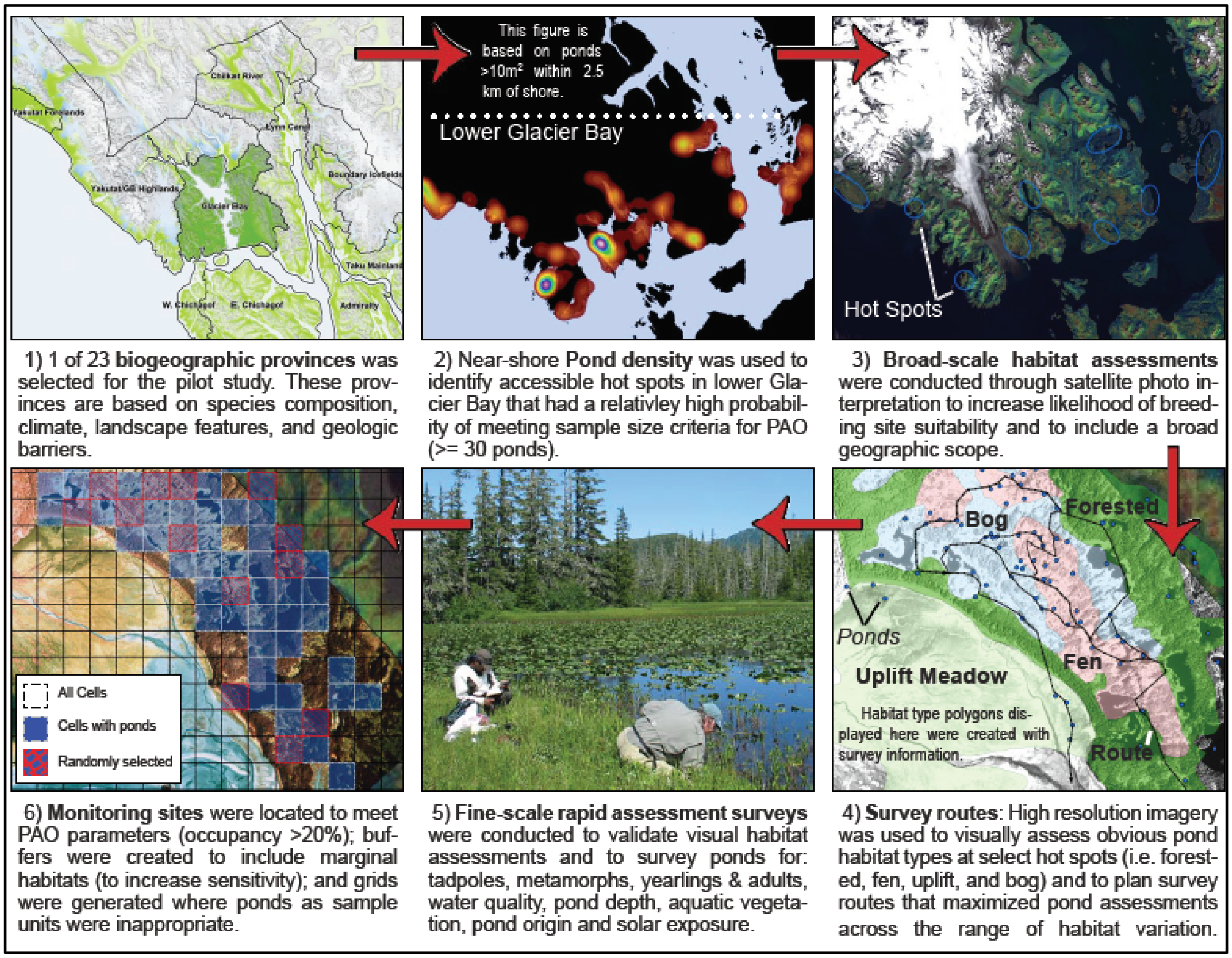


Figure 164. Western toad monitoring site selection process as is presented in Pyare et al. (2005).

The sites in GLBA where breeding was documented ranged in size from <1 m^2 (10.8 ft^2) to >9 km^2 (3.5 mi^2). Pyare et al. (2007) did not observe significant microhabitat differences between sites where breeding was detected and where breeding was not detected, although floating vegetation was significantly less at sites where eggs were present. Additionally, the majority of sites with breeding present (75%) or western toad evidence (88%) were associated with some sort of disturbance phenomena (e.g., uplift, glacial recession, anthropogenic modification) (Pyare et al. 2007).

Abundance (for all life stages)

Anderson (2004)

From 2001-2003, Anderson (2004) reported 38 observations which resulted in a count of 1,342 western toads (all life stages) in GLBA. In 2001 and 2002, the western toad was the most commonly observed amphibian species during the study efforts across all Alaskan national parks. Similarly, the species was the most abundant amphibian observed in GLBA during all 3 years of sampling (Anderson 2004), with 1,303 tadpoles, 20 sub-adults, 20 adults, and 1 ‘other life stage’ observed (Table 64). It is difficult to determine any trends in abundance using data from Anderson (2004) (or NPS 2015) as the reporting is entirely dependent on human observation. Because of this and the fact that there is unequal observation effort throughout the park, toad abundance and concentration/density trends from Anderson (2004) and the NPS database are unreliable (Fraley 2010).

Table 64. Life stage abundance estimates in GLBA from 2001-2003 as observed by Anderson (2004).

Life Stage	2001	2002	2003	Total
Adults	8	2	10	20
Subadult	11	2	5	18
Tadpole	0	1,233	70	1,303
Other	1	0	0	1
Total	**20**	**1,237**	**85**	**1,342**

NPS Amphibian Sighting Database (NPS 2015)

As was mentioned previously, the NPS Amphibian Sighting Database (NPS 2015) includes observations of western toads from 1991-2010, and most of the data from 2001-2003 are from Anderson (2004). Thus, there is some overlap in discussion between this subheading and Anderson (2004). There are reports of western toads annually in the park from 2000-2010, and sporadic observations that have been reported in 1991, 1993, and 1997. The number of adults observed in the park has ranged from one (1993) to 21 (2007), with total adult observations totaling 89 for the database’s duration (Figure 165, Appendix K).

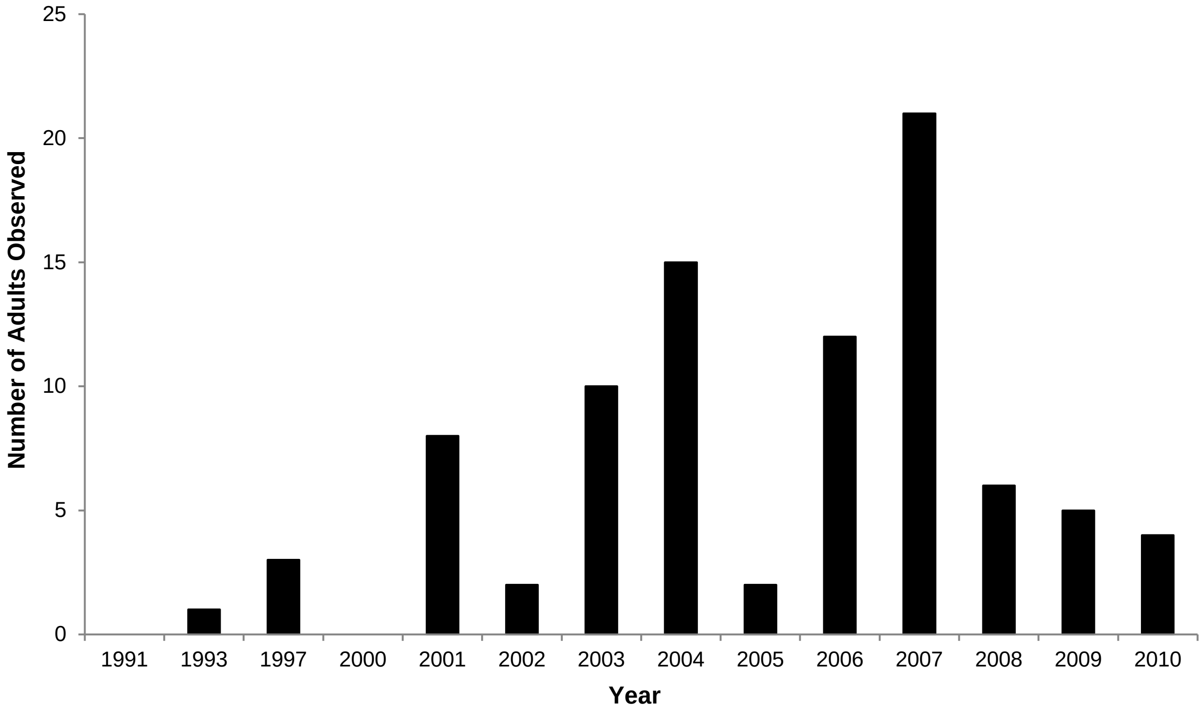


Figure 165. The number of adult western toads reported in the GLBA area from 1991-2010 (NPS 2015). It should be noted that observation effort was not equal among years.

Subadult western toads have been observed in lower numbers in GLBA when compared to adults between 1991 and 2010, and have only been reported in 7 years (2001-2004, 2006-2008). Abundance estimates have ranged from one (2008) to 24 (2004), and total subadult observations totaled 58 for the duration of the inventory. Tadpoles in GLBA and Gustavus were observed and reported in higher numbers, primarily due to the sheer number of eggs laid in a pond/lake (between 3,239 and 10,872 eggs are laid per clutch in Colorado western toads; Carey et al. 2005). Observed tadpole abundance estimates ranged from 100 (2000) to 14,900 (2004; Appendix K), although it needs to be emphasized that these numbers are estimates as the actual number of tadpoles is difficult to determine by an observer. Total tadpole observations from 1993-2010 were 17,959 (Appendix K). Similar to Figure 162 in the distribution measure, Figure 166 displays the abundance of the western toad (all life forms) using all available distribution data. Examples of these datasets includes: museum and voucher specimens (1967-1969, 1971, as well as undated specimens), citizen science data, Anderson (2004), AGAP (2013), Gotthardt and Pyare (2013), and NPS (2015). Figure 167 displays abundance data for several priority breeding ponds in the Gustavus region.

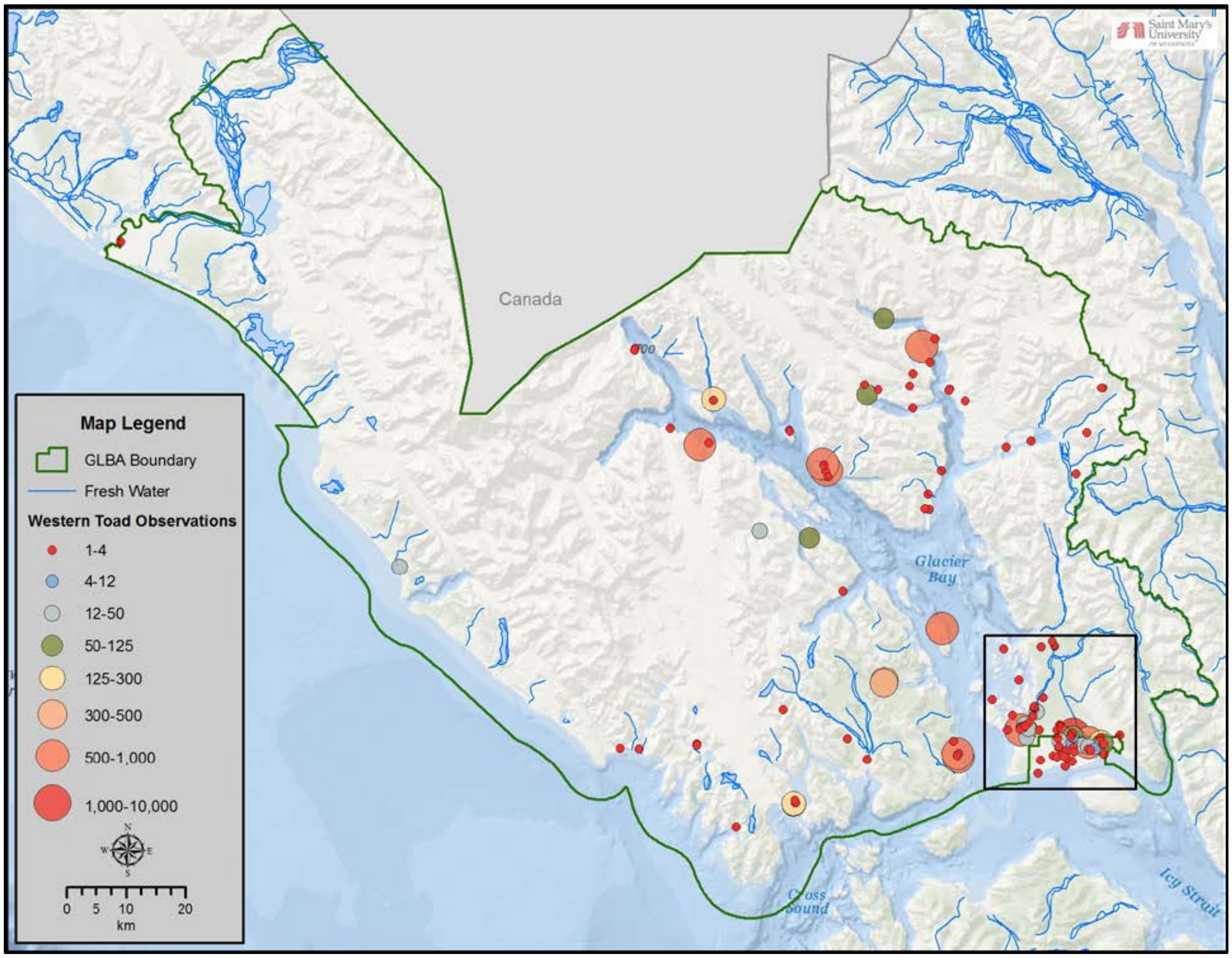


Figure 166. Number of western toad observations in the GLBA area from 1967-2010 (NPS 2015). The black rectangle around the Gustavus, AK area is highlighted and expanded in Figure 167. It should be noted that there is unequal observation area across the park and preserve.

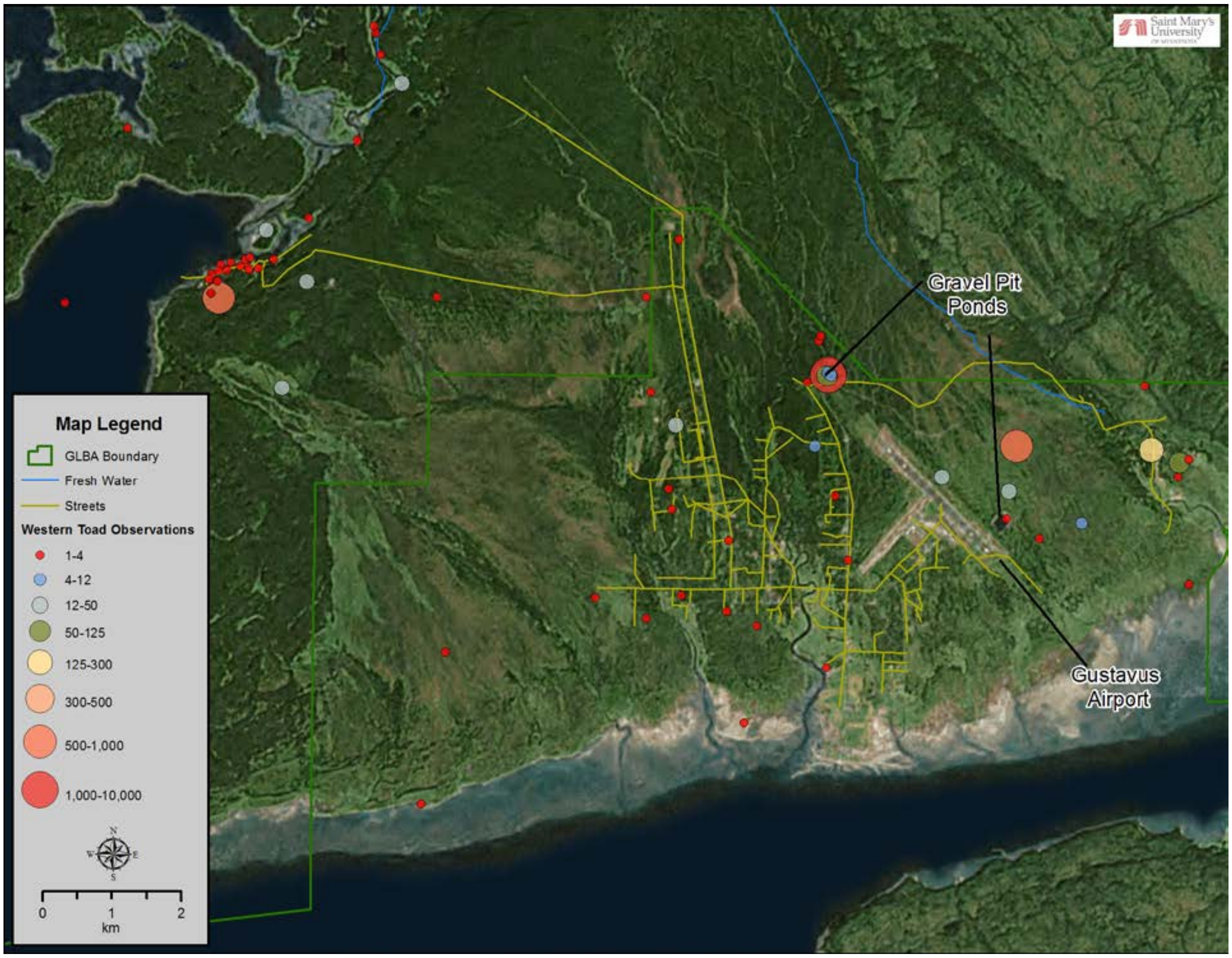


Figure 167. Number of western toad observations in the Gustavus, AK area from 1967-2010 (NPS 2015). It should be noted that there is unequal observation area across the park and preserve.

Pyare et al. (2007)

Pyare et al. (2007) did not report on abundance for the western toad in GLBA, largely due to the large geographic scale of the park. However, the report did provide anecdotal reports from local residents that indicated a perceived decline in western toad abundance in lower GLBA. This report echoes the local community's concern over a perceived decline in recent years as reported in Carstensen et al. (2003) and Anderson (2004).

Threats and Stressor Factors

GLBA staff identified several potential threats to the western toad population in the park, including chytrid fungus, climate change, increased presence of disease vectors (e.g., red leg disease, *Saprolegnia ferax* [fungal egg infection], other diseases), the influence of isostatic uplift on wetland habitat, and contaminants; the sources of these threats are both natural and anthropogenic. Currently, the western toad is listed as a "near threatened" species in the IUCN Red List of threatened species, which states that although the total population is unknown, the western toad has been in decline in portions of its range (Rocky Mountain populations, portions of Sierra Nevada California) since the 1970s (Hammerson et al. 2004).

Chytrid Fungus

The aquatic *Batrachochytrium dendrobatidis*, or chytrid fungus, causes chytridiomycosis, a lethal skin disease in amphibians that is linked to population declines in many areas throughout the world (Weldon et al. 2004, Hossack et al. 2009). Fungal zoospores parasitize the host's keratinized skin and mouthparts, causing a thickening of the skin (i.e., hyperplasia). This skin thickening inhibits moisture and electrolyte uptake, ultimately culminating in asystolic cardiac arrest (Voyles et al. 2009). Aquatic zoospores can persist in moist, cool temperatures, and direct contact with infected individuals or zoospores causes infection. Zoospores can be transported via anthropogenic mechanisms (e.g., infected gear and equipment) and may also be transported on birds or invertebrates (Johnson and Speare 2005). Chytridiomycosis is affecting hundreds of species around the world (Kriger 2006). In several locations within the state of Alaska, the fungus has been positively identified in amphibians (Figure 168; Olson 2014).

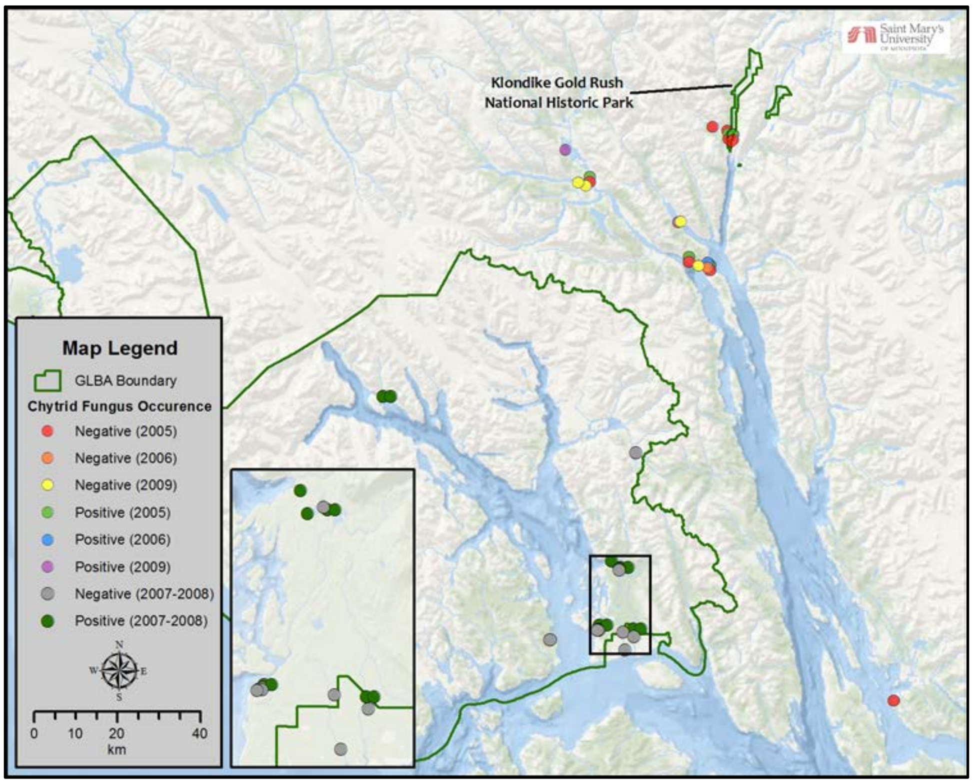


Figure 168. Chytrid sampling results in the GLBA region from 2005-2008. The Gustavus, Bartlett Cove, and Bartlett River area of GLBA has been expanded in an inset in the upper left corner of this figure. Field samples were collected by the NPS, and Pisces Molecular (Boulder, CO) conducted the DNA analysis of all samples.

The chytrid fungus was positively identified in the park during chytrid fungus testing conducted between 5 June 2007 and 18 August 2008. Five out of 18 western toads tested positive for chytrid fungus, and locations where positive results where documented in the park are shown in Figure 168 (NPS 2008). The areas with chytrid occurrence are closely aligned with western toad observations that are documented for the park shown in Figure 162. However, the fungus is likely more widespread in GLBA and has not been broadly detected due to limited sampling efforts (Soiseth, written communication, 17 August 2015). Of particular interest is the fact that the chytrid fungus was documented in a habitat that was recently deglaciated (Russel Island, NW reach of the West Arm), which may support a dispersal strategy of the fungus that is aerial in nature (i.e., transport of fungus through birds or insects). This is a topic of particular concern for managers, and warrants future investigation and study to determine the total levels of occurrence in the park, and potential mechanisms of dispersal.

Climate change

Kiesecker et al. (2001) linked amphibian pathogen outbreaks to climate-induced changes in UV-B exposure. The decreased water depths from precipitation pattern shifts were exposing egg masses to the harmful UV-B rays; this left the eggs susceptible to disease (Kiesecker et al. 2001). *Saprolegina farax* infection has caused large number of western toad egg mortalities in shallow lakes and ponds of western North America; the egg masses turn white and die within a few days of deposition (Pounds 2001). Considering GLBA has concerns with wetland and pond desiccation, that may also be exacerbated by changing regional climate and isostatic uplift, this may become a threat to the western toads currently inhabiting the park.

Climate change has the potential to affect amphibians in many capacities, and the causes for population declines in many amphibian species, or even individual populations, may be different depending on the region or ecosystem they inhabit (Beever and Belant 2011). Synergistic interactions between different variables and factors in the environment can create unique situations in different environments. According to Ovaska (1997), changes in temperature and moisture patterns driven by climate change could cause physiological stress in amphibians and would affect their mobility patterns, timing and duration of breeding, growth, hibernation, and aestivation patterns (see also Donnelly and Crump 1998, and Blaustein et al. 2001).

Increased presence of vectors (disease)

An additional threat to western toads is the pathogenic oomycete *Saprolegnia ferax*, which is a fungus that can infect all stages of amphibians (Beever and Belant 2011), although mortality is often most significant in developing embryos (Blaustein et al. 1994, Romansic et al. 2007, 2009). Western toads that develop in shallower water are exposed to higher levels of UV-B radiation, which can increase levels of mortality caused by the Saprolgenia fungus (Kiesecker et al. 2001).

Isostatic Uplift

Isostatic uplift occurs when glaciers recede; the resulting uplift is a rebound of the continental crust, which is less dense than the underlying material in the earth’s mantle, after the immense weight of the glacier is removed. The effect isostatic uplift has on the land that is most concerning for the western toad is the drying of wetland areas which reduces the amount and quality of available aquatic habitats required by western toads to complete their life cycle. According to Larson et al. (2005) GLBA has one of the fastest uplift rates globally, with uplift rates reported at 30 mm/year (Larson et al. 2005).

Data Needs/Gaps

The western toad has been infrequently studied in the GLBA region, with the most intensive survey occurring over a decade ago. The NPS Amphibian Sighting Database is a useful tool to monitor infrequent/anecdotal toad observations, but it is difficult to determine population trends or health using this data alone as there is really no survey or sampling protocol. Data related to abundance and distribution are needed if an assessment of current condition is to take place. A long term monitoring program would be ideal, but the large geographic scale of the park would make monitoring park-wide a difficult undertaking. Monitoring spaced out at regular intervals at a select few sampling sites (e.g., breeding ponds, other freshwater areas) with a strict sampling protocol would help managers to obtain a better population estimate and could lead to a more accurate assessment of the current health of the western toad population in the GLBA region.

Additional data needs in GLBA include investigating the effect of climate change on amphibians in the park. Of particular importance are changing precipitation patterns, as Kiesecker et al. (2001) indicated that toads that breed in areas of shallow water (often due to low snow pack) are more susceptible to *Saprolegnia ferax* infection. Potential temperature shifts are also of concern, and the NPS has begun the process of installing a number of weather stations throughout the park and preserve in order to monitor air temperature, precipitation, and other parameters. Combining these data with stream temperature and other water quality parameters already being collected will help park managers to better understand the changing western toad habitats and may help to identify causes of population trends or shifts.

Overall Condition

Abundance for All Life Stages

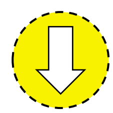
The project team defined the *Significance Level* for abundance as a 3. The data related to western toad abundance in the park area are relatively limited in scope (i.e., entirely opportunistic, relatively limited coverage area) and dated (most recent study was in 2003). The overall consensus among park and amphibian experts and reports of anecdotal accounts of previous western toad abundance suggests that they are significantly declining in abundance. This merits the investment of monitoring efforts to assess their trends of abundance in GLBA (Anderson 2004, Hammerson et al. 2004, Pyare et al. 2007). Additionally, there are current threats to the species in the park that will, or are already, directly affecting abundance of western toads. Examples of these threats include: the presence of *Batrachochytrium dendrobatidis* in the park, climate change, the loss of wetland habitat due to isostatic uplift, and human development in the Gustavus area (which would affect abundance if the borrow ponds are sources and the park represents a sink). Due to these factors, a *Condition Level* of 2 was assigned to this measure.

Distribution for All Life Stages

The project team defined the *Significance Level* for distribution as a 3. The distribution of the various life forms of western toads has been relatively far-reaching considering the large geographic area and range of habitats of GLBA. Western toads in the GLBA area have been found in borrow ponds, streams, ponds, rivers, saltwater and estuarine habitats, and bogs in basically all areas of the park (Figure 1). The western toad is very adaptive and wide-ranging in GLBA (Taylor 1983), but more research is needed to fully understand the distribution patterns of all of the life forms in the park. The wide range of habitats and area that the species has been documented in the park is enough to assign this measure a *Condition Level* of 1, however this needs to be interpreted cautiously as no formal study of distribution for the life stages of the western toad has occurred in the park.

Weighted Condition Score

The *Weighted Condition Score* for western toads in GLBA was determined to be 0.50, indicating that this component is of moderate concern. A declining trend was assigned to the graphic, due in part to the anecdotal accounts of perceived declines in abundance in the Gustavus area, and the current threats that exist in the park. A low confidence border was assigned to the graphic, as this assessment relies heavily on data that are dated or are opportunistically collected. Additionally, accounts of population declines in the Gustavus area are anecdotal, and may only represent *perceived* declines in abundance.

Western Toads			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.50**
Abundance (all life stages)	3	2	
Distribution (all life stages)	3	1	

4.20.6. Sources of Expertise

- Chad Soiseth, GLBA Fisheries Biologist
- Tania Lewis, GLBA Wildlife Biologist

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4.21. Air Quality

4.21.1. Description

Air pollution can significantly affect natural resources and their associated ecological processes. Consequently, air quality in parks and wilderness areas is protected and regulated through the 1916 Organic Act, and the Clean Air Act of 1977 (CAA) and the CAA's subsequent amendments. The CAA defines two distinct categories of protection for natural areas, Class I and Class II airsheds. Class I airsheds receive the highest level of air quality protection as offered through the CAA; only a small amount of additional air pollution is permitted in the airshed above baseline levels (EPA 2013a). For Class II airsheds, the increment ceilings for additional air pollution above baseline levels are slightly greater than for Class I areas and allow for moderate development (EPA 2013a). GLBA is designated as a Class II airshed; however, the park meets the criteria of a Class I airshed.

The air quality of southeast Alaska is generally believed to be among the most pristine in the world due to its low population densities, lack of large-scale industrial development, and vast wildland areas (Photo 53) (Dillman et al. 2007, Schirokauer et al. 2014). However, this high air quality is currently threatened by global industrial pollution and local sources, such as cruise ships. The potential impacts of air pollution at GLBA include diminished visibility for visitors, degradation of forest health, changes to ecological structure (e.g., replacement of pollution-sensitive species with pollution-tolerant species), alteration of soil and aquatic chemistry, and bioaccumulation of ecological contaminants (Furbish et al. 2000). Air quality is of high value to SEAN park managers, and contaminants from global sources are a growing concern due to their possible impacts on a wide variety of SEAN Vital Signs (Moynahan et al. 2008).

Photo 53. John Hopkins Inlet on a "Fairweather Day" (NPS photo).

4.21.2. Measures

- Lichen sampling (nitrogen, sulfur, nitrite concentrations)
- Atmospheric deposition of sulfur and nitrogen

- Mercury deposition
- Particulate matter (PM)
- Visibility

Lichen Sampling

Lichens are often used to monitor air quality since they absorb nutrients directly from their surroundings (Furbish et al. 2000, Blett et al. 2003). Relatively low levels of sulfur, nitrogen, and some heavy metals adversely affect many lichen species, causing changes in growth and reproduction, physiological processes (e.g., photosynthesis), and morphological appearance (Blett et al. 2003, USFS 2007). Some pollutants may even lead to the elimination of particularly sensitive species, leading to an alteration in lichen community composition (Blett et al. 2003, Schirokauer et al. 2014). Studies have found that sensitivity to air pollution among lichens varies by growth form. Fruticose or shrubby lichens are generally most sensitive, foliose or leafy lichens are moderately sensitive, and crustose or flat lichens are least sensitive (Blett et al. 2003). Epiphytic lichens from the genera *Alectoria*, *Bryoria*, *Ramalina*, *Lobaria*, *Nephroma*, and *Usnea* are thought to be some of the most sensitive (Blett et al. 2003).

Lichens are considered an integral part of the temperate rain forest ecosystem of southeastern Alaska (LaBounty 2005). They play a key role in nutrient cycles and are valuable sources of forage, shelter, and nesting materials for a variety of wildlife species (Blett et al. 2003). Lichen tissue sampling and community composition surveys were conducted to help inform the development of the SEAN air quality monitoring protocol (Schirokauer et al. 2014).

Atmospheric Deposition of Sulfur and Nitrogen

Sulfur and nitrogen oxides are emitted into the atmosphere primarily through the burning of fossil fuels, industrial processes, and agricultural activities (EPA 2012). While in the atmosphere, these emissions form compounds that may be transported long distances and settle out of the atmosphere in the form of pollutants such as particulate matter (e.g., sulfates, nitrates, ammonium) or gases (e.g., nitrogen dioxide, sulfur dioxide, nitric acid, ammonia) (NPS 2008, EPA 2012). Atmospheric deposition can be in wet (i.e., pollutants dissolved in atmospheric moisture and deposited in rain, snow, low clouds, or fog) or dry (i.e., particles or gases that settle on dry surfaces as with windblown dusts) form (EPA 2012). Deposition of sulfur and nitrogen can have significant effects on ecosystems including acidification of water and soils, excess fertilization or increased eutrophication, changes in the chemical and physical characteristics of water and soils, and accumulation of toxins in soils, water and vegetation (NPS 2008, reviewed in Sullivan et al. 2011a and 2011b).

Mercury

Sources of atmospheric mercury include fuel combustion and evaporation (especially coal-fired power plants), waste disposal, mining, industrial sources, and natural sources such as volcanoes and evaporation from mercury-enriched soils, wetlands, and oceans (EPA 2008). Mercury deposited into rivers, lakes, and oceans can accumulate in various aquatic species, resulting in exposure to wildlife and humans (EPA 2008). Elevated or unnatural levels of mercury (e.g., methylmercury) may affect wildlife in a number of ways, including slower growth rate and development, abnormal behaviors, reduced fertility, and mortality (EPA 2014b). High levels of mercury or methylmercury in humans have led to damage to the nervous system and reproductive organs as well as adversely affecting motor skills and the sensory system (EPA 1997).

Particulate Matter (PM) and Visibility

Particulate matter (PM) is a complex mixture of extremely small particles and liquid droplets suspended in the atmosphere. Fine particles ($PM_{2.5}$) are those smaller than 2.5 micrometers in diameter (EPA 2009). Particulate matter largely consists of acids (such as nitrates and sulfates), organic chemicals, metals, and soil or dust particles (EPA 2009, EPA 2013b). Fine particles are a major cause of reduced visibility (haze) in many national parks and wildernesses (EPA 2012). $PM_{2.5}$ can be directly emitted from sources such as forest fires, or they can form when gases emitted from power plants, industries, and/or vehicles react with air (EPA 2009, EPA 2012). Particulate matter either absorbs or scatters light. As a result, the clarity, color, and distance that humans can see decreases. Water in the atmosphere causes particles like nitrates and sulfates to expand, increasing their light-scattering efficiency (EPA 2012). $PM_{2.5}$ is also a concern for human health as these particles can easily pass through the throat and nose and enter the lungs (EPA 2009, EPA 2012, EPA 2013b). Short-term exposure to these particles can cause shortness of breath, fatigue, and lung irritation (EPA 2009, EPA 2012, EPA 2013b).

4.21.3. Reference Conditions/Values

The NPS Air Resources Division (ARD) developed an approach for rating air quality conditions in national parks, based on the current National Ambient Air Quality Standards (NAAQS), ecosystem thresholds, and visibility improvement goals (NPS 2011a). Table 65 shows the air quality index values used to assess air quality in national parks. Assessment of current condition of nitrogen and sulfur atmospheric deposition is based on wet (rain and snow) deposition. Visibility conditions are assessed in terms of a Haze Index, a measure of visibility (termed deciviews) that is derived from calculated light extinction and represents the minimal perceptible change in visibility to the human eye (NPS 2011a). The "good condition" metrics may be considered the reference condition for GLBA (Table 65).

Table 65. National Park Service Air Resources Division air quality index values (NPS 2011a).

Condition	Wet deposition of N or S (kg/ha/yr)	Visibility (dv*)
Significant Concern	>3	>8
Moderate Condition	1-3	2-8
Good Condition	<1	<2

* A unit of visibility proportional to the logarithm of the atmospheric extinction; one deciview represents the minimal perceptible change in visibility to the human eye.

NPS (2011b) provided the current NAAQS for cumulative impacts to human health for particulate matter, sulfur dioxide, and nitrogen dioxide. Impact levels ranged from negligible to major. The negligible level represents the range of concentrations that are considered to be policy relevant background (PRB) concentrations, which were set by the EPA (NPS 2011b). The minor level represents a range from the PRB to 50% of the NAAQS, while the moderate level represents a range from 51-79% of the NAAQS. The major impact level represents a range from 80-100% of the NAAQS for the respective pollutants (NPS 2011b; Table 66).

Table 66. Concentrations of pollutants (particulate matter, sulfur dioxide, nitrogen dioxide) categorized by impact level (negligible, minor, moderate, major) set for human heath (NPS 2011b).

Impact Level	Particulate Matter (PM_{10}) (µg/m^3)	Particulate Matter ($PM_{2.5}$) (µg/m^3)	1-Hour Sulfur Dioxide (ppm [µg/m^3])	1-Hour Nitrogen Dioxide (ppm)
Major	120–150	29–35	0.06–0.075 (169–212)	0.079–0.1
Moderate	78–119	21–28	0.035–0.059 (99–166)	0.05–0.079
Minor	12–77	6–20	0.002–0.034 (6–96)	0.002–0.049
Negligible	0–11	0–5	0–0.001 (0–3)	0–0.001

For lichen sampling, the reference conditions are thresholds determined by Dillman et al. (2007) through extensive lichen sampling and elemental analysis in southeastern Alaska's Tongass National Forest. These thresholds are based on lichen tissue analysis from 88 plots, 51 of which are in wilderness areas considered "pristine" (Dillman et al. 2007). Thresholds are species-specific and are available for the three species sampled in GLBA (Table 67).

Table 67. Lichen tissue thresholds in percent (%) (sulfur and nitrogen) or parts per million (ppm) for some of the sampled elements, as determined by Dillman et al. (2007).

Species	Sulfur (%)	Nitrogen (%)	Potassium (ppm)	Lithium (ppm)	Silicon (ppm)	Phosphorus (ppm)	Lead (ppm)
Alectoria sarmentosa	0.06	0.56	2,413.25	0.40	134.75	913.75	5.00
Hypogymnia enteromorpha	0.09	0.88	3,284.34	0.71	563.82	1,597.23	10.13
Platismatia glauca	0.08	0.80	2,523.88	0.60	635.83	1,115.00	3.52

4.21.4. Data and Methods

Monitoring in the Park

Schirokauer et al. (2014) initiated an airborne contaminant monitoring survey throughout southeast Alaska, including GLBA, in 2008-2009. The survey focuses on the effects of contaminants on lichen tissue. There were over 286 sites across southeastern Alaska used for lichen community and tissue sample collection plots, primarily in the Tongass National Forest, with three sites in GLBA (Bartlett Cove, Blue Mouse Cove, and Tarr Inlet; Figure 169). Sampling methods included lichen collection focusing on three species used as bioindicators (*Hypogymnia enteromorpha*, *H. inactiva*, and *Platismatia glauca*), and installing lichen community composition plots. Elemental analysis of epiphytic lichens was used to describe air pollution exposure. It should be noted that lichens suitable for analysis were not present in the Tarr Inlet location (Schirokauer and Geiser 2011). Ion exchange resin tube samplers were also used to measure wet deposition of nitrogen as ammonium (NH_4^+) and nitrate (NO_3^-) and sulfur as sulfate (SO_4^{2-}). (Schirokauer et al. 2014).

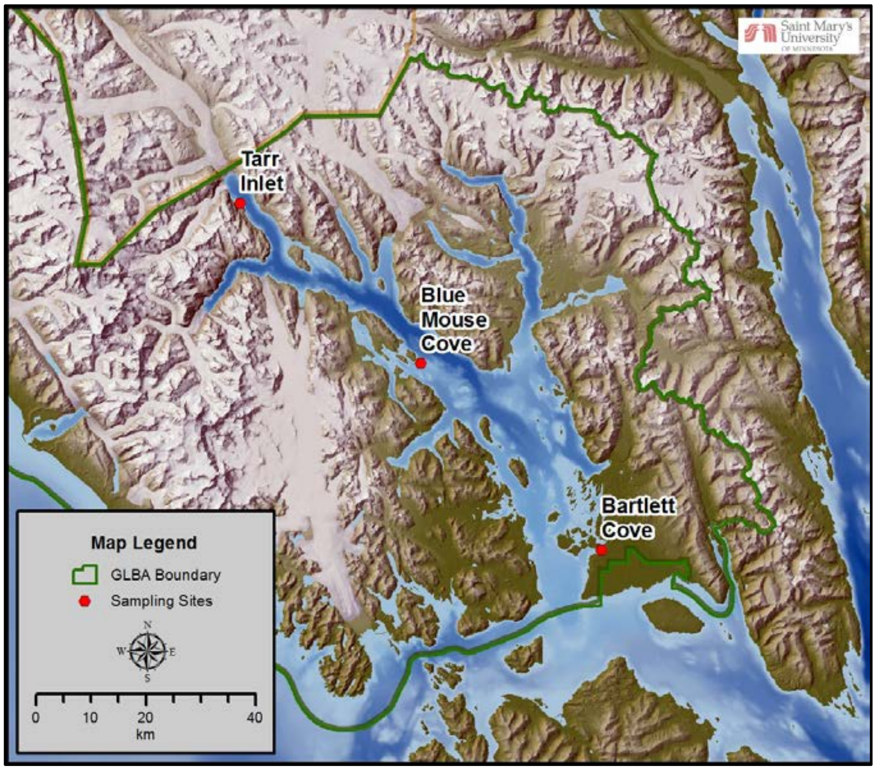

Figure 169. The three lichen sampling sites located in GLBA (Schirokauer et al. 2014).

The Mercury Deposition Network (MDN) (MDN 2014) compiled annual mercury deposition concentrations for GLBA and four other sites in Alaska (Figure 170). Annual data from the GLBA site were only collected between 2010 and 2013. The MDN used automated precipitation chemistry collectors and gages to measure mercury concentrations in precipitation. The collector is equipped with a glass funnel, connecting tube, bottle for collecting samples, and an insulated enclosure to house the sampling train. Samples are collected on Tuesday mornings or daily, depending on the presence of precipitation (MDN 2014).

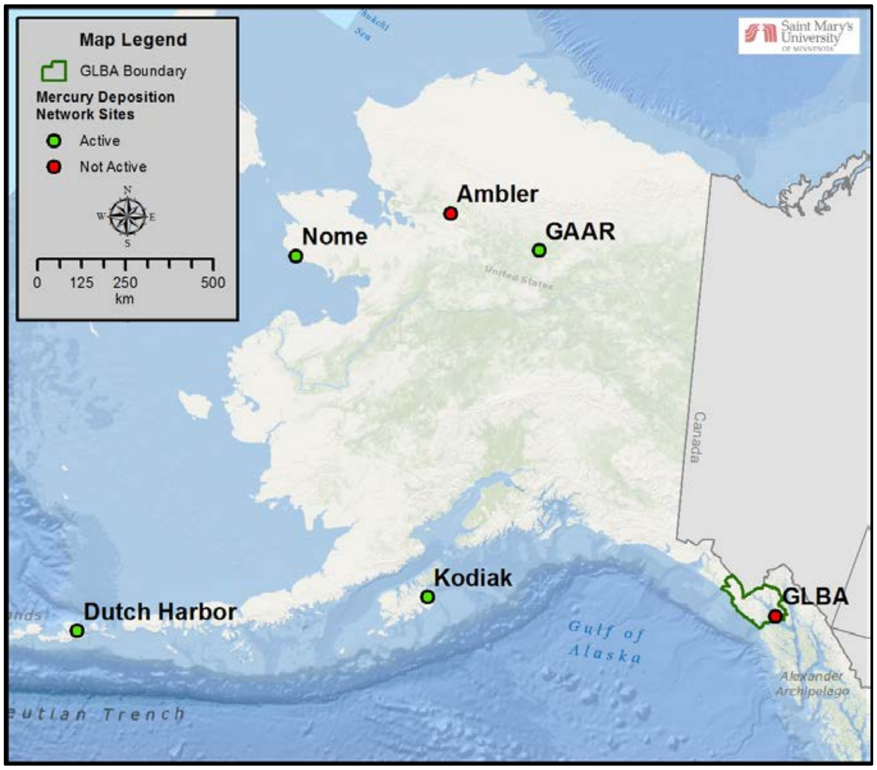

Figure 170. Mercury Deposition Network sampling sites in Alaska in relation to GLBA (MDN 2014).

Pirhalla et al. (2014) conducted a study on particulate matter from cruise ship emissions in GLBA in 2008. The purpose of this study was to determine the effects of cruise ships on the park's air quality. Weather Research and Forecasting Model (WRF) simulations and chemistry were used to determine the particulate matter (<10µm; PM_{10}) emitted. There were several buoys and land-based sites in southeast Alaska that were used in the WRF simulations (Figure 171). One buoy and one land-based site were located in Gustavus near GLBA.

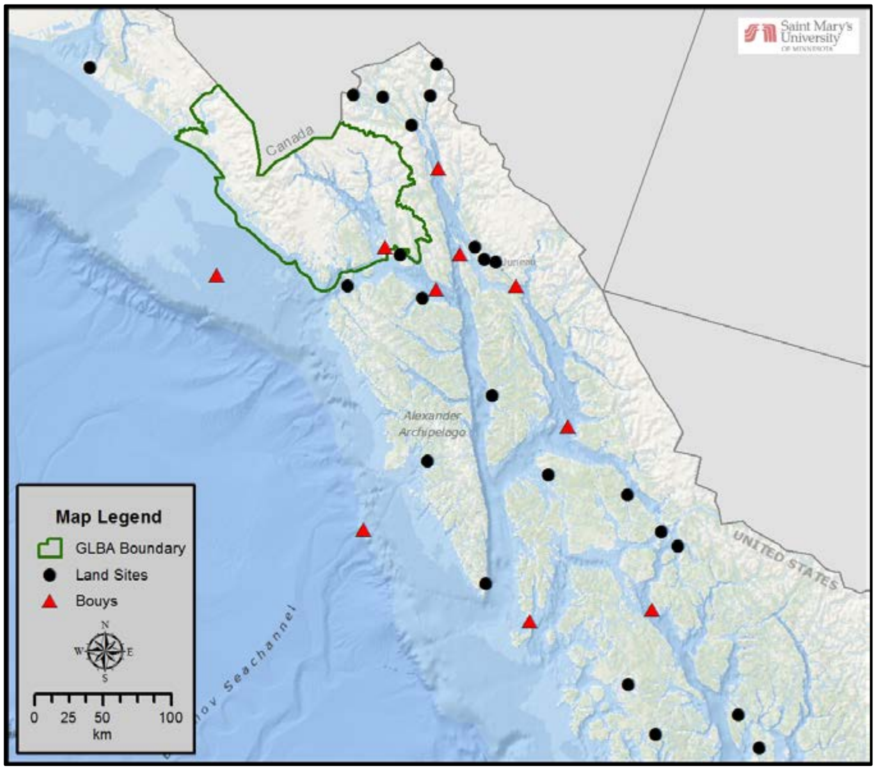

Figure 171. Buoys and land based sites used in the Pirhalla et al. (2014) particulate matter study.

Special Air Quality Studies

Sullivan et al. (2011a) assessed the relative sensitivity of national parks to the potential effects of acidification caused by atmospheric deposition from nitrogen and sulfur compounds. The relative risk for each park was assessed by examining three variables: the level of exposure to emissions and deposition of nitrogen and sulfur; inherent sensitivity of park ecosystems to acidifying compounds (N and/or S) from deposition; and level of mandated park protection against air pollution degradation (i.e., wilderness and Class I). The outcome was an overall risk assessment that estimates the relative risk of acidification impacts to park resources from atmospheric deposition of nitrogen and sulfur (Sullivan et al. 2011a). Using the same approach, Sullivan et al. (2011b) assessed the sensitivity of national parks to the effects of nutrient enrichment by atmospheric deposition of nitrogen. The outcome was an overall risk assessment that estimates the relative risk to park resources of nutrient enrichment from increased nitrogen deposition.

Other Air Quality Data Resources

The National Atmospheric Deposition Program–National Trends Network (NADP) database provided annual average summary data for nitrogen and sulfur concentration and deposition in the southeast Alaska region. Monitoring site AK02, located in Juneau, Alaska, is approximately 61 km (38 mi) southwest of GLBA (Figure 172). Although no longer operational, this site provided deposition data for the region from 2004 through 2013. It should be noted that air quality data were not retrieved in 2009. Results from monitors located within 16 km (10 mi) from parks are generally considered to be representative of park conditions (Ellen Porter, NPS Air Resources Division Air

Quality Specialist, phone communication, 25 October 2012). Data recorded at monitors beyond this distance from parks may represent regional conditions, but may not be representative of actual park conditions.

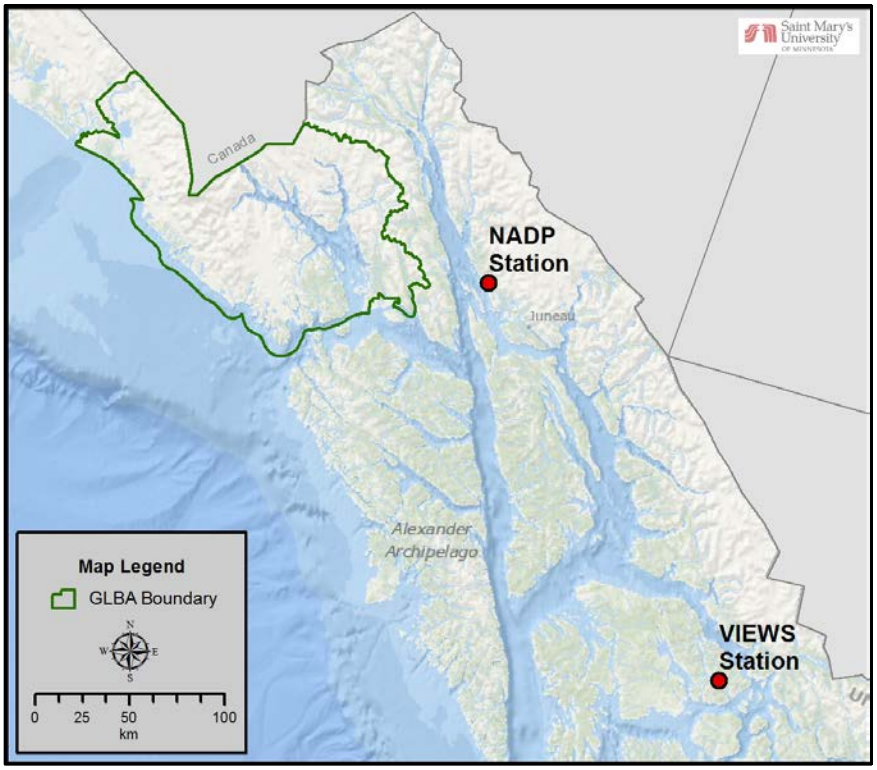

Figure 172. Location of NADP and VIEWS stations in relation to GLBA.

The Interagency Monitoring of Protected Visual Environments Program (IMPROVE) actively monitors visibility conditions in Class I airsheds across the U.S. The IMPROVE monitoring site nearest to GLBA is located southeast of Juneau, Alaska, in Petersburg (monitor ID PETE1), approximately 500 km (311 mi) from the park. The Visibility Information Exchange Web System (VIEWS) database provides average annual visibility monitoring data (in deciviews) and trend graphics for the PETE1 monitoring site from 2005-2008 for the clearest and haziest 20% of days for the region (VIEWS 2010). Due to its distance from GLBA, data from this monitor are likely representative of conditions in the area overall, but may not accurately represent conditions at the park specifically, thus data are interpreted with care.

The EPA Air Trends database provides annual average summary data for sulfur dioxide near GLBA. Monitoring site number 02-220-0008 is operated by Sitka County and is located in Sitka, Alaska, approximately 285 km (177 mi) south of GLBA (EPA 2014a). The distance of this monitor from GLBA and the variations among terrain make it difficult to extrapolate data accurately; thus, data from this monitoring station were not considered in this assessment. It should be noted that sulfur dioxide data from this station were only recorded in 1991 (230 ppb) and 1992 (309 ppb).

4.21.5. Current Condition and Trend

Lichen Sampling

Schirokauer et al. (2014) documented three lichen species suitable for elemental analysis at two of the three study sites in GLBA. Three lichen species were collected at Bartlett Cove and two species were collected at Blue Mouse Cove (Table 68). The third sampling site (Tarr Inlet) did not have any lichens suitable for elemental analysis during this study period.

Table 68. Lichen species collected at two of the three sampling sites within GLBA in 2008 and 2009 (Schirokauer and Geiser 2011).

Species	Year	Number of Samples
Bartlett Cove		
Hypogymnia enteromorpha	2008	3
Platismatia glauca	2008	2
Platismatia glauca	2009	2
Alectoria sarmentosa	2009	2
Blue Mouse Cove		
Hypogymnia enteromorpha	2008	4
Hypogymnia enteromorpha	2009	2
Platismatia glauca	2008	2
Platismatia glauca	2009	2

Schirokauer et al. (2014) reported elemental concentrations from these 2008-2009 lichen samples. Potassium was the only element collected from *Platismatia glauca* at the Bartlett Cove site that had concentrations exceeding the threshold established by Dillman et al. (2007). Several elements collected from *P. glauca* and *Hypogymnia enteromorpha* samples at the Blue Mouse Cove site exceeded the Dillman et al. (2007) thresholds (Table 67). Those elements include potassium, lithium, nitrogen, and silicon (Schirokauer et al. 2014). Graphs showing elemental concentrations that exceeded thresholds at GLBA are included in Appendix L.

Atmospheric Deposition of Sulfur and Nitrogen

Schirokauer et al. (2014) measured wet deposition (e.g., from precipitation) of nitrogen as ammonium (NH_4^+) and nitrate (NO_3^-) and sulfur as sulfate (SO_4^{2-}). Within GLBA, sampling was conducted in an open area at Tarr Inlet, and below tree canopy (canopy throughfall) at Bartlett Cove, Blue Mouse Cove, and Tarr Inlet (Figure 169). All nitrogen measurements from these sites fall within the "good condition" range identified by the NPS ARD (<1 kg/ha) (Table 69). Sulfur deposition was in the good condition range at the Tarr Inlet open site, but in the moderate condition range at all throughfall sites in both years (Schirokauer et al. 2014).

Table 69. Annual atmospheric wet deposition data in kg/ha for three sites in GLBA (Schirokauer et al. 2014).

Site	N as NH_4		N as NO_3		S as SO_4	
	2008	2009	2008	2009	2008	2009
Open site						
Tarr Inlet	0.13	0.03	0.25	0.04	0.82	0.12
Canopy Throughfall Sites						
Bartlett Cove	0.36	0.19	0.02	0.02	1.03	1.89
Blue Mouse Cove	0.30	0.03	0.00	0.17	1.12	2.05
Tarr Inlet	0.49	0.18	0.05	0.05	2.71	1.56

Relative risk of acidification and nutrient enrichment of ecosystems was assessed by examining exposure to nitrogen deposition and acidification, inherent sensitivity of park ecosystems, and mandates for park protection. Sullivan et al. (2011a) ranked GLBA as having low exposure to acidifying (nitrogen and sulfur) pollutants, high ecosystem sensitivity to acidification, and high park protection due to its Class II airshed status. The ranking of overall risk from acidification due to acid deposition was moderate relative to other parks (Sullivan et al. 2011a). In a separate examination, Sullivan et al. (2011b) used the same approach to assess the sensitivity of national parks to nutrient enrichment effects from atmospheric nitrogen deposition relative to other parks. GLBA was ranked as having very low risk for nitrogen pollutant exposure, low ecosystem sensitivity, and very high park protection mandates (Class II airshed). The ranking of overall risk of effects from nutrient enrichment from atmospheric nitrogen deposition was low relative to other parks (Sullivan et al. 2011b).

<u>Mercury Deposition</u>

There are no current or long-term monitoring data for mercury deposition or atmospheric concentration in GLBA. There was previously a monitoring station located in the park; however, the station is now inactive and data were only collected in 2011 and 2012. The MDN (2014) station recorded an annual average wet deposition of mercury in GLBA of 2.9 $\mu g/m^2$ in 2011 and 3.1 $\mu g/m^2$ in 2012 (MDN 2014). Annual average mercury depositions may be considered of good condition if concentrations are <4 $\mu g/m^2$ (NADP 2011). According to some sources (Landers et al. 2008, Jaegle 2010), mercury deposition in Alaska is considered to be low. Jaegle (2010) mentioned that Alaska parks had significantly lower levels of mercury in lichen samples than the conterminous U.S. Mercury levels in Alaska parks ranged from 10-20 ng Hg/g while the lower 48 states ranged from 100-200 ng Hg/g.

<u>Particulate Matter (PM)</u>

Pirhalla et al. (2014) recorded relatively low levels of particulate matter in GLBA in 2008, even with the presence of cruise ships. The number of cruise ships traveling through the park is limited to no more than two each day to protect park resources and value. The average number of cruise ships traveling through the park in 2008 was 1.4 ships per day (Pirhalla et al. 2014). Particulate matter concentrations never exceeded the EPA NAAQS thresholds (150 $\mu g/m^3$), although there were a few instances where the hourly concentration reached 50 µg/m^3, which may have reduced visibility in the area. The particulate matter concentration was the strongest on 19 July 2008; concentrations exceeded 44 µg/m^3 in the Bay area and around surrounding waters (e.g., Cross Sound, Gulf of Alaska) (Pirhalla et al. 2014). The daily average reached 22 µg/m^3 in parts of the Bay. According to Pirhalla et al. (2014), the presence of inversions may result in emissions becoming trapped closer to the surface, and concentrations diminish as they rise higher into the atmosphere. For example, the average PM_{10} concentrations on 19 July 2008 were 20.5 µg/m^3 near the surface, 6.6 µg/m^3 above inversion levels, and 3.1 µg/m^3 400 m (1,312 ft) above inversion levels. Even on the day with the highest recorded levels of particulate matter, PM_{10} concentrations did not exceed the reference condition for air quality considered to be of good condition.

Visibility

Visibility impairment occurs when airborne particles and gases scatter and absorb light; the net effect is called “light extinction,” which is a reduction in the amount of light from a view that is returned to an observer (EPA 2003). The clearest and haziest 20% of days each year are also examined for parks (NPS ARD 2014), as these are the measures used by states and EPA to assess progress towards meeting the national visibility goal. Conditions measured near 0 dv are clear and provide excellent visibility, and as dv measurements increase, visibility conditions become hazier. The most current visibility estimates near GLBA were recorded between 2005 and 2008 (VIEWS 2010) (Figure 173). Visibility data (in dv) were collected at Petersburg (monitor ID PETE1), approximately 500 km (331 mi) southeast of GLBA; data show the average visibility on the clearest and haziest 20% of days each year for the region (VIEWS 2010). The clearest days fall into the *Moderate Condition* category, and the haziest days fall into the *Significant Concern* category.

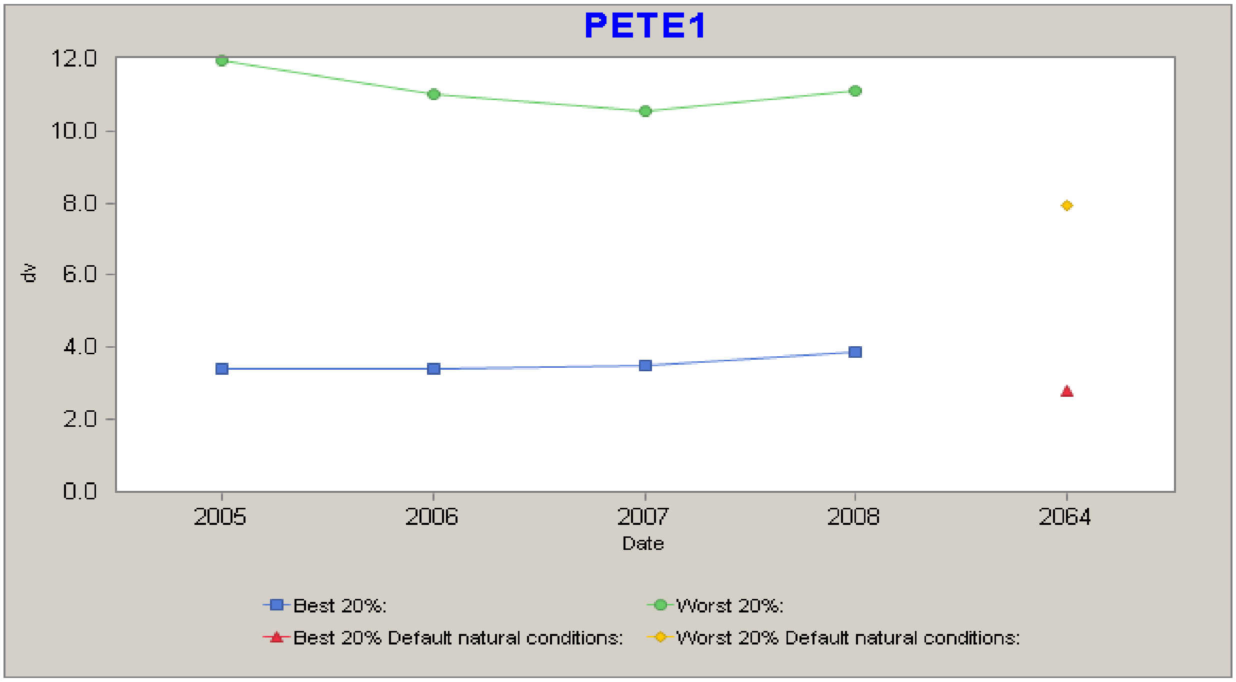

Figure 173. Annual average visibility in the GLBA region, 2005-2008 (VIEWS 2010). The Petersburg (PETE1) monitor is located southeast of Juneau, Alaska, approximately 500 km (311 mi) from GLBA. Values shown indicate average visibility during the 20% clearest and 20% haziest days each year. Values at 2064 are the natural background visibility conditions, set by EPA Regional Haze Rule.

Threats and Stressor Factors

GLBA staff identified two potential threats and stressors to the air quality in the park. Those threats include industrial activity and local point sources of contaminants (e.g., boats, Gustavus). Industrial air pollution sources are rare in Alaska, but airborne contaminants can be transported long distances, even over the Pacific Ocean from Asia, to southeastern Alaska (Landers et al. 2008).

Local point sources of contaminants can impact the air quality in GLBA. Point sources include boats (e.g., cruise ships, tour vessels) and the city of Gustavus, which is located on the southeast border of GLBA. According to Vequist (1993), air pollution in GLBA dates back to the 1970s and has potentially increased as a result of increased cruise ship visits. Boating activity, such as cruise ships, create visual blue-grey emissions, which results in impaired scenic vistas (Vequist 1993). Temperature inversions, which occur in fjords, have been known to trap boat emissions, causing those emissions to linger. Emissions have been documented lasting longer than 10 minutes with some lasting as long as 98 minutes (Vequist 1993). The city of Gustavus is another potential point source of pollution (e.g., wood smoke from houses); however, the city's small population size limits the amount of impact from this local source.

Data Needs/Gaps

There are no active air quality monitors within the preferred distance (16 km [10 mi]) to accurately represent conditions in the park. There are no long-term data regarding the air quality measures either. The closest active monitors are located more than 61 km (38 mi) away (Juneau, Alaska). The nearest active NADP monitor that provides annual averages for nitrogen and sulfur deposition is located in Juneau, Alaska. The nearest CASTNet and IMPROVE sites, that monitor acid deposition and visibility respectively, are located in Petersburg, over 500 km (310 mi) southeast of GLBA. The establishment of long-term, consistent monitoring of lichen samples, atmospheric deposition of nitrogen and sulfur, mercury deposition, particulate matter, and visibility would help managers better understand the local air quality conditions in and around GLBA and how they may affect other park resources. The SEAN is currently developing a monitoring plan for airborne contaminants, which is scheduled to be completed by 2018 (Michael Bower, SEAN Program Manager, written communication, 12 August 2015).

Overall Condition

Lichen Sampling

The *Significance Level* for lichen sampling is a 2. Schirokauer and Geiser (2011) documented three lichen species suitable for elemental analysis at two of three sites in GLBA. There were three lichen species collected at Bartlett Cove and two species collected at Blue Mouse Cove (Table 68). Weekly average concentrations of NO_2 at the Bartlett Cove site were below 5 µg/m^3 throughout the study. Weekly concentrations of NH_3 and NO_x were also low at Bartlett Cove, and concentrations did not exceed 20 µg/m^3 for either analyte.

Schirokauer et al. (2014) reported that elemental concentrations from lichen in Bartlett Cove had the lowest seasonal means in NO_2, NO_x, and SO_2 among all the sampling stations in 2008 and 2009, and Bartlett Cove had the lowest seasonal mean of HNO_3 in 2008 (Table 67). Potassium was the only element collected from *Platismatia glauca* at the Bartlett Cove site that had concentrations exceeding the threshold. Several elements collected from *P. glauca* and *Hypogymnia enteromorpha* samples at the Blue Mouse Cove site exceeded the thresholds. Those elements include potassium, lithium, nitrogen, and silicon. Schirokauer et al. (2014) found that air quality at most sites was in good condition, which was expected for the SEAN. Although Schirokauer et al. (2014) suggests that air quality is in good condition in the SEAN (including GLBA) the data are over 5 years old. Air quality conditions may have changed since, so a *Condition Level* was not assigned for this measure.

Atmospheric Deposition of Sulfur and Nitrogen

The *Significance Level* for atmospheric deposition of nitrogen and sulfur is a 2. Sullivan et al. (2011a) ranked GLBA as having low exposure to acidifying (nitrogen and sulfur) pollutants, high ecosystem sensitivity to acidification, and high park protection due to its Class II airshed status. The ranking of overall risk from acidification due to acid deposition was moderate relative to other parks (Sullivan et al. 2011a). Sullivan et al. (2011b) also ranked GLBA as having very low risk for nitrogen pollutant exposure, low ecosystem sensitivity, and very high park protection mandates (Class II airshed). The ranking of overall risk of effects from nutrient enrichment from atmospheric nitrogen deposition was low relative to other parks (Sullivan et al. 2011b). There were no data sources that measured atmospheric deposition of sulfur or atmospheric deposition of nitrogen in the park. Instead, the previous data sources used extrapolation to suggest that atmospheric deposition in GLBA is of good condition. As a result, this measure was not assigned a *Condition Level*.

Mercury Deposition

The *Significance Level* for mercury deposition is a 2. The MDN (2014) data display low annual average concentrations of mercury deposition in GLBA; however, there are not enough years of data to establish a trend, nor is the monitoring station currently active. Due to the lack of recent data, a *Condition Level* could not be assigned.

Particulate Matter

The *Significance Level* for particulate matter is a 2. Pirhalla et al. (2014) recorded relatively low levels of particulate matter in GLBA in 2008, even with the presence of cruise ships. Particulate matter concentrations never exceeded the EPA NAAQS thresholds (150 $\mu g/m^3$). There were a few instances where the hourly concentration reached 50 $\mu g/m^3$, which may have reduced visibility in the area. Even on the day with the highest recorded levels of particulate matter, PM_{10} concentrations did not exceed the reference condition for air quality considered to be of good condition. Even though Pirhalla et al. (2014) documented low levels of particulate matter in the park, the study is over 5 years old, and the levels of particulate matter may have changed since then. As a result, a *Condition Level* was not assigned to this measure.

Visibility

The *Significance Level* for visibility is a 2. There were no available visibility data in GLBA Because of the lack of data, a *Condition Level* could not be assigned.

<u>Weighted Condition Score</u>

The WCS for this component could not be assessed due to the lack of data for each of the selected measures. The lack of long-term data did not allow a trend to be assigned.

Air Quality			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Lichen Sampling	2	n/a	
Atmospheric Deposition of Sulfur and Nitrogen	2	n/a	
Mercury Deposition	2	n/a	
Particulate Matter	2	n/a	
Visibility	2	n/a	

4.21.6. Sources of Expertise

- Dave Schirokauer, DENA Chief of Resources (former KLGO Natural Resource Program Manager)
- Michael Bower, SEAN Program Manager

4.21.7. Literature Cited

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4.22. Water Quality

4.22.1. Description

With more than one half of the total surface area of GLBA covered by some form of water, water quality is a critical issue for the park. Approximately 445,155 ha (1.1 million ac) of freshwater are contained within GLBA, primarily in the form of snow and ice, rivers, lakes, and ponds (NPS 2008; Table 70). The remaining water, roughly 232,695 ha (575,000 ac), is either saltwater or brackish (NPS 2008). The watersheds within the park are relatively small, and most streams are short (less than 20 km [12 mi]) with steep gradients (5-20%) (Eckert et al. 2006). Freshwater input into these streams is provided predominantly by snow and glacial meltwaters. Glacial retreat within GLBA continually creates new landforms, including streams and ponds (Eckert et al. 2006).

Photo 54. View of Bartlett Cove (NPS photo).

Table 70. Water area in GLBA (NPS 2008).

Form	Hectares	Acres	Fresh/Salt
Snow and Ice	413,463	1,021,688	Fresh
Marine	226,011	558,485	Salt
Lake	15,130	37,387	Fresh
Riverine	12,109	29,923	Fresh
Estuarine	5,596	13,829	Salt
Pond	1,636	4,043	Fresh
Aquatic Herbaceous	538	1,330	Fresh

The importance of water quality has been recognized by the SEAN I&M Program, as the network designated both marine (saltwater) contaminants and freshwater contaminants as Vital Signs

(Moynahan et al. 2008). Pollutants in water can accumulate in food chains and endanger top predators and humans alike (EPA 2009). According to Eckert et al. (2006), mercury (Hg) and POPs are the global contaminants of highest concern in Alaska. Concentrations of Hg and other metals, POPs, and PAHs are included in SEAN's long-term contaminant monitoring program (Tallmon 2012).

Potential impacts to water resources at GLBA from these contaminants are considered a matter of critical importance. Most pollutants originate outside the park and are transported either directly or indirectly (Nagorski et al. 2011). Airborne POPs and Hg can be transported long distances on the prevailing winds and then deposited in GLBA. Much of this indirect atmospheric deposition in the park originates from the industrializing nations in Asia (Landers et al. 2008). Other methods of indirect pollutant input include biological vectors such as migratory animals (Blais et al. 2007, Baker et al. 2009). Direct anthropogenic inputs come from local human activities within and around GLBA.

Because of the differences in water characteristics and potentially different sources and types of pollutants or contaminants, freshwater and marine waters will be discussed separately in the current condition section of this NRCA.

4.22.2. Measures

Freshwater:

- Mercury (Hg) and methylmercury (MeHg)
- Persistent organic pollutants (POPs)

Marine:

- Metals
- POPs
- Polycyclic aromatic hydrocarbons (PAHs)
- Ocean acidification

Metals (including Hg and MeHg)

Depending on the type and amount of metals, their effect on water quality can range from beneficial to toxic. Metals in the environment can come from many diverse point and non-point sources. Metals that have originated from anthropogenic sources can often be correlated with specific processes or industries (Landers et al. 2008).

Mercury is one such metal that is entering national parks in Alaska through atmospheric deposition from local, regional, and trans-Pacific sources (Landers et al. 2008). Mercury is an elemental pollutant with a complex life cycle in the atmosphere and biosphere, which leads to some difficulty in detecting its origin (Landers et al. 2008). Anthropogenic sources (e.g., combustion, smelting, and petroleum refining) are thought to account for 75% of the Hg that enters the atmosphere, with the remainder originating from geologic and biogenic sources (Landers et al. 2008). Mercury enters national parks in Alaska through atmospheric deposition from local, regional, and trans-Pacific sources (Landers et al. 2008).

In water, Hg is often converted to methylmercury (MeHg), a neurotoxin 100 times more toxic than elemental Hg (USGS 2000). MeHg biomagnifies in the aquatic food web and can be transferred to new environments through animal movements (i.e., "biovectors"). For example, developing salmon can accumulate MeHg and other contaminants as they grow in pelagic waters and ultimately deposit them into their riverine spawning environment (Quinn 2005, Baker et al. 2009). Migratory birds and aquatic insects, such as mayflies (Order Ephemeroptera), have also been known to transfer contaminants to new environments (Blais et al. 2007, Walters et al. 2008).

Cadmium (Ca), Arsenic (As), and Tributyltin (TBT) are three additional metals of concern that have been monitored by the SEAN marine contaminants program (Tallmon 2012). Cadmium is used on machinery and equipment in metal coatings or platings (EPA 2013a). Arsenic is an odorless and tasteless metal that can enter the water supply either naturally or from agricultural or industrial processes (EPA 2013b). Arsenic is associated with a host of physical abnormalities such as stomach pain, vomiting, numbness of the hands and feet as well as being linked to many types of cancers (EPA 2013b). Tributyltin is a compound that was used primarily in paint as an anti-fouling agent and biocide (EPA 2004). Tributyltin is an endocrine-disrupting chemical that can have profound negative impacts on reproductive systems of aquatic organisms (EPA 2004). Impairments in growth, reproduction, and survival of many marine species from TBT exposure have been documented (Bryan et al. 1986, Hagger et al. 2005).

<u>Persistent Organic Pollutants (POPs)</u>

POPs are very stable organic compounds that can be transported over long distances by wind and water. These chemicals enter the water system via effluent releases (e.g., industrial waste), atmospheric deposition, runoff, and by other means (EPA 2009). Currently, 90 countries have agreed to reduce the production and use of 12 specific POPs under a treaty called the Stockholm Convention formulated by the United Nations Environment Programme (UNEP) (EPA 2009). These 12 POPs, known as the "dirty dozen," are aldrin, cholrdane, dichlorodiphenyltrichloroethane (DDT), dieldrin, endrin, heptachlor, hexachlorobenzene, mirex, toxaphene, PCBs, dioxins, and furans (EPA 2009). The United States has already stopped production of many POPs mentioned in the agreement, but POPs are still unintentionally released from industrial processes and combustion (EPA 2009).

Persistent organic pollutants can concentrate in the bodies of many different organisms and may cause physical, behavioral, and reproductive abnormalities in wildlife and humans (EPA 2009). POPs are lipid-soluble and bioconcentrate in the fatty tissue of organisms, negatively affecting organisms at higher trophic levels. As with Hg, biovectors may play a role in transporting POPs to new environments. A strong correlation between salmon runs and PCB concentrations in lake sediments was found by Krummel et al. (2003). Decaying salmon may concentrate contaminants in other species by providing a large and accessible food source for a variety of wildlife (Blais and Smol 2006).

Glacial ice is known to be a repository for the atmospheric deposition of contaminants, including POPs (Bogdal et al. 2010). When the glacial ice melts, POPs and other contaminants may be released from the glacier into the environment. Glaciers cover approximately 38% of GLBA and are central to the park's identity (Loso et al. 2014). This glacial ice may be a potential contamination source for park waters, although the extent to which this is an issue in GLBA is unknown.

Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are a group of over 100 different chemicals that result from the burning of organic substances such as coal, oil, gas, wood, and garbage (ATSDR 1995). Although PAHs often enter the environment naturally from such sources as volcanoes and forest fires, there are many anthropogenic sources of PAHs, including coal burning, residential wood burning, exhaust from vehicles, and discharge from industrial and wastewater treatment plants (ATSDR 1995). PAHs can be found in air, water, and soil. When airborne, these chemicals can travel great distances before falling as rain or settling to the earth. Certain PAHs have been found to cause tumors in laboratory animals and have been shown to impair reproduction (ATSDR 1995). Short and long-term exposure to PAHs has also been found to cause harm to an animal's skin, body fluids, and immune system (ATSDR 1995).

Ocean Acidification

Ocean acidification is a term that refers to the lowering of the ocean's pH (i.e., hydrogen ion concentration). Atmospheric carbon dioxide (CO_2) most often reacts with ocean water and combines with a free carbonate ion in the water, releasing hydrogen ions and leaving the water more acidic (Mathis et al. 2014; Figure 174). Because this process utilizes a carbonate ion from the ocean, there are fewer carbonate ions remaining in the marine environment. This reduction in free carbonate ions impacts the ability of marine organisms such as mollusks to create and/or maintain their protective shells and skeletons (Mathis et al. 2014). Organisms that rely on carbonate play a critical role in the marine ecosystem as either key food species or as builders of shelter (corals) for animals as well as a form of natural erosion control for shorelines and beaches (Cohen and Holcomb 2009). As a result, OA is a direct risk to Alaska's food chain and commercial fisheries industry (Mathis et al. 2014).

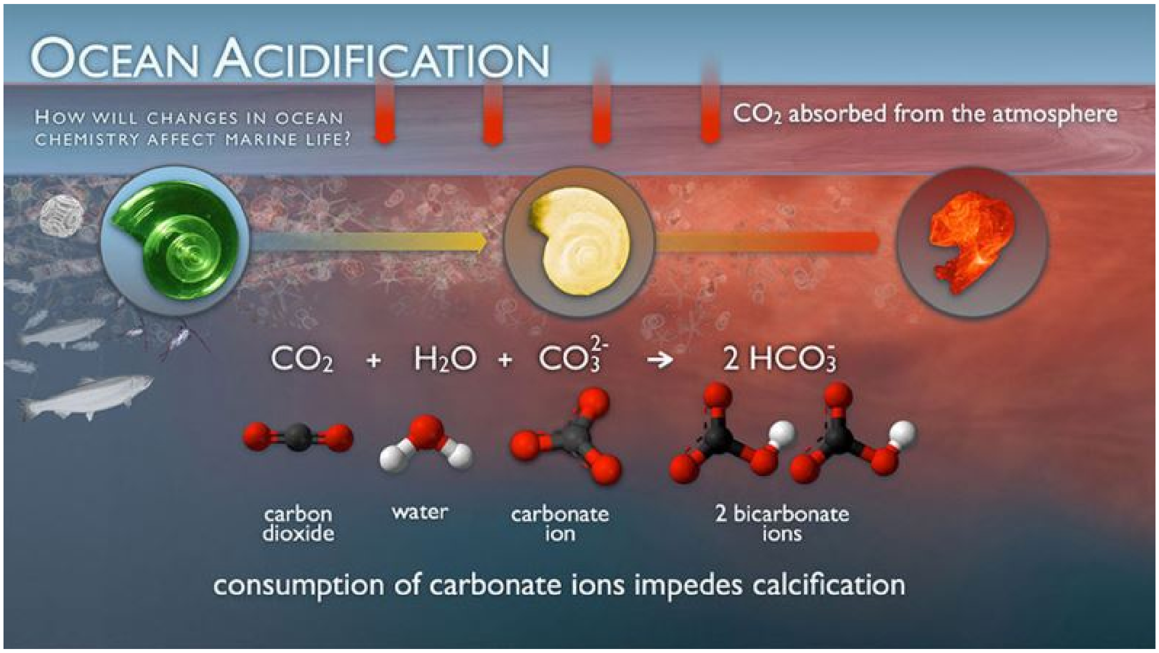

Figure 174. Diagram of the ocean acidification process (NOAA PMEL 2015).

Ocean acidification is directly linked to increased levels of CO_2 in the atmosphere (Mathis et al. 2014). Atmospheric CO_2 has increased since the industrial revolution of the 18th and 19th centuries. About one third of this CO_2 increase has been absorbed by our oceans (Sabine et al. 2004). The impact from the increased CO_2 in ocean waters has resulted in a decrease of ocean pH by ~ 0.1 units since preindustrial times (Byrne et al. 2009).

4.22.3. Reference Conditions/Values

Reference conditions for the various contaminant measures were obtained from the AK DEC water quality standards (AK DEC 2008, 2012). The criteria for the protection of aquatic life in freshwater and marine waters are shown in Table 71. Criteria are not available for POPs as a group, but are set for specific chemicals; several that have been tested for or detected at GLBA have been provided as examples in Table 71. The reference condition for ocean acidification will be the typical pH range for North Pacific surface waters of 7.6-8.0 (Byrne et al. 2009). Since ocean acidification is linked to atmospheric CO_2 levels, an additional reference could be the 2014 mean global atmospheric CO_2 level of 395 ppm (NOAA ESRL 2015).

Table 71. Reference standards for various contaminant measures (AK DEC 2008, unless otherwise noted). CMC indicated the Criteria Maximum Concentration and CCC indicates the Criterion Continuous Concentration.

Contaminant	Aquatic Life Criteria	
	Freshwater	Marine
Metals		
Arsenic	340 µg/l CMC (acute) / 150 µg/l CCC (chronic)	69 µg/l CMC (acute) / 36 µg/l CCC (chronic)
Cadmium	varies with hardness of water	40 µg/l CMC (acute) / 8.8 µg/l CCC (chronic)
Tributyltin	0.46 µg/l CMC (acute) / 0.072 µg/l CCC (chronic)	0.42 µg/l CMC (acute) / 0.0074 µg/l CCC (chronic)
Mercury (Hg)[A]	1.4 µg/l CMC (acute) / 0.77 µg/l CCC (chronic)	1.8 µg/l CMC (acute) / 0.94 µg/l CCC (chronic)
POPs		
Chlordane (CHLD)	2.4 µg/l CMC (acute) / 0.0043 µg/l CCC (chronic)	0.09 µg/l CMC (acute) / 0.004 µg/l CCC (chronic)
Dieldrin	0.24 µg/l CMC (acute) / 0.056 µg/l CCC (chronic)	0.71 µg/l CMC (acute) / 0.0019 µg/l CCC (chronic)
Mirex	0.001 µg/l CCC (chronic)	0.001 µg/l CCC (chronic)
Heptachlor	0.52 µg/l CMC (acute) / 0.0038 µg/l CCC (chronic)	0.053 µg/l CMC (acute) / 0.0036 µg/l CCC (chronic)
PCBs	0.014 µg/l CCC (chronic)	0.03 µg/l CCC (chronic)
PAHs[B]		
TAH	Total aromatic hydrocarbons (TAH) in water may not exceed 10 µg/L.	Total aromatic hydrocarbons (TAH) in water may not exceed 10 µg/L.

[A] The recommended criteria were derived from data for inorganic mercury (II), but are applied here to total mercury. If a substantial portion of the mercury in the water column is methylmercury, the criteria will probably be under protective.

[B] AK DEC 2012

Table 72. Range of mussel contaminant levels considered “low” by the NOAA Mussel Watch program, based on nationwide samples (Kimbrough et al. 2008).

Contaminant	Low Range*
Metals	
Arsenic	5-11 µg/g
Cadmium	0-3 µg/g
Butyltins	1-39 ng/g
Mercury (Hg)	0.00-0.17 µg/g
PAHs	63-1,187 ng/g

* Ranges were given in the source in ppm or ppb dry weight; ppm = µg/g, ppb = ng/g.

Table 72 (continued). Range of mussel contaminant levels considered “low” by the NOAA Mussel Watch program, based on nationwide samples (Kimbrough et al. 2008).

Contaminant	Low Range*
POPs	
Chlordanes (CHLD)	0-8 ng/g
Dieldrins	0-8 ng/g
PCBs	3-153 ng/g
DDTs	0-112 ng/g

* Ranges were given in the source in ppm or ppb dry weight; ppm = µg/g, ppb = ng/g.

4.22.4. Data and Methods

Eckert et al. (2006) provides a comprehensive summary of the GLBA water quality conditions (freshwater and marine) and any possible impairment to GLBA waters as of 2006. This report includes valuable historical background on the park, a summary of water quality conditions, and lists of contaminants of concern (Eckert et al. 2006).

Nagorski et al. (2011) reported on freshwater contaminants present within SEAN parks. These data are based on samples taken from 16 different streams in and around GLBA in June–July of 2007. Streams were classified into three categories based on watershed age: young (<100 years), medium (100-200 years old), and old (>1,000 years old). In addition to water and streambed samples, Nagorski et al. (2011) also sampled benthic invertebrates and juvenile coho salmon to provide a baseline for Hg and POPs in SEAN watersheds. The SEAN is currently developing a freshwater contaminants monitoring program based on this work and additional pilot work conducted in 2013. A protocol will be in place by the end of 2017 (Sergeant, written communication, 16 October 2015).

Tallmon (2012) analyzed mussel tissue and sediment samples collected from GLBA intertidal zones in July and August of 2007, 2009, and 2011. Contaminants selected for analysis included several metals (arsenic, cadmium, and mercury), PAHs, and POPs. Refer to Chapter 4.2.4 of this NRCA (Marine Shoreline) for further description of this research and a map of sampling locations.

Eagles-Smith et al. (2014) analyzed mercury levels in Dolly Varden tissues from three water bodies in GLBA (Falls Creek, North Skidmore Lake, and Stonefly Lake). Fifteen samples were taken from each site in 2012 (Eagles-Smith et al. 2014).

Mathis et al. (2014) conducted an ocean acidification risk assessment for selected Alaska fisheries (shellfish, salmon, and other finfish). The report describes the OA process and the potential impacts to animals dependent on calcium to create and/or maintain their hard shells and skeletons. Reisdorph and Mathis (2014) used observed and projected aragonite saturation rates from GLBA to assess ocean acidification risk. Aragonite is a calcium carbonate compound utilized by shell-building organisms. Aragonite saturation rates were calculated from water samples collected at 22 stations within and just outside Glacier Bay between July 2011 and July 2012 (Reisdorph and Mathis 2014).

GLBA's unique susceptibility to OA due to the presence of tidewater glaciers is also discussed in detail.

4.22.5. Current Condition and Trend

<u>Freshwater</u>

Mercury and Methylmercury

The presence of Hg has been detected throughout the waters of GLBA. Of particular concern for GLBA and Alaska in general is that since the reduction of Hg emissions in the continental U.S. (1970s), Alaska's level of mercury deposition has not correspondingly decreased (Figure 175; Engstrom and Swain 1997, Eckert et al. 2006). This strongly suggests that the primary source of mercury contamination to the area is from distant, developing countries and largely beyond the control of the state and parks (Engstrom and Swain 1997, Pacyna and Pacyna 2002).

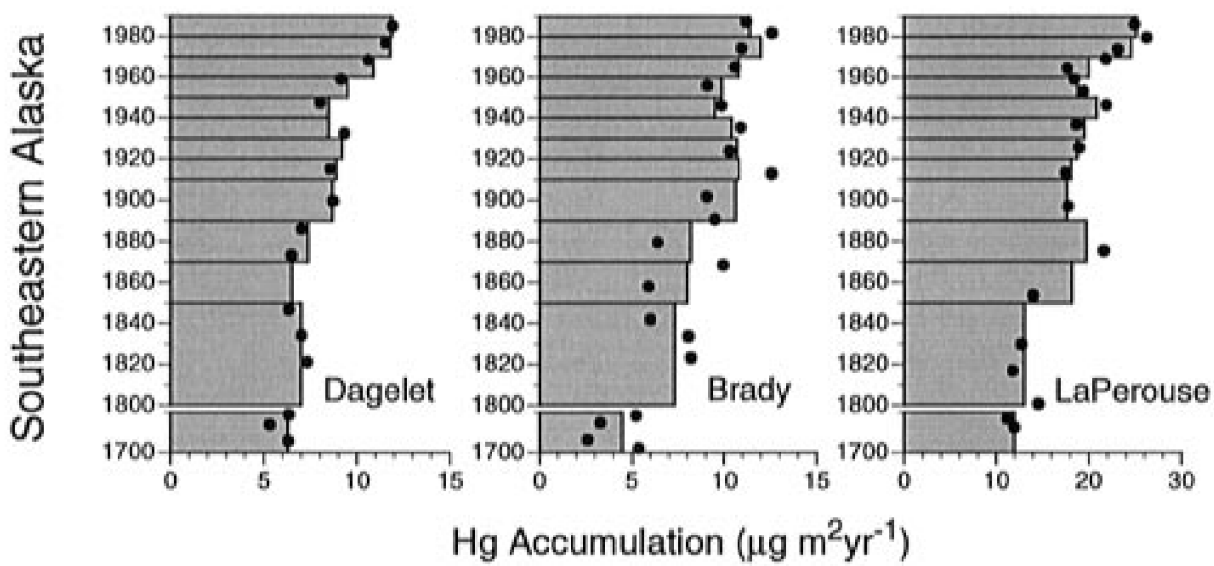

Figure 175. Pb-dated cores from selected lakes in GLBA (Engstrom and Swain 1997).

Hg concentrations from GLBA water (filtered) and particulate samples demonstrated a pattern of increased concentration with the age of the watershed (Table 73; Nagorski et al. 2011). The younger watersheds in GLBA (<100 yrs) are characterized by receiving glacial runoff and have well-drained, thin soils with shifting stream channels, exposed bedrock and little vegetative cover (Photo 55; Nagorski et al. 2011). These younger watersheds have little ability to retain Hg and complete its subsequent conversion to MeHg due to these characteristics (Nagorski et al. 2011).

Table 73. Water (filtered) and particulate HgT and MeHg concentrations in sampled streams. “SW total” is the sum of filtered and particulate concentrations (Nagorski et al. 2011).

Watershed Age	Stream	HgT Filtered (ng/L)	HgT Particulate (ng/L)	HgT SW Total (ng/L)	MeHg Filtered (ng/L)	MeHg Particulate (ng/L)	MeHg SW Total (ng/L)
Young	Stonefly Cr.	0.47	0.97	1.44	0.02	0.01	0.03
	Gull Cr.	0.39	0.2	0.59	0.04	0.02	0.05
	Nunatak Cr.	0.08	0.19	0.27	0.01	<0.01	0.01
	Mean	0.31	0.45	0.76	0.02	0.01	0.03
	Std. dev.	0.21	0.45	0.6	0.01	0.01	0.02
Medium	Reid Cr.	0.89	3.73	4.62	0.04	0.01	0.04
	Tyndall	1.28	0.07	1.35	0.03	<0.01	0.03
	Ice Valley	0.22	0.11	0.33	0.1	<0.01	0.11
	Vivid Lake Cr.	0.35	1.39	1.74	<0.01	<0.01	<0.01
	Oystercatcher	0.84	0.13	0.97	0.06	<0.01	0.06
	Fingers South	0.39	0.99	1.38	0.02	<0.01	0.03
	Berg Bay So.	0.56	0.77	1.33	0.02	0.01	0.02
	Rush Point	0.33	0.18	0.51	0.04	<0.01	0.04
	Mean	0.61	0.92	1.53	0.04	<0.01	0.04
	Std. dev.	0.36	1.24	1.34	0.03	<0.01	0.03
Old	Carolus River	0.42	0.26	0.68	0.02	0.01	0.02
	East Falls Cr.	0.53	0.09	0.62	0.01	<0.01	0.02
	Rink Cr.	0.98	0.16	1.14	0.21	0.01	0.21
	E. Pleasant Is.	3.37	0.17	3.54	0.09	0.01	0.1
	W. Pleasant Is.	3.08	0.17	3.25	0.06	<0.01	0.07
	Mean	1.68	0.17	1.85	0.08	<0.01	0.08
	Std. dev.	1.43	0.06	1.43	0.08	<0.01	0.08

Photo 55. A stream below Brady Glacier (NPS photo).

The medium watersheds (100-200 yrs) are characterized by stands of spruce (*Picea* spp.), hemlock (*Tsuga* spp.), and cottonwood trees with stable stream channels and increased biodiversity (Nagorski et al. 2011). These watersheds are more impacted by mercury than young watersheds, but potentially less impacted than older watersheds. Water samples in the younger and medium-aged watersheds showed only 20-25% of the average value of incoming atmospheric deposition (of Hg) (Nagorski et al. 2011). The decreased amount of Hg, and consequently MeHg, in these watersheds is believed to be due to the lack of wetlands and organic rich peatlands (Benoit et al. 1999). The young and medium watersheds do not contain the carbon to support the sulfate-reducing bacteria necessary for the methylation process (Nagorski et al. 2011). This results in low levels of MeHg and consequently very limited exposure for biotic organisms, thus resulting in very low levels of bioaccumulation (Nagorski et al. 2011). These watersheds have been shown to be net sinks of atmospherically derived Hg (St. Louis et al. 1996).

The older GLBA watersheds (>1,000 yrs) are characterized by high channel complexity with significant wetlands (Photo 56) as well as mature forests (Nagorski et al. 2011). The abundance of wetlands in these older watersheds contributes to an abundance of dissolved organic carbon (Nagorski et al. 2011). It is the presence of carbon that not only stimulates the methylation process (Hg to MeHg), but also provides binding sites for the transport of the MeHg into the water network (Benoit et al. 1999, Nagorski et al. 2011). Watersheds such as these are typically considered net sources of MeHg to streams (St. Louis et al. 1996).

Photo 56. Pond/Wetland area in GLBA (NPS photo).

Levels of Hg and MeHg in GLBA samples are relatively low compared to other sampled locations in the U.S. and are three to four orders of magnitude below the EPA and AK DEC standards for aquatic life (AK DEC 2008, Nagorski et al. 2011). However, Hg and MeHg are still a concern for GLBA due to several factors (Nagorski et al. 2014). First, the primary source of the contamination is on the Asian continent and therefore beyond U.S. regulatory control. Second, the abundance of wetlands within GLBA provides ample environment to facilitate the conversion of Hg to MeHg. Third, the presence of salmon runs in GLBA streams has the potential to import and concentrate Hg and MeHg from other areas (Nagorski et al. 2014).

Mean mercury levels in fish from GLBA streams ranged from 2.5-80.6 ng/g (Table 74; Nagorski et al. 2011). Mercury levels in GLBA juvenile (age 0+) coho salmon were well below the NOER level of 200 ng/g identified by Beckvar et al. (2005). Levels in coho salmon (age 1+) were recorded as high as 80 ng/g, but still did not exceed the NOER level (Nagorski et al. 2011).

Table 74. Total Hg (ng/g wet weight) in coho salmon tissue samples from GLBA (Nagorski et al. 2011). n=sample size. Comparable studies to this are also listed.

		Age 0 Coho		Age 1 Coho		Age 2 Coho	
Watershed Age	**Stream**	**n**	**HgT (ng/g)**	**n**	**HgT (ng/g)**	**n**	**HgT (ng/g)**
Young	Stonefly Creek	5	5.2	–	–	–	–
	Gull Creek	15	4.2	–	–	–	–
	Nunatak Creek	3	3.3	–	–	–	–
	Mean	–	**4.2**	–	–	–	–
	Std. Dev	–	**0.9**	–	–	–	–

* Originally reported as dry weight and converted to wet weight assuming 20% solids.

Table 74 (continued). Total Hg (ng/g wet weight) in coho salmon tissue samples from GLBA (Nagorski et al. 2011). n = sample size. Comparable studies to this are also listed.

Watershed Age	Stream	Age 0 Coho		Age 1 Coho		Age 2 Coho	
		n	HgT (ng/g)	n	HgT (ng/g)	n	HgT (ng/g)
Medium	Reid Creek	5	14.1	3	70	–	–
	Tyndall Stream	6	3.1	–	–	–	–
	Ice Valley Stream	5	2.8	–	–	–	–
	Vivid	8	3.4	–	–	–	–
	Oystercatcher	6	5.1	–	–	–	–
	Fingers South	2	2.5	–	–	–	–
	Berg Bay South	4	8.9	2	30.8	–	–
	Rush Point Stream	11	3.4	1	80.6	1	49.2
	Mean	–	**5.4**	–	–	–	–
	Std. Dev	–	**4.1**	–	–	–	–
Old	Carolus River	3	13.4	–	–	–	–
	East Falls Creek	4	3.5	–	–	–	–
	Rink Creek	5	9.2	–	–	–	–
	E. Pleasant Is.	1	13.9	–	–	–	–
	W. Pleasant Is.	5	5.6	–	–	–	–
	Mean	–	**9.1**	–	–	–	–
	Std. Dev	–	**4.6**	–	–	–	–

Comparable Studies

Location	Reference	Hg (ng/g)	Fish species	Age
Juneau-McGinnis Cr.	Nagorski, unpublished data	8.9	Coho	Juvenile-Age 0+
Juneau-Fish Cr.	Nagorski, unpublished data	73	Coho	Juvenile-Age 0+
Juneau-Peterson Cr.	Nagorski, unpublished data	34	Coho	Juvenile-Age 0+
Innoko NWR, AK	Mueller and Matz 2002	40	Coho	Juvenile
Kuskokwim area, AK	Gray et al. 1996	70	Coho	Juvenile
Illinois Cr., AK	Winters 1996	12-28*	Coho	Juvenile-Age 0+ and 1+
Voyageurs NP, MN	Wiener et al. 2006	36-190*	Yellow perch	Juvenile-Age 1
Columbia River, OR	Webb et al. 2006	34 ± 3	Sturgeon	Juvenile
Cook Inlet region, AK	Frenzel 2000	16-42*	Slimy sculpin	Adults

* Originally reported as dry weight and converted to wet weight assuming 20% solids.

Mercury levels in GLBA mayfly larvae samples (Table 75) and streambed sediment (Table 76) were also both below national averages (Nagorski et al. 2011). As with water and fish samples, Hg and MeHg levels in mayfly and sediment samples were lowest in young watersheds and highest in older watersheds (Nagorski et al. 2011).

Table 75. Total Hg and MeHg concentrations in the two sampled mayfly families. "NP" indicated that the family was not present in stream and "NS" indicated the family was present but there was not sufficient mass for analysis (Nagorski et al. 2011).

		Baetidae			Heptageniidae		
Watershed Age	**Streams**	**MeHg (ng/g)**	**HgT (ng/g)**	**% methyl**	**MeHg (ng/g)**	**HgT (ng/g)**	**% methyl**
Young	Stonefly Creek	NS	NS	NS	NP	NP	NP
	Gull Creek	14.5	22.2	66%	NP	NP	NP
	Nunatak Creek	4.7	12.6	37%	13.0	21.0	62%
	Mean	**9.6**	**17.4**	**0.5**	**13.0**	**21.0**	**0.6**
	Std. Dev	**7.0**	**6.8**	**0.2**	**N/A**	**N/A**	**N/A**
Medium	Reid Creek	20.5	59.3	35%	NS	NS	NS
	Tyndall Stream	13.1	38.4	34%	27.2	67.5	40%
	Ice Valley Stream	5.30	12.37	43%	8.43	17.24	49%
	Vivid	10.1	31.7	32%	NP	NP	NP
	Oystercatcher	29.2	52.5	56%	15.1	50.6	30%
	Fingers South	7.1	19.5	37%	9.5	51.1	19%
	Berg Bay South	10.5	24.9	42%	12.6	28.2	45%
	Rush Point Stream	27.6	36.3	76%	38.4	57.6	67%
	Mean	**15.43**	**34.38**	**0.44**	**18.55**	**45.37**	**0.42**
	Std. Dev.	**9.20**	**15.92**	**0.15**	**11.83**	**18.89**	**0.17**
Old	Carolus River	19.1	29.3	65%	21.5	44.8	48%
	East Falls Creek	19.0	36.5	52%	20.5	64.8	32%
	Rink Creek	58.3	51.3	114%	NP	NP	NP
	E. Pleasant Island	NS	NS	NS	26.6	55.2	48%
	W. Pleasant Island	NS	NS	NS	39.2	52.8	74%
	Mean	**32.1**	**39.0**	**0.8**	**26.9**	**54.4**	**0.5**
	Std. Dev	**22.7**	**11.2**	**0.3**	**8.6**	**8.2**	**0.2**

Table 76. Total Hg and MeHg concentrations in particulates and bed sediment in GLBA (Nagorski et al. 2011). (BDL= Below Detection Limit).

Watershed Age	Stream	Particulates HgT (ng/g)	Particulates MeHg (ng/g)	Bed Sediment HgT (ng/g)	Bed Sediment MeHg (ng/g)
Young	Stonefly Creek	33	0.4	7	0.05
	Gull Creek	131	10.0	11	0.44
	Nunatak Creek	184	BDL	5	0.01
	Mean	**116**	**BDL**	**8**	**0.17**
	Std. dev.	**77**	–	**3**	**0.24**
Medium	Reid Creek	92	0.2	N/A	0.02
	Tyndall	742	BDL	13	0.03
	Ice Valley	231	0.0	19	0.01
	Vivid Lake Creek	145	BDL	41	0.02
	Oystercatcher	235	BDL	13	0.03
	Fingers South	45	BDL	N/A	N/A
	Berg Bay South	94	0.7	23	0.04
	Rush Point	467	BDL	22	0.10
	Mean	**256**	**BDL**	**22**	**0.04**
	Std. dev.	**237**	–	**10**	**0.03**
Old	Carolus River	108	2.1	22	0.01
	East Falls Creek	122	BDL	N/A	0.05
	Rink Creek	125	4.5	7	0.04
	E. Pleasant Island	935	40	16	0.04
	W. Pleasant Island	347	BDL	14	0.06
	Mean	**327**	**BDL**	**15**	**0.04**
	Std. dev.	**354**	–	**6**	**0.02**

Mercury levels in Dolly Varden from three GLBA water bodies sampled in 2012 were also below the NOER for fish of 200 ng/g (Beckvar et al. 2005, Eagles-Smith et al. 2014). Geometric means for the three sites ranged from 56.4-114.7 ng/g, with an overall park mean of 90.2 ng/g (Table 77). The high variation between sites suggests that local processes are likely driving Hg bioaccumulation in the park (Eagles-Smith et al. 2014). Mercury concentrations detected at GLBA in this study were similar to levels found in Dolly Varden at other Alaska sites (40-230 ng/g; Jewett and Duffy 2007).

Table 77. Total Hg concentrations (ng/g wet weight) in Dolly Varden tissue samples from GLBA water bodies (Eagles-Smith et al. 2014).

Site	Geometric Mean	C.I.$_{95}$
Falls Creek	56.4	50-63.6
North Skidmore Lake	113.7	102.3-126.4
Stonefly Lake	114.7	91.9-143.1
Park mean	**90.2**	**79.1-102.9**

POPs

Persistent organic pollutants arrive in GLBA much the same way that Hg is deposited. The atmospheric deposition patterns, as well as ocean currents, favor cooler environments such as GLBA for depositing POPs (Figure 176). POPs tend to evaporate at the warmer tropical and subtropical environments and condense and be deposited in cooler environments (Wania and Mackay 1996). The cooler environments further exacerbate the POP problem by slowing decomposition and thereby making them even more persistent (Wania and Mackay 1996).

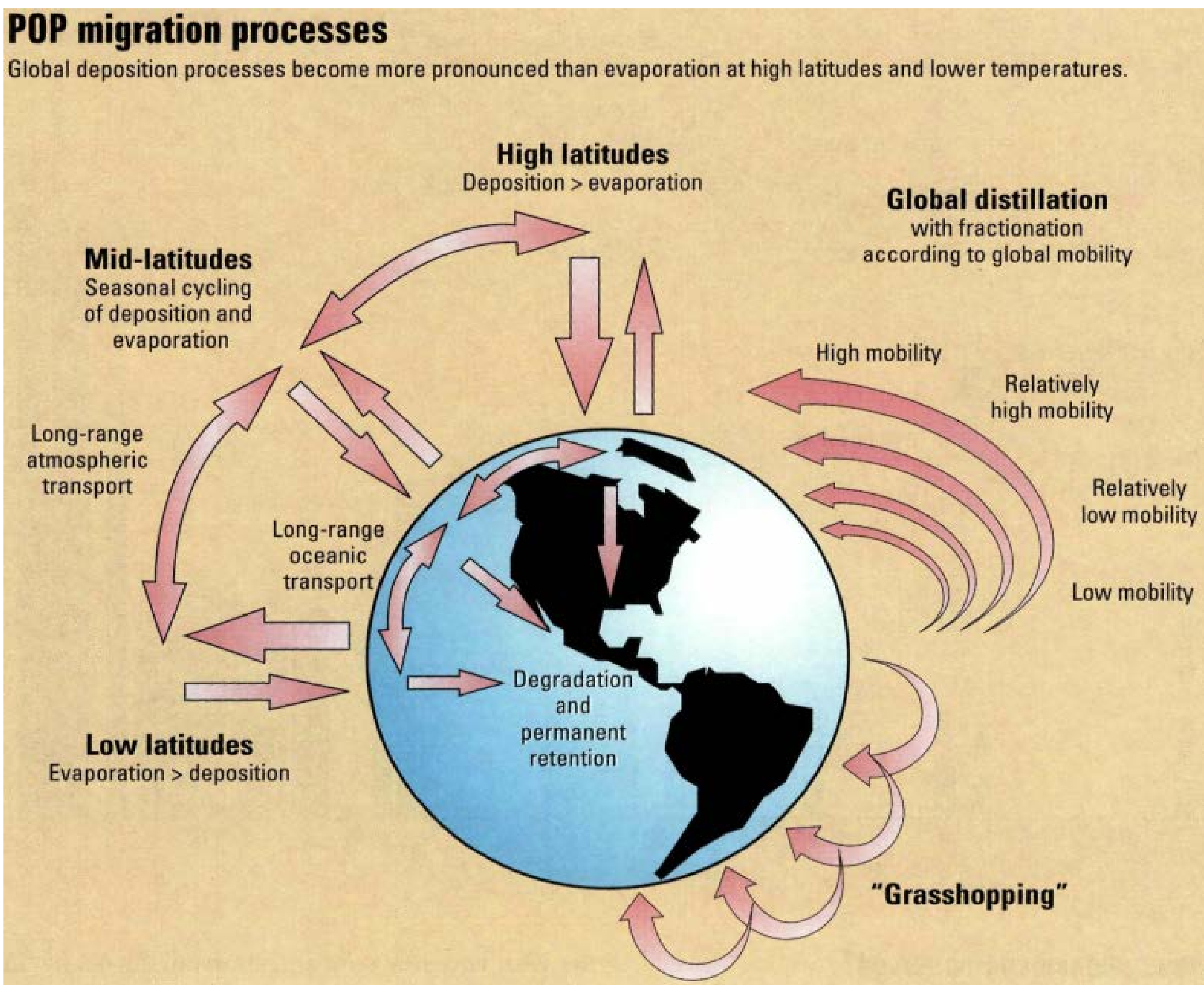

Figure 176. Migration of POPs worldwide (Wania and Mackay 1996).

Because of the highly volatile nature of some POPs, such as αHCH, they exhibit a uniquely inverted distribution pattern, as shown in Figure 177. The αHCH distribution is believed to originate from the southern edge of the distribution (Wania and Mackay 1996) and increase to the north and east. Note that concentrations increase as the distance from the source increases. While this pattern is not seen with all POPs, it does highlight the risk of POP concentration in the cooler climates of the higher latitudes (Wania and Mackay 1996).

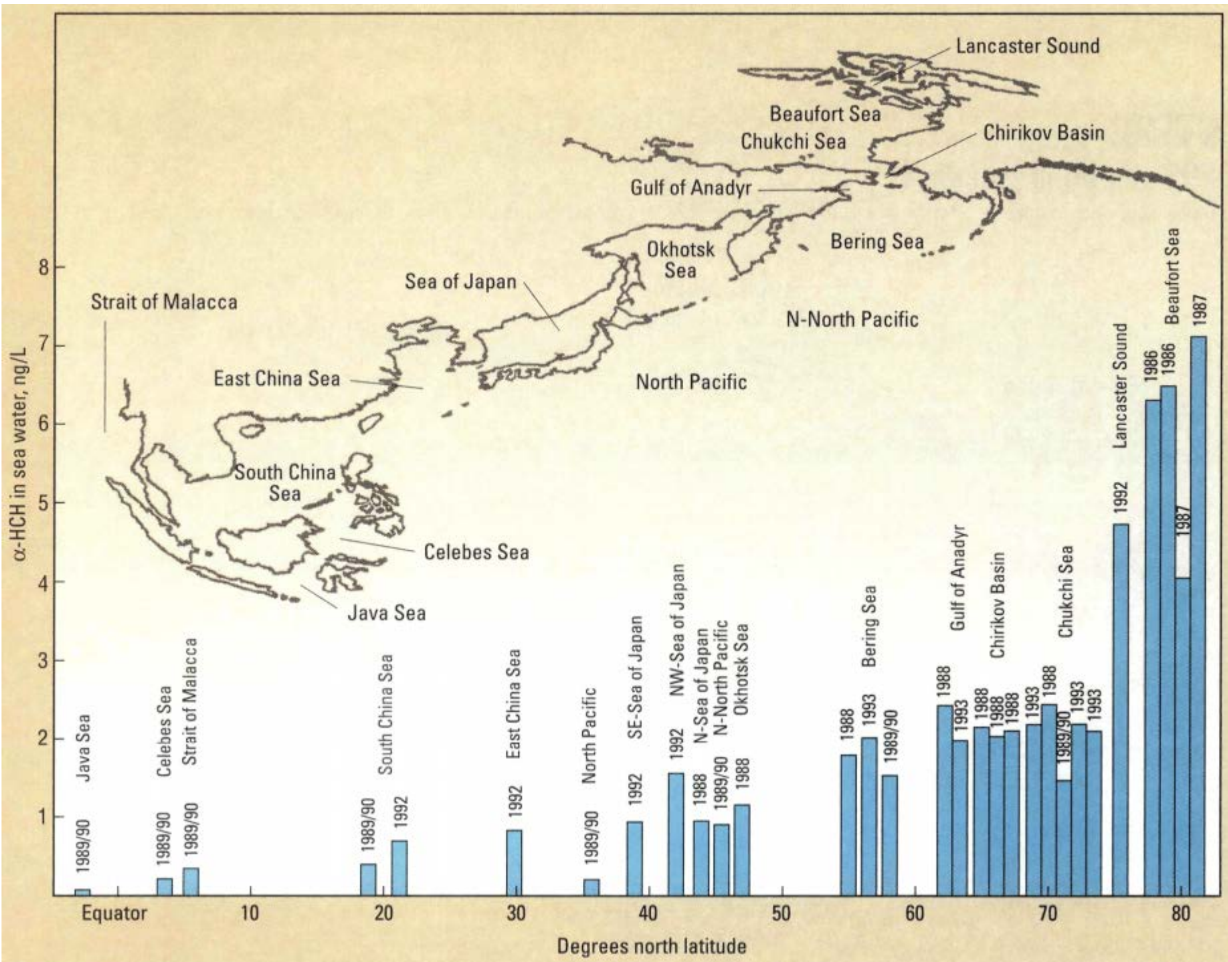

Figure 177. αHCH distribution in sea water. αHCH concentration increases from south (source) to north (depositional area) (Wania and Mackay 1996).

Coho salmon in GLBA were tested for the presence of 77 POPs (see Table 3 in Nagorski et al. [2011] for a full list). These POPs fall into three general groupings: industrial/urban use (e.g., PCBs, PBDEs), current use pesticides (e.g., endosulfan, lindane), and historic-use pesticides (e.g., aldrin, HCB). The types of POPs detected often varied between watersheds, but results from the majority of samples were below quantification limits (Table 78).

Table 78. Summed concentrations of various POPs (in ng/g, wet weight) in GLBA.

Study	Stream	Location	Sample Size	% Lipid	HCB[A]	∑HCHs[A]	∑CHLDs[A]	PCBs			Dieldrin[A]	Mirex[A]
								∑DDTs[A]	∑ 40 CB[A]	∑BDEs[A]		
Nagorski et al. 2011 *(young of the year coho captured in natal streams)*	Stonefly Cr.	GLBA	n=5	1.20%	<LOQ	<LOQ	<LOQ	1.2	1.7	<LOQ	<LOQ	<LOQ
	Gull Cr.	GLBA	n=10	1.40%	0.62	<LOQ	0.6	2.1	3.2	<LOQ	<LOQ	<LOQ
	Nunatak Cr.	GLBA	n=4	1.30%	0.29	<LOQ	<LOQ	0.7	1.1	<LOQ	<LOQ	<LOQ
	Reid Cr.	GLBA	n=5	0.90%	<LOQ	<LOQ	<LOQ	0.92	<LOQ	<LOQ	<LOQ	<LOQ
	Tyndall Stream	GLBA	n=6	1.30%	0.58	<LOQ	0.45	2.4	3.1	<LOQ	<LOQ	<LOQ
	Ice Valley Stream	GLBA	n=5	1.10%	<LOQ	<LOQ	<LOQ	0.94	2.7	<LOQ	<LOQ	<LOQ
	Vivid Stream	GLBA	n=7	1.60%	0.43	<LOQ	<LOQ	0.87	1.1	<LOQ	<LOQ	<LOQ
	Oystercatcher	GLBA	n=4	1.20%	<LOQ	<LOQ	0.25	1.1	2.1	<LOQ	<LOQ	<LOQ
	Fingers South	GLBA	n=3	1.40%	<LOQ	<LOQ	<LOQ	0.9	3.8	<LOQ	<LOQ	<LOQ
	Berg Bay South	GLBA	n=4	1.10%	<LOQ	<LOQ	<LOQ	1.1	1.6	<LOQ	<LOQ	<LOQ
	Rush Point Stream	GLBA	n=1	1.60%	0.35	<LOQ	0.21	0.91	1.9	<LOQ	<LOQ	<LOQ
	Carolus River	GLBA	n=3	1.40%	<LOQ	<LOQ	<LOQ	0.51	0.37	<LOQ	<LOQ	<LOQ
	East Falls Cr.	GLBA	n=4	1.40%	0.53	<LOQ	<LOQ	1.4	0.99	<LOQ	<LOQ	<LOQ
	Rink Cr.	GLBA	n=5	1.20%	0.29	<LOQ	0.22	0.86	2.3	<LOQ	<LOQ	<LOQ
	Pleasant Island East	adj. to GLBA	n=1	0.50%	<LOQ	<LOQ	<LOQ	<LOQ	0.92	<LOQ	<LOQ	<LOQ
	Pleasant Island West	adj. to GLBA	n=7	0.90%	0.26	<LOQ	0.2	1.4	2.1	<LOQ	<LOQ	<LOQ
	Indian River[B]	SITK	n=12	0.70%	<LOQ	<LOQ	<LOQ	1.3	8.2	<LOQ	<LOQ	<LOQ
	Indian River dup[B]	SITK	n=12	0.90%	<LOQ	<LOQ	<LOQ	1.4	8.9	<LOQ	<LOQ	<LOQ
	Taiya River[B]	KLGO	n=8	1.50%	0.71	<LOQ	0.54	1.4	2.9	<LOQ	0.23	<LOQ
	Skagway River[B]	DS of KLGO	n=5	1.40%	0.89	<LOQ	1.9	3.8	4.1	<LOQ	0.21	<LOQ

[A] LOQ = Limit of Quantification; DL = Detection Limit (Nagorski et al. 2011).

[B] Other studied areas, also in in gray, included for reference.

Table 78 (continued). Summed concentrations of various POPs (in ng/g, wet weight) in GLBA.

Study	Stream	Location	Sample Size	% Lipid	HCB[A]	∑HCHs[A]	∑CHLDs[A]	PCBs			Dieldrin[A]	Mirex[A]
								∑DDTs[A]	∑ 40 CB[A]	∑BDEs[A]		
Johnson et al. 2005 *(Estuarine coho, yearlings)*[B]	Alsea Bay, OR[B]		n=3	1.2 ±0.1%	0.20	N/A	0.17	1.4	5.9	N/A	2.5	<DL
	Coos Bay, OR[B]		n=1	1.2 ±0.1%	0.16	N/A	0.2	1.8	14	N/A	3.3	0.64
	Grays Harbor, WA[B]		n=1	1.2 ±0.1%	0.13	N/A	0.35	3.4	27	N/A	<DL	<DL
	Willapa Bay, WA[B]		n=1	1.2 ±0.1%	0.13	N/A	0.44	0.9	6.4	N/A	<DL	<DL
	Yaquina Bay, OR[B]		n=3	1.2 ±0.1%	0.09	N/A	0.1	1.7	11	N/A	<DL	<DL
Olson et al. 2008 *(Estuarine coho, yearlings)*[B]	Commencement Bay, WA[B]		n=6	4.40%	0.63	N/A	N/A	4.2	6.7	N/A	N/A	N/A
Tallmon, unpubl. Data *(Intertidal mussels)*	GLBA avg.	GLBA	47 sites	N/A	<LOQ	<LOQ	<LOQ	<LOQ	1.0	<LOQ	<LOQ	<LOQ
	SITK avg. *(exclude harbor)*[B]	SITK	2 sites	N/A	<LOQ	<LOQ	<LOQ	0.82	9.1	<LOQ	<LOQ	<LOQ
	KLGO avg.[B]	KLGO	1 site	N/A	<LOQ	<LOQ	<LOQ	0.17	1.9	<LOQ	<LOQ	<LOQ

[A] LOQ = Limit of Quantification; DL = Detection Limit (Nagorski et al. 2011).

[B] Other studied areas, also in in gray, included for reference.

While POPs were found in all coho salmon samples from GLBA (Nagorski et al. 2011), none of the sample results exceeded the criteria for the protection of wildlife outlined by Newell et al. (2000). Most detected concentrations of POPs were one to three orders of magnitude below the recommended criteria for piscivorous wildlife (Newell et al. 2000, Nagorski et al. 2011). The persistence of POPs is amply demonstrated in GLBA in that none of the current use POPs (pesticides) were detected; the only POPs detectable were from historic POPs, most of which were banned more than 30 years ago (Nagorski et al. 2011).

Marine

Metals

The primary source of Hg in the marine environment is from atmospheric deposition and distribution by ocean currents (Sunderland et al. 2009). In GLBA, Hg may also enter the marine environment directly from glacial outwash. Mercury and other metals deposited onto the glaciers from the atmosphere are stored until the glacier melts or calves, when they are ultimately released into the marine environment (Sunderland et al. 2009).

Levels of Hg in the North Pacific Ocean have been increasing for at least two decades (Sunderland et al. 2009). The North Pacific Intermediate Waters (NPIW) current originates off the coast of Japan and travels in an easterly direction (clockwise). These waters are known to be rich in Hg and are suspected to be the source for the Hg increase in the North Pacific (Sunderland et al. 2009). If MeHg concentrations increase at the projected rate (contemporary atmospheric deposition rate), Hg concentrations in pelagic marine fish could be seriously impacted (Sunderland et al. 2009). However, a slightly lower methylation rate has been found in marine environments compared to freshwater environments (Baker et al. 2009).

Mussel samples from GLBA were analyzed for four different metals (Table 79) (Tallmon 2012). Overall, the levels of these contaminants in GLBA were very low. Arsenic (As) and cadmium (Cd) levels in GLBA mussel samples ranged between 0.26-1.80 µg/g (As) and 0.15-1.3 µg/g (Cd). These levels are far below national average for mussels of 12-22 µg/g (As) and 4-9 µg/g (Cd), as reported in Kimbrough et al. (2008). Hg in mussel samples ranged from 0.0022-0.017 µg/g and were also well below national averages for mussels of 0.18-0.35 µg/g (Kimbrough et al. 2008). Tributyltin (TBT) in GLBA mussel samples were all below the limit of quantitation.

Table 79. Metal contamination levels in mussel tissue samples from GLBA (2007, 2009, 2011) reported in µg/g (wet tissue) except TBT (ng/g) (Tallmon 2012).

Sample	Site Description	As	Cd	Hg	TBT [B]
2007					
1801606	Bartlett Cove	0.71	0.52	0.0093	<LOQ
1801607	Bartlett Cove	0.6	0.41	0.0088	<LOQ
1801608	**Bartlett Boat Ramp [A]**	0.88	0.49	0.0082	<LOQ
1801609	Ripple Cove	0.7	0.51	0.0086	NA
1801610	N Rush Point	0.67	0.63	0.0091	NA
1801611	S Whidbey Psg	0.83	0.82	0.0071	NA
1801612	N Drake Island	0.77	0.9	0.0063	<LOQ
1801613	Geikie Inlet Isl	0.46	0.4	0.0079	NA
1801614	Sebree Island	0.77	0.75	0.0057	NA
1801615	N Caroline Pt	0.89	0.9	0.0067	<LOQ
1801616	Muir Pt	0.63	0.51	0.0058	NA
1801617	N Pt George	0.73	0.74	0.0073	NA
1801618	Gateway Knob	0.68	0.55	0.0066	<LOQ
1801619	Hunters Cove	0.53	0.51	0.0065	NA
1801620	Spokane Cove	0.55	0.69	0.0065	NA
1801621	**Bartlett Fuel Dock [A]**	0.51	0.41	0.0094	<LOQ
1801622	Bartlett R Trib	0.96	0.37	0.011	NA
1801623	S. Stump Cove	1.1	0.75	0.0065	NA
1801624	Westdahl Pt	0.6	0.51	0.0057	NA
1801625	N Nunatak Cr	1	0.76	0.0074	NA
1801626	McBride Spit South	1	0.68	0.0074	NA
1801628	Tidal inlet	0.91	1.2	0.0065	<LOQ
1801629	E Russell Rocks	0.71	0.6	0.0051	NA
1801630	Russell Fan	0.77	0.53	0.0057	NA
1801631	Russell Island	1	0.58	0.007	<LOQ
1801632	N of Russell Fan	0.71	0.41	0.0071	NA
1801633	S Tarr Inlet	1.1	0.49	0.0053	NA
1801634	Tarr Inlet	1	0.47	0.0076	<LOQ
1801635	W Hazelton Camp	1.8	0.76	0.0069	NA
1801636	Blue Mouse Cove	0.56	0.37	0.0075	NA
1801638	Upper Excursion	0.55	0.44	0.0083	NA
1801640	**Excursion Fish Plant [A]**	0.47	0.6	0.0086	<LOQ
1801641	Lower Excursion	0.5	0.84	0.0046	NA

[A] Sites, also in **bold,** were selected as “hot” control sites.

[B] LOQ = Limit of Quantitation

Table 79 (continued). Metal contamination levels in mussel tissue samples from GLBA (2007, 2009, 2011) reported in µg/g (wet tissue) except TBT (ng/g) (Tallmon 2012).

Sample	Site Description	As	Cd	Hg	TBT [B]
2007 (continued)					
1801642	NE Pleasant Island	0.57	0.4	0.0046	NA
1801643	E Carolus Riv	0.63	0.5	0.0081	NA
1801645	W Carolus	0.6	0.73	0.0067	<LOQ
1801646	W Pt Dundas	0.49	0.53	0.0067	NA
1801648	W Dundas Bay	0.49	0.3	0.0068	NA
1801649	**Outer Elfin Cove [A]**	0.75	0.47	0.01	<LOQ
1801650	Mouth Rush Pt Cr	0.56	0.61	0.0071	NA
1801701	Graves	1.1	0.9	0.0097	NA
1801702	Torch Bay N	0.71	1	0.01	<LOQ
1801703	Dixon Harbor	0.8	0.73	0.011	NA
1801704	Lituya Bay	0.52	0.39	0.0057	NA
1801708	Berg Bay	0.51	0.37	0.0075	<LOQ
2009					
200901	E Russell Rocks	0.26	0.15	0.0025	<LOQ
200902	W Hazelton Camp	0.52	0.18	0.0022	<LOQ
200903	Ripple Cove	0.53	0.26	0.0023	<LOQ
200904	Bartlett Cove	1.1	0.75	0.0084	<LOQ
2011					
201101	E Russell Rocks	1.8	1	0.017	<LOQ
201102	W Hazelton Camp	1.3	1.2	0.012	<LOQ
201103	Ripple Cove	1.4	1.3	<LOQ	<LOQ
201104	Bartlett Cove	1.1	1.3	0.01	<LOQ

[A] Sites, also non-year values in **bold,** that were selected as "hot" control sites.

[B] LOQ = Limit of Quantitation

POPs

POPs in the marine environment generally have the same sources and properties as POPs in the freshwater environment. Atmospheric deposition, facilitated by ocean current distribution, is the primary method by which POPs are deposited in the GLBA environment (Wania and Mackay 1996, Eckert et al. 2006). In GLBA, POPs may also enter the marine environment directly from glacial outwash. POPs deposited onto the glaciers from the atmosphere are stored until the glacier melts or calves, when they are ultimately released into the marine environment (Bogdal et al. 2010).

Based on mussel samples collected at GLBA, POP levels in GLBA are considered very low (Table 80; Tallmon 2012). POP levels in GLBA mussels were below samples from other parts of the U.S. (Kimbrough et al. 2008, Tallmon 2012) and well within seafood standards set by the National Academy of Sciences (NAS 1991) (2,000 ng/g for ∑PCB, 5,000 ng/g for ∑DDT, and 300 ng/g for

∑CHLD). The low levels of POPs that were detected indicate that these contaminants do not pose an immediate threat to GLBA marine waters.

Table 80. POPs and PAHs from mussel samples (ng/g) in or near GLBA (Tallmon 2012). POP abbreviations are: CHLD = chlordanes, DDT = dichloro diphenyl trichloroethanes, HCH = hexachlorocyclohexanes, PCB = polychlorinated biphenyls, PBDE = polybrominated biphenyl ethers.

Sample	Site Description	TPAH[B]	∑CHLD[B]	∑DDT[B]	∑HCH[B]	∑PCB[B]	∑PBDE[B]
2007							
1801603	Berg Bay	NA	<LOQ [B]	<LOQ	<LOQ	1	<LOQ
1801604	Berg Bay	NA	<LOQ	<LOQ	<LOQ	1.3	<LOQ
1801605	Berg Bay	NA	<LOQ	<LOQ	<LOQ	1.3	<LOQ
1801606	Bartlett Cove	<LOQ	<LOQ	<LOQ	<LOQ	1.4	<LOQ
1801607	Bartlett Cove	<LOQ	<LOQ	<LOQ	<LOQ	0.62	<LOQ
1801608	**Bartlett Boat Ramp [A]**	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ
1801609	Ripple Cove	<LOQ	<LOQ	<LOQ	<LOQ	0.77	<LOQ
1801610	N Rush Point	<LOQ	<LOQ	<LOQ	<LOQ	0.69	<LOQ
1801611	S Whidbey Psg	<LOQ	<LOQ	<LOQ	<LOQ	0.79	<LOQ
1801612	N Drake Island	<LOQ	<LOQ	<LOQ	<LOQ	0.64	<LOQ
1801613	Geikie Inlet Isl	<LOQ	<LOQ	<LOQ	<LOQ	0.65	<LOQ
1801614	Sebree Island	<LOQ	<LOQ	<LOQ	<LOQ	0.7	<LOQ
1801615	N Caroline Pt	<LOQ	<LOQ	<LOQ	<LOQ	0.14	<LOQ
1801616	Muir Pt	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ
1801617	N Pt George	<LOQ	<LOQ	<LOQ	<LOQ	0.72	<LOQ
1801618	Gateway Knob	<LOQ	<LOQ	<LOQ	<LOQ	0.78	<LOQ
1801619	Hunters Cove	<LOQ	<LOQ	<LOQ	<LOQ	0.65	<LOQ
1801620	Spokane Cove	NA	<LOQ	<LOQ	<LOQ	0.67	<LOQ
1801621	**Bartlett Fuel Dock [A]**	1488.3	<LOQ	<LOQ	<LOQ	2.2	<LOQ
1801622	Bartlett R Trib	<LOQ	<LOQ	<LOQ	<LOQ	1	<LOQ
1801623	S Stump Cove	<LOQ	<LOQ	<LOQ	<LOQ	1.7	<LOQ
1801624	Westdahl Pt	<LOQ	<LOQ	<LOQ	<LOQ	1.5	<LOQ
1801625	N Nunatak Cr	<LOQ	<LOQ	<LOQ	<LOQ	1.1	<LOQ
1801626	McBride Spit S	<LOQ	<LOQ	<LOQ	<LOQ	0.72	<LOQ
1801627	*McBride Spit S [C]*	<LOQ	NA	NA	NA	NA	NA
1801628	Tidal inlet	<LOQ	<LOQ	<LOQ	0.18	0.87	<LOQ
1801629	E Russell Rocks	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ
1801630	Russell Fan	<LOQ	<LOQ	<LOQ	<LOQ	0.78	<LOQ
1801631	Russell Island	<LOQ	<LOQ	<LOQ	<LOQ	1	<LOQ

[A] Sites, also non-year values in **bold,** that were selected as “hot” control sites

[B] LOQ = Limit of Quantitation

[C] Sites, also in *italic*, that were sediment samples

Table 80 (continued). POPs and PAHs from mussel samples (ng/g) in or near GLBA (Tallmon 2012). POP abbreviations are: CHLD = chlordanes, DDT = dichloro diphenyl trichloroethanes, HCH = hexachlorocyclohexanes, PCB = polychlorinated biphenyls, PBDE = polybrominated biphenyl ethers.

Sample	Site Description	TPAH[B]	∑CHLD[B]	∑DDT[B]	∑HCH[B]	∑PCB[B]	∑PBDE[B]
2007 (continued)							
1801632	N Russell Fan	<LOQ	<LOQ	<LOQ	0.21	1.2	<LOQ
1801633	S Tarr Inlet	<LOQ	NA	NA	NA	NA	NA
1801634	Tarr Inlet	<LOQ	<LOQ	<LOQ	<LOQ	0.66	<LOQ
1801635	W Hazelton Camp	<LOQ	<LOQ	<LOQ	<LOQ	0.66	<LOQ
1801636	Blue Mouse Cove	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ
1801637	*Blue Mouse Cove* [C]	3.59	NA	NA	NA	NA	NA
1801638	Upper Excursion	<LOQ	<LOQ	<LOQ	<LOQ	0.54	<LOQ
1801639	*Upper Excursion* [C]	6.94	NA	NA	NA	NA	NA
1801640	**Excursion Fish Plant** [A]	13.55	0.45	0.25	<LOQ	1.8	<LOQ
1801641	Lower Excursion	<LOQ	<LOQ	<LOQ	<LOQ	0.79	<LOQ
1801642	NE Pleasant Island	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ
1801643	E Carolus Riv	<LOQ	<LOQ	<LOQ	<LOQ	1	<LOQ
1801644	*E Carolus Riv* [C]	<LOQ	NA	NA	NA	NA	NA
1801645	W Carolus	<LOQ	<LOQ	<LOQ	<LOQ	1.1	<LOQ
1801646	W Pt Dundas	<LOQ	<LOQ	<LOQ	<LOQ	1.1	<LOQ
1801647	*W Pt Dundas* [C]	<LOQ	NA	NA	NA	NA	NA
1801648	W Arm Dundas	<LOQ	<LOQ	<LOQ	<LOQ	1.2	<LOQ
1801649	**Outer Elfin Cove** [A]	69.74	<LOQ	0.48	<LOQ	3.7	6.3
1801650	Mouth Rush Pt Cr	<LOQ	<LOQ	<LOQ	<LOQ	0.77	<LOQ
1801701	Graves	<LOQ	<LOQ	<LOQ	<LOQ	0.65	<LOQ
1801702	Torch Bay N	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ
1801703	Dixon Harbor	<LOQ	<LOQ	<LOQ	<LOQ	0.84	<LOQ
1801704	Lituya Bay	NA	<LOQ	<LOQ	<LOQ	1.1	<LOQ
1801708	Berg Bay	137.66	NA	NA	NA	NA	NA
2009							
200901	E Russell Rocks	0.83	<LOQ	<LOQ	<LOQ	0.36	<LOQ
200902	W Hazelton Camp	1.09	<LOQ	<LOQ	<LOQ	0.33	<LOQ
200903	Ripple Cove	0.48	<LOQ	<LOQ	<LOQ	0.35	<LOQ
200904	Bartlett Cove	0.78	<LOQ	<LOQ	<LOQ	0.36	<LOQ

[A] Sites, also non-year values in **bold,** that were selected as “hot” control sites

[B] LOQ = Limit of Quantitation

[C] Sites, also in *italic*, that were sediment samples

Table 80 (continued). POPs and PAHs from mussel samples (ng/g) in or near GLBA (Tallmon 2012). POP abbreviations are: CHLD = chlordanes, DDT = dichloro diphenyl trichloroethanes, HCH = hexachlorocyclohexanes, PCB = polychlorinated biphenyls, PBDE = polybrominated biphenyl ethers.

Sample	Site Description	TPAH[B]	∑CHLD[B]	∑DDT[B]	∑HCH[B]	∑PCB[B]	∑PBDE[B]
2011							
201101	E Russell Rocks	0.34	<LOQ	<LOQ	0.32	<LOQ	0.16
201102	W Hazelton Camp	0.3	<LOQ	<LOQ	0.33	0.32	<LOQ
201103	Ripple Cove	0.84	<LOQ	<LOQ	0.41	<LOQ	<LOQ
201104	Bartlett Cove	0.67	<LOQ	<LOQ	0.29	<LOQ	0.5

[A] Sites, also non-year values in **bold,** that were selected as “hot” control sites

[B] LOQ = Limit of Quantitation

[C] Sites, also in *italic*, that were sediment samples

PAHs

Polycyclic aromatic hydrocarbons have been detected at very low levels in GLBA (Table 80). One notable exception occurred in a 2007 mussel sample from Berg Bay, with a PAH level of 137 ng/g (Tallmon 2012). Possible explanations for this elevated level include recent boating activity and/or the low level of seawater exchange between Berg Bay and GBP (Tallmon 2012). Most samples where PAHs were detectable were from selected control sampling locations or “hot” spots of human activity where the contaminants were anticipated (Tallmon 2012). The highest level of PAHs was detected at the Bartlett fuel dock control site, where a level of 1,488 ng/g was recorded. It should be noted for reference that this high reading falls in the “medium” range compared to the rest of the U.S., as reported in Kimbrough et al. (2008).

Even though PAHs currently occur at very low concentrations in GLBA, their potential impacts are still a concern. GLBA is visited by cruise ships, commercial fishing boats, and other small watercraft. In addition to this traffic, a boat dock with fueling station exists in Bartlett Cove. The potential for contamination through an accident must be considered. Tallmon (2012) reported that a small cruise ship ran aground during their survey, but did not produce a large spill.

Ocean Acidification

Due to the high latitude of GLBA, its waters are more vulnerable to OA than marine environments at lower latitudes (Reisdorph and Mathis 2014). This is in part due to high latitude oceans having naturally lower concentrations of calcium carbonate. The reduced amount of calcium carbonate ions in the water translates to a reduced ability to neutralize the CO_2 and less buffering capacity in the water to maintain pH (Mathis et al. 2014). Additionally, the large seasonal variations in nutrients, sunlight, and freshwater input (Reisdorph and Mathis 2014) in GLBA can impact vulnerability.

Atmospheric CO_2 has increased from approximately 280 ppm during preindustrial times to 396 ppm in 2013 (NOAA 2014). With the oceans absorbing more than 25% of all atmospheric CO_2 from anthropogenic sources (Mathis et al. 2014), this CO_2 increase correlates to a decrease of approximately 0.1 units of pH since preindustrial times (Byrne et al. 2009). A further decrease of 0.1-0.5 pH units is predicted over the next century (Orr et al. 2005, Reisdorph and Mathis 2014).

GLBA is also more susceptible to OA as it has seven tidewater glaciers that directly add freshwater into the marine environment (Reisdorph and Mathis 2014). This dilution leaves the marine environment more vulnerable to the effects of atmospheric CO_2 absorption than other areas. The historic annual glacial melt coupled with the current deglaciation that GLBA is experiencing (Loso et al. 2014) has increased the direct freshwater input in recent years.

The influx of freshwater directly from the glacier is naturally lower in total alkalinity (TA) and reduces the normal buffering capacity of the marine environment (Reisdorph and Mathis 2014). The meltwater's direct entry to the marine environment from tidal glaciers does not allow the runoff to flow across the landscape and interact with the carbonate bedrock, which would increase its alkalinity and buffering capacity (Brown 2002). Reisdorph and Mathis (2014) demonstrated that increased atmospheric deposition of CO_2 alone was not responsible for the OA in GLBA waters. Decreased TA concentrations from glacial melt were determined to be the major influence during the months with the highest glacial meltwater (summer-fall) (Reisdorph and Mathis 2014).

Reisdorph and Mathis (2014) utilized observed aragonite saturation states from GLBA and the current rate of CO_2 increase in the atmosphere (~1.5 ppm yr^{-1}) to illustrate a likely future scenario concerning OA within GLBA (Table 81). The saturation levels of aragonite highlight the risk to mollusks and other invertebrates that rely on this mineral's availability for shell building. When the aragonite saturation level drops below 1.0 (due to more acidic water), the mineral will dissolve; when its saturation state is above 1.0, the mineral will be formed. Based on these projections, ocean acidification will become severe enough in 100-150 years that aragonite saturation rates in GLBA will be below 1.0 for most of the year (Reisdorph and Mathis 2014).

Table 81. Seasonal observed and projected (based on 2011-2012 observations) average aragonite saturation states (ΩA) for GLBA (Reisdorph and Mathis 2014). Values below 1.0, meaning aragonite will dissolve, are in **bold**.

Season	Obs. ΩA	ΩA 50 yr	ΩA 100 yrs	ΩA 150 yrs
Summer 2011	1.38 ± 0.59	1.13 ± 0.42	**0.97 ± 0.33**	**0.85 ± 0.27**
Fall 2011	**0.89 ± 0.09**	**0.79 ± 0.08**	**0.72 ± 0.07**	–
Winter 2012	1.19 ± 0.24	1.04 ± 0.18	**0.92 ± 0.14**	**0.83 ± 0.11**
Spring 2012	1.88 ± 0.62	1.52 ± 0.38	1.28 ± 0.26	1.11 ± 0.19
Summer 2012	1.51 ± 0.63	1.25 ± 0.48	1.07 ± 0.38	**0.94 ± 0.32**

Threats and Stressor Factors

Threats and stressors to GLBA water quality include long-range transport of airborne contaminants, vessel traffic, point source pollution, and climate change (Eckert et al. 2006). The majority of airborne contaminants are carried to Alaska via long-range atmospheric pathways from areas as far away as eastern Asia (Jaffee et al. 1999, Pacyna and Pacyna 2002). Levels of mercury and POPs have shown significant increases in Alaska within the last few decades, with detectable levels found in the fat of a variety of mammals and have caused eggshell thinning in raptors (Pacyna and Pacyna 2002). Due to the world-wide atmospheric deposition of some contaminants, glaciers have become a pollution sink (Bogdal et al. 2010). As the glacial ice melts and releases these contaminants, the glaciers themselves become a source of pollution (Bogdal et al. 2010).

Vessel traffic can impact water quality, due to the risk of contaminant exposure through chemical spills (e.g., fuel, oils, hydraulic fluid) or wastewater discharge (NPS 1998, Wuebben et al. 2000). Many fuels and other oils are persistent chemicals, meaning the contamination could be transported long distances by wind and/or currents (Eley 2000). The NPS maintains a vessel fueling and underground petroleum storage facility at Bartlett Cove near the public dock, where fuel spills could occur and threaten the park's water quality (Wuebben et al. 2000).

As mentioned previously, biological transport of pollutants is also a concern for GLBA. Fish, birds, and even insects can play a role in the incidental transport of contaminants into foreign environments (Blais et al. 2007, Mogren et al. 2013). Salmon are a special concern for GLBA, as most streams experience spawning runs annually (Moynahan et al. 2008). Salmon develop as much as 95% of their biomass in the marine environment and ultimately deposit this biomass back into the freshwater environment where their carcasses accumulate after spawning (Quinn 2005, Baker et al. 2009).

Fish processing waste may also pose a threat to GLBA, as a seafood processing facility operates at Excursion Inlet during the summer months, employing as many as 500 people (Eckert et al. 2006). The outfall of fish processing waste is approximately 0.5 km (0.3 mi) from the NPS mid-channel boundary in Excursion Inlet. Waste is discharged into 18 m (60 ft) of water, but on a benthic slope that quickly reaches 30 m (100 ft) depth within 20-40 m (65-130 ft) from the outfall (Soiseth, written communication, 22 October 2015). According to data reported to the AK DEC by the processor as part of the company's APDES permit requirements (General Permit #AK-G52-0059), 1.6-6.1 million kg of waste were discharged annually between 2001 and 2014 (unpublished data from Chad Soiseth, 22 October 2015). From 2010-2014, mean annual waste discharge was lower than during the early 2000s, with a 5-year average of 3.2 million kg; however, discharge varied greatly between years, from 2.1 million kg or less in 3 of the years to 5.8 million kg in 2013 (unpublished data). The impacts from this waste have not been studied in detail.

Climate change may impact GLBA's water quality, due to the influence of water temperature on water chemistry. GLBA's climate is predicted to become warmer over the next century, with the greatest temperature increases occurring in the winter (SNAP et al. 2009). Warmer air temperatures could increase water temperatures, which would impact several water quality parameters. For example, warmer waters hold less dissolved oxygen (necessary for aquatic organisms) than cooler waters (USGS 2010). Increased water temperatures can also enhance the toxicity of metals, including the conversion of Hg to MeHg (USGS 2010, van der Velden 2013). Since 2010, SEAN has monitored the Salmon River in GLBA for hourly measurements of water temperature, dissolved oxygen, specific conductance, and pH (Sergeant and Johnson 2015). While no Salmon River water quality trends have been detected to date, this long-term monitoring will allow future researchers to assess watershed changes caused in part by climate change (Sergeant, written communication, 6 November 2015). Lastly, a warmer climate is likely to accelerate glacial melting, which could introduce pollutants stored in glaciers into GLBA's fresh and marine waters (Bogdal et al. 2009).

Data Needs/Gaps

Long-term monitoring programs for water quality and marine contaminants (e.g., Hg, POPs, and PAHs) have been in place for several years at GLBA (Sergeant, written communication, 16 October 2015). A freshwater contaminants protocol is currently under development and will be completed by the end of 2017 (Sergeant, written communication, 6 November 2015). As data accumulates over time through this monitoring, it will allow managers to identify any changes or trends in contaminant levels. Continued water quality monitoring of the Salmon River by SEAN (as described in Sergeant and Johnson 2015) should help detect any changes in water temperature and chemistry related to climate change. Water temperature data from GBP collected as part of an ongoing oceanographic monitoring program (Danielson 2012) could also detect water temperature changes in the park's marine waters. Further study of the potential threats to water quality from contaminants stored in GLBA's glaciers may help managers understand the likely impacts of a warming climate. A repetition of the Reisdorph and Mathis (2014) study of aragonite saturation states in several years could provide insight into the rate or severity of OA in GBP.

Overall Condition

Freshwater

Mercury (Hg)

The project team defined the *Significance Level* for Hg as a 3. Current Hg concentrations in sampled GLBA streams were three to four orders of magnitude below the EPA thresholds for aquatic organisms and human health (Nagorski et al. 2011).Therefore, Hg is of low concern (*Condition Level* = 1). Research suggests that distant developing countries are the source of Hg contamination in Alaska, meaning these sources are beyond the control of the state and the national parks (Landers et al. 2008).

Methylmercury (MeHg)

The project team defined the *Significance Level* for MeHg as a 3. In GLBA, wetlands play an important role in the conversion of inorganic Hg to the more toxic MeHg (Eckert et al. 2006, Wiener et al. 2006). While the prevalence of wetlands in some areas of GLBA is a concern with regard to MeHg, the overall levels of Hg in park samples are well below EPA levels of concern for human health or for aquatic organisms. Therefore, MeHg is currently of low concern (*Condition Level* = 1).

Persistent Organic Pollutants (POPs)

The *Significance Level* for POPs was defined as a 3. Historically used POPs (banned in the U.S. for more than 30 years) were detected in salmon tissue collected from numerous GLBA streams (Nagorski et al. 2011). Still, the overall levels of POPs detected in GLBA were low and fell one to three orders of magnitude below the criteria recommended for the protection of piscivorous wildlife (Newell et al. 2000, Nagorski et al. 2011). Therefore, POPs are of low concern (*Condition Level* = 1).

Marine

<u>*Metals*</u>

The project team defined the *Significance Level* for metal contaminants as a 3. Mussel samples from GLBA intertidal zones showed very low levels of the four metals analyzed, particularly in comparison to other locations in the U.S. (Table 79; Kimbrough et al. 2008, Tallmon 2012). As a result, metals are currently of low concern (*Condition Level* = 1).

<u>*Persistent Organic Pollutants (POPs)*</u>

The *Significance Level* for POPs was defined as a 3. Historic use of POPs were found in all GLBA marine sediments (Nagorski et al. 2011). However, POP levels in GLBA mussel samples are considered very low and are below levels found in other parts of the U.S. (Table 80; Kimbrough et al. 2008, Tallmon 2012). No POP levels exceeded the criteria for the protection of human health or wildlife (Nagorski et al. 2011). Based on this, POPs are of low concern (*Condition Level* = 1).

<u>*Polycyclic Aromatic Hydrocarbons (PAHs)*</u>

PAHs were assigned a *Significance Level* of 3. While PAHs have been detected in GLBA, they have generally been at very low levels (Tallmon 2012, Table 80). Notable exceptions occurred in 2007 mussel samples from Berg Bay and the Bartlett fuel dock, where PAH levels were well above all other GLBA samples (Tallmon 2012). However, even the highest GLBA sample falls in the "medium" range identified for the U.S. as a whole (Kimbrough et al. 2008, Tallmon 2012).With this in mind, PAHs are also considered of low concern (*Condition Level* = 1).

<u>*Ocean Acidification (OA)*</u>

The project team defined the *Significance Level* for OA as a 3. Ocean acidification is a direct risk to Alaska's food chain, as it threatens all marine organisms that use calcium carbonate (e.g., aragonite) to build and maintain their shells or skeletons (Reisdorph and Mathis 2014). Reisdorph and Mathis (2014) modeled a likely future scenario concerning OA within GLBA showing that ocean acidification will become severe enough in 100-150 years that aragonite will be more likely to dissolve than to form shells. However, the current concern regarding impacts of OA in GLBA is low (*Condition Level* = 1).

<u>Weighted Condition Score</u>

The *Weighted Condition Score* for water quality was determined to be 0.33, indicating that this component is currently of low concern. The trend arrow assigned to this measure represents an unchanged condition. While the current condition of this component is of low concern, water quality remains a critically important resource due to the tremendous amount of marine and freshwater reserves present at the park.

Water Quality				
Measures		Significance Level	Condition Level	WCS = 0.33
Freshwater	Mercury	3	1	
	Methylmercury	3	1	
	Persistent Organic Pollutants	3	1	
Marine	Metals	3	1	
	Persistent Organic Pollutants	3	1	
	Polycyclic Aromatic Hydrocarbons	3	1	
	Ocean Acidification	3	1	

4.22.6. Sources of Expertise

- Chris Sergeant, SEAN Ecologist
- Chad Soiseth, GLBA Fisheries Biologist

4.22.7. Literature Cited

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4.23. Underwater Soundscape

4.23.1. Description

The definition of soundscape in a national park is the total ambient sound level of the park, comprised of both natural ambient sound and human-made sounds (NPS 2000). The NPS's mission is to preserve natural resources, including natural soundscapes, associated with the national park units. Intrusive sounds are of concern to park visitors, as they detract from their natural and cultural resource experiences (NPS 2000). In addition, vessel traffic or other human-caused noise sources can interrupt interpretive programs being held within a park. According to a survey conducted by the NPS, many visitors come to national parks to enjoy, equally, the natural soundscape and natural scenery (NPS 2000).

Noise not only affects visitor experience, it can also alter the behavior of wildlife. Repeated noise can cause chronic stress to animals, possibly affecting their energy use, reproductive success, and long-term survival (Radle 2007). Many factors affect how visitors and wildlife perceive and respond to noise. Primary acoustical factors include the loudness, frequency (i.e., pitch), and duration of the noise. Non-acoustical factors, such as climate, vegetation, topography, and individual hearing sensitivity also play a role in how visitors and wildlife respond to noise (Mestra Greve Associates, DLB 2014).

Underwater soundscape has been classified as a Vital Sign in the SEAN, including GLBA (Moynahan et al. 2008). Preserving the underwater soundscapes as natural and undisturbed as possible is an objective for GLBA because nearly all visitations occur by motorized vessels, and vessels produce underwater noise that decreases the distance over which marine animals can communicate and detect predators and prey. Humpback whales, harbor seals, and other marine mammals produce sounds for vital life functions essential to survival and reproduction. Fish and invertebrates also depend on the natural soundscape at a vital life history phase (Stanley et al. 2010, Vermeij et al. 2010). For example, some larval fish and invertebrate species use reef sounds to navigate to suitable habitat (Simpson et al. 2004). While these biological phenomena have not been demonstrated for species in Glacier Bay, underwater sound monitoring has detected sounds produced by a variety of species, known and unknown.

Underwater soundscape varies greatly seasonally and diurnally. Similar to terrestrial soundscapes, underwater soundscapes are contributed to by natural and anthropogenic sources. Sources vary from wind-generated surface noise to humpback whale songs to marine vessel noise (Kipple and Gabriele 2007). It should be noted that sound travels about 4.4 times faster and attenuates more slowly through water as compared to air. In addition, the decibel scale that is used for in-air sounds is different than that used for underwater sounds (Kipple and Gabriele 2007). The intensity level of an underwater sound is lower than that of an in-air sound with the same dB value. Wind and rain are the primary underwater sound sources in Glacier Bay, as with most marine environments. Biological sources (e.g., humpback whales, harbor seals) are important contributors to the natural soundscape at GLBA. The primary manmade noise in the park is marine vessel traffic (Kipple and Gabriele 2007).

4.23.2. Measures

- Underwater natural ambient sound level (dBa) at frequencies of interest
- Proportion of hourly samples without vessel noise
- Duration of noise-free intervals
- Masking index of communication space for vocal species

4.24.3 Reference Conditions/Values

The reference condition for underwater soundscape is that of the historic condition of the area prior to park establishment and vessel traffic. Marine vessel traffic is essential for visitor use and park administration in order to access GLBA, but it degrades the acoustic habitat for marine species. To meet its legal obligations and protect marine resources from adverse effects, the NPS actively manages vessel traffic during GLBA visitor and shoulder seasons (May-September). Historic ambient noise is thought to have had quiet periods throughout the day, which is the natural condition for marine creatures (McKenna et al. 2014). Figure 178 displays ambient noise based on a 1-year study in GLBA in 2011. Figure 178 displays the 5th, 25th, 50th, 75th and 95th percentile frequency distribution statistics for sound level as a function of frequency averaged over the entire year. The 5th percentile represents the quietest conditions in GLBA throughout the study, and may serve as a reference for historic ambient noise conditions (Gabriele, written communication, 5 May 2015). Historic proportion of noise-free hourly samples is 100%, and the duration of noise-free intervals is perpetual. The reference condition for the masking index of communication space for vocal species is currently unavailable.

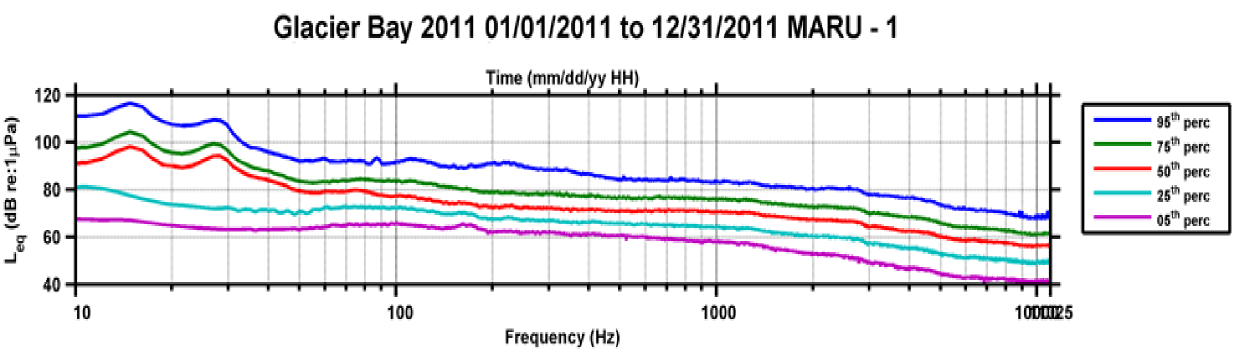

Figure 178. Ambient noise based on a 1-year study period from 1 January through 30 December 2011. This figure shows the 5th, 25th, 50th, 75th and 95th percentile frequency distribution statistics for sound level as a function of frequency averaged over the entire year (Excerpt from Clark et al. *In Prep*). The 5th percentile in this figure represents the quietest conditions in GLBA during Clark et al. (*In Prep*) and serves as the reference condition for historic ambient noise conditions.

4.23.4. Data and Methods

Kipple and Gabriele (2003) conducted an underwater soundscape study in GLBA between 2000 and 2002. This 2-year study was the beginning of an ongoing project to characterize the underwater acoustic environment in the park. GLBA staff and the Naval Surface Warfare Center Detachment (NSWCD) worked together to record and analyze underwater acoustics by using a hydrophone. In May 2000, the hydrophone was installed and anchored to the sea floor (50 m [164 ft]) just south of the entrance to Bartlett Cove (Figure 179; Photo 57). It should be noted that in May 2001 the hydrophone was replaced and relocated just east of the previous location in 30 m (99 ft) of water. A shore-based data acquisition system was linked to the hydrophone to record continuous 30-second noise samples every hour throughout a 24-hour time period for 20 months. During this study, soundscape data were collected between August 2000 and August 2002. Trends (e.g., long-term, short-term, seasonal) were observed and noise sources were identified. Other data attributes included contributions, types, prevalence, and frequency of occurrence.

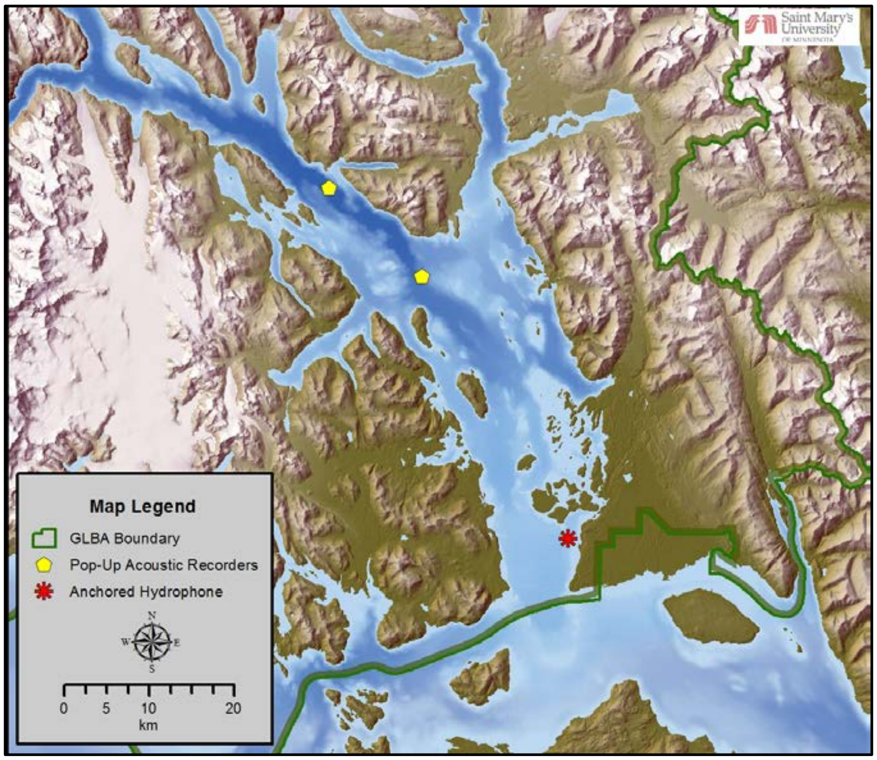

Figure 179. Location of anchored hydrophone in Bartlett Cove and the two autonomous “pop-up” acoustic recorders in the Upper Bay (Kipple and Gabriele 2003, Gabriele et al. 2010, McKenna et al. 2014).

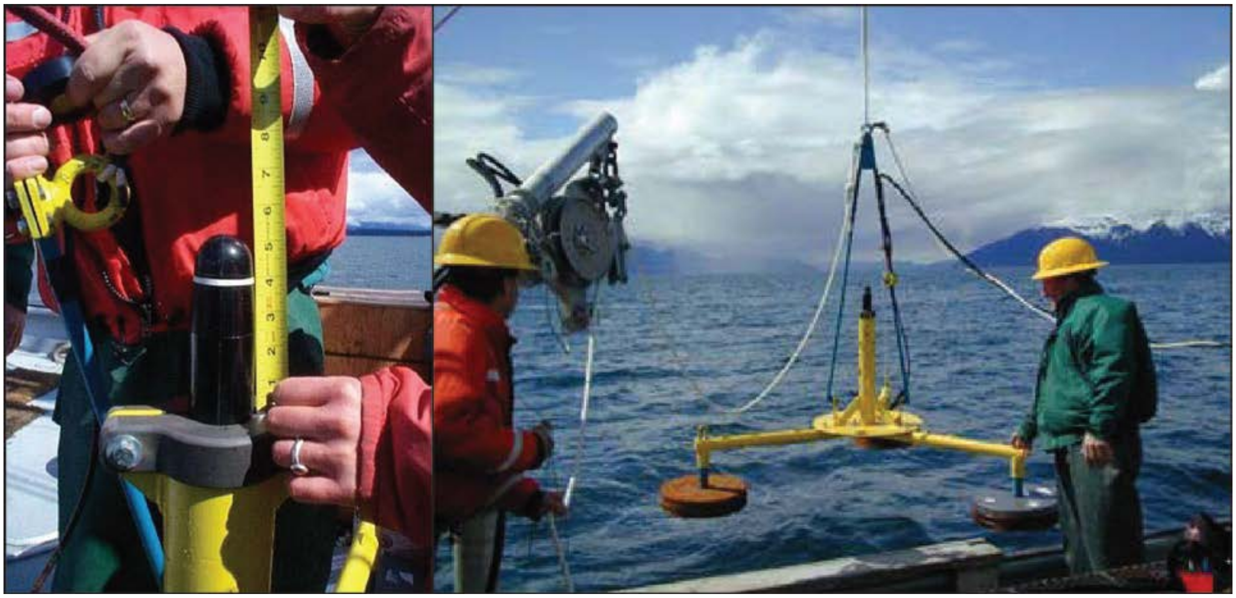

Photo 57. Noise monitoring hydrophone being relocated and installed at GLBA in May 2001 (Kipple and Gabriele 2003).

Gabriele et al. (2010) summarized underwater soundscape study efforts in GLBA between 2000-2002 and 2007-2008. This study was conducted to determine the underwater sound environment and the effects of cruise ship noise on humpback whales. Continuous data were collected using the anchored hydrophone (used in the previous studies) and two autonomous “pop-up” acoustic recorders. Data from the “pop-up” acoustic recorders were only collected for 45 days (22 June – 6 August) in 2007 (Figure 179; Photo 57). Methods used included calibrated measurements of individual vessels, ambient noise recording, and modeling to predict the effects of vessel quotas and speeds.

Gabriele et al. (2015) calculated metrics of the acoustic environment in Glacier Bay Proper to determine the reduction in communication space due to aggregate vessel traffic. Humpback whales and harbor seals were the focus of this analysis. Using empirical data on the acoustic environment (soundscape) collected at GLBA, a computer model was used to demonstrate the percent reduction in animal communication space during three levels (low [21 vessels], moderate [28 vessels], and high [34 vessels]) of vessel traffic. Calibrated noise signatures were used to determine the distance vessel noise travels (Gabriele et al. 2015). Distance traveled by humpback whale and harbor seal vocals was estimated using best available information (Gabriele et al. 2015). Proxy tracks traveled by vessels were created using GIS and Automatic Identification System (AIS), tracks were based on known destinations and speed capabilities of vessels (Gabriele et al. 2015). Humpback whale and harbor seal distributions were determined based on visual survey data. These distributions were then moved based upon behavioral parameters from scientific literature and professional judgment (Gabriele et al. 2015).

McKenna et al. (2014) compiled underwater acoustics data in the park from 2000-2002 and 2007-2008 to compare ambient acoustic condition changes. The study used data from the cabled hydrophone system that was anchored to the ocean floor near Bartlett Cove in 30 m (99 ft) of water (Figure 179; Photo 57). A shore-based data acquisition system was linked to the hydrophone to record continuous 30-second noise samples every hour throughout a 24-hour time period. For each acoustic sample, one-third octave sound pressure levels (SPL) (10-31.5 kHz) were archived. The 30-second audio clips and one-third octave SPLs were reviewed by a trained acoustic analyst for the presence of biological, physical, human-made sources, and any system noise.

4.23.5. Current Condition and Trend

<u>Underwater Natural Ambient Sound Level (dB) at Frequencies of Interest</u>
Kipple and Gabriele (2003) used 10-31.5 kHz as the frequency of interest for their study. Natural sounds include wind, rain, and biological noises such as whale whoops, grunts, squeaks, and songs. The most prevalent natural ambient sound was wind-generated noise, and wind noises ranged from 67-102 dB, with an average sound level of 84 dB (Figure 180). On average, rain sound levels generated louder and higher frequency (16 kHz) noise than wind (1 kHz); rainfall noise ranged from 91-110 dB. The most prevalent source of biological ambient sound were harbor seals; their calling peaked in July and comprised >85% of acoustic sampling. Humpback whale songs, the next most common, comprised about 10% of acoustic samples across all years, and peaked during September at 12%. Killer whale sounds were also documented, although detected infrequently.

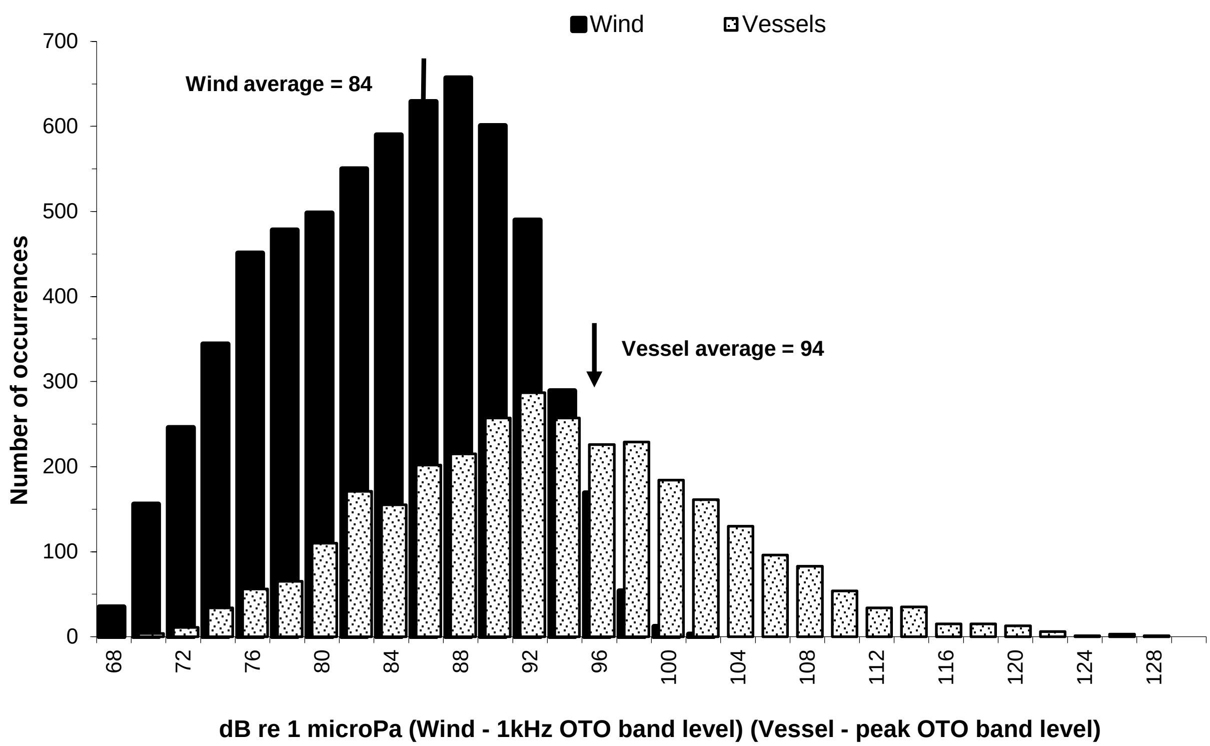

Figure 180. Distribution of underwater sound levels (wind, vessel) in GLBA between 2000 and 2002 (Kipple and Gabriele 2003, 2007).

McKenna et al. (2014) also used 10-31.5 kHz as the frequency of interest for their study. Wind noise was the dominating natural ambient sound being heard 28-38% of the time from May-September. Mean sound pressure levels during wind-dominated periods averaged 83.5 dB and ranged from 70-100 dB. Another natural ambient sound heard was rain which was present in 8-14% of the acoustic samples. Rain noise had a little higher frequency then wind, with a mean sound pressure level at 88.8 dB and a range between 70-100 dB. During the visitor season, harbor seals dominated the acoustic environment, particularly in the month of July (heard at a level of 155 dB). The occasional sound of humpback whales could be heard in roughly 10% of the samples, with sound ranging from 80-85 dB.

Proportion of Hourly Samples without Vessel Noise

Kipple and Gabriele (2003) documented the proportion of hourly samples without vessel noise and proportions of seasonal samples without vessel noise by season. The winter season had the highest percent of ambient noise samples without vessel noise (approximately 89%). Percent of samples without vessel noise ranged from 58-62% in spring and fall, respectively (Figure 181). The highest amount of vessel noise occurred in the summer months (visitor season), and approximately 41% of the samples were without vessel noises in the summer.

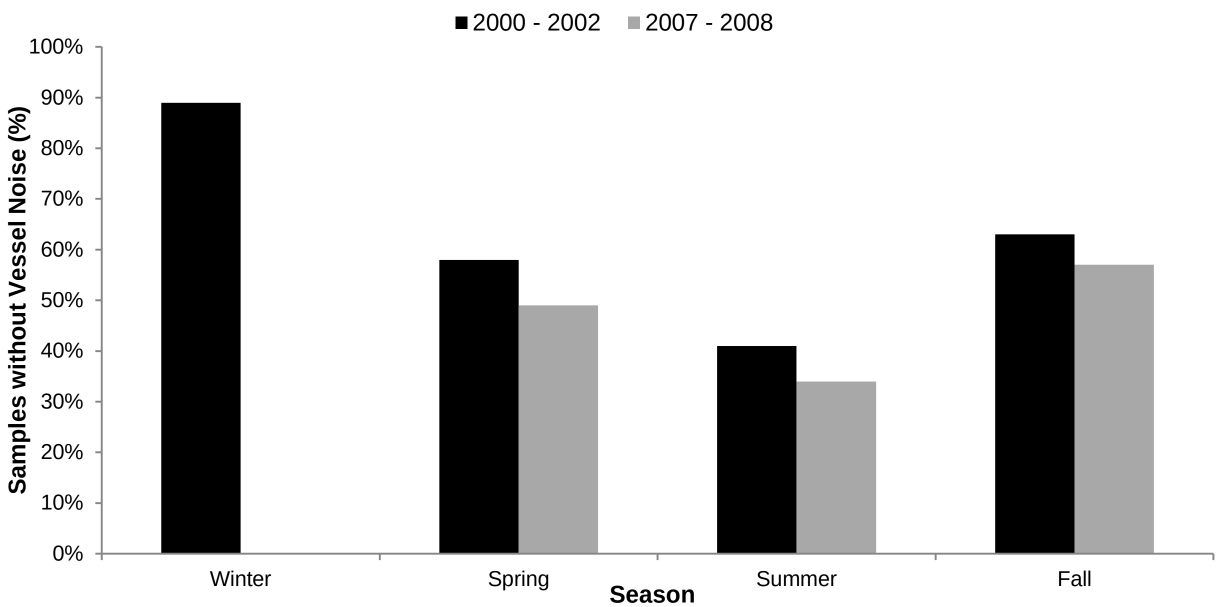

Figure 181. Proportion of ambient noise samples without vessel noise by season (Kipple and Gabriele 2003, Gabriele et al. 2010).

Gabriele et al. (2010) documented the proportion of ambient noise without vessel noise by three of the four seasons for 2007 and 2008. It should be noted that proportions were not calculated for the winter season in 2007 and 2008.

The season with the highest proportion of ambient noises without vessel noise was fall (approximately 57% of all samples; Figure 181). The highest proportion of vessel noise occurred in the summer months (visitor season), as approximately 34% of the samples were without vessel noises.

McKenna et al. (2014) documented that the overall occurrence of vessel noise in lower Glacier Bay increased significantly from 51% to 59% over the time periods analyzed (2000-2002 and 2007-2008). On average, small vessel (<15 m [<50 ft] long) noise presence increased from <20% of samples (2000-2002) to approximately 30% of the samples (2007-2008). Medium vessel (15 m–30 m [50 ft–100ft] long) noise was present in the highest proportion of acoustic samples (40-50% during the day). Higher proportions (87-90%) of hourly samples containing vessel noise occurred between 0500 and 2100 hours for both study periods. There was an increase in proportions of hourly samples with vessel noises between the 2000–2002 study and the 2007–2008 study (Figure 182). Hourly vessel noise seems to have increased between 0800 and 1400 hours, and between 1600 and 1900 hours since 2002. The analysis showed that large vessels (>30 m [>100 ft] long) may be entering and leaving the park at a similar time each day and therefore present in the same acoustic samples, which results in less time with large vessel noise present.

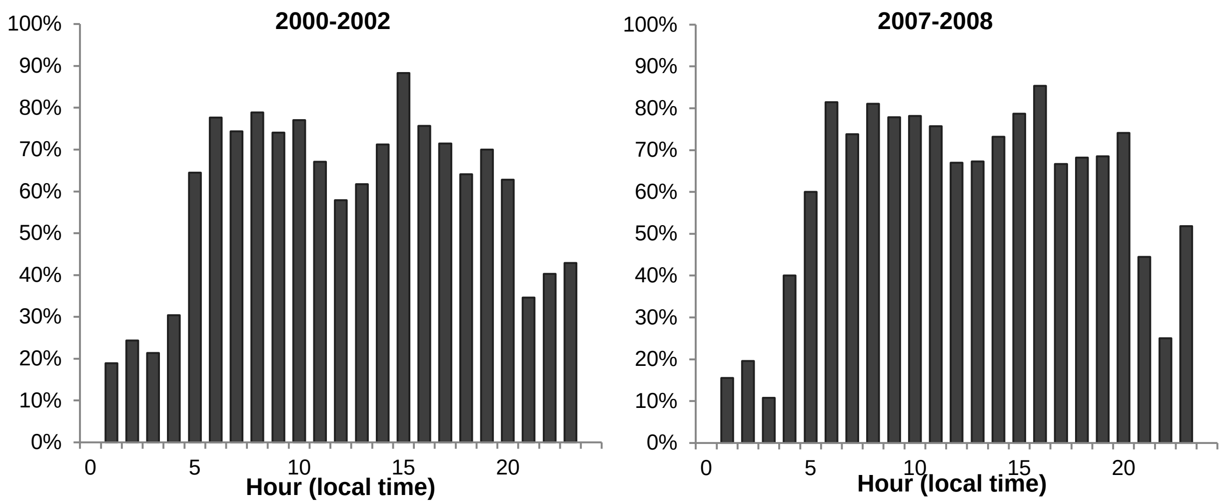

Figure 182. Proportion of hourly samples with vessel noise in 2000-2002 (left) and 2007-2008 (right) (McKenna et al. 2014).

<u>Duration of Noise-Free Intervals</u>

There were no data on the duration of noise-free intervals in the park. The studies mentioned above concentrated only on the sounds that could be heard both biological and anthropogenic. Determining the duration of noise-free intervals is important for assessing amount of time that underwater soundscape is unaffected by anthropogenic noise. McKenna et al. (2014) described an increase in the proportion of samples with vessel noise, but did not record the duration of noise-free intervals. The increase in proportion of samples with vessel noise may mean that the duration of noise-free intervals has decreased from 2000-2008, which is less than the historic reference condition.

Masking Index of Communication Space for Vocal Species

Determining the masking index of communication space is important because it allows managers to understand how the loudness, proximity, and source of a sound affect vocal marine species. Introduction of anthropogenic sounds (e.g., tour boats, cruise ships) has been known to alter or mask biological signals between marine animals. Masking occurs when sound disrupts a biological signal between two or more animals and can completely or partially conceal a signal within its communications space. Communication space is described as the active and physical space around an animal used for communication.

The masking index of communication space was analyzed in Glacier Bay Proper using computer modeling. The acoustic monitoring data from the GBP was used in the analysis to determine to what degree vessel noise is compromising the vocal ability of two species, humpback whales and harbor seals (Gabriele et al. 2015). Looking at frequencies of both the humpback whales and harbor seal vocals, and vessels that enter the GBP revealed a severe reduction in communication space during vessel traffic. This is because vessel traffic emits noise levels (71-224 Hz) that are within the frequency bands of animal vocals (71-355 Hz: humpback whales and harbor seals) and reduces their available communication space.

The results clearly indicated that vessel noise within GBP is significantly impacting the acoustic environment (Gabriele et al. 2015). For both harbor seal roars and humpback whale songs and calls, a significant reduction in communication space occurred in GBP with the presence of vessel traffic (Gabriele et al. 2015). These losses in communication space were highest on days of high vessel traffic and were most disruptive to humpback whale calls, called “whups,” but also significantly disrupted whale song and harbor seal roars as well (Gabriele et al. 2015). The rates of disruption for humpback whale “whups” calls, songs, and harbor seal roars varied with the total number of vessels present.

Humpback whale whups (71-224 Hz) are short duration, low source level sounds in the lowest frequency band. Communication space for whale whups was reduced by 80% to 99% during low, moderate, and high vessel traffic conditions (Gabriele et al. 2015). Humpback whale songs (224-708 Hz) are loud, long duration sounds that are within the widest and highest frequency band that was modeled for this study. Even these vocal communications had 30-60% reductions in the communication space during vessel traffic. Harbor seal roars (178-355 Hz) are a low source level sound and have longer duration and higher frequency than humpback whale whups. These roars communication space was reduced by 60-80% (Gabriele et al. 2015). All three vocals had the largest reduction in communication space during daytime, when vessel traffic was highest (Gabriele et al. 2015).

Reductions in communication space had a noticeable pattern with season and time of day. Summer is the GLBA visitor season, vessel traffic conditions during this time result in substantial losses of communication space for humpback whales and harbor seals.

Threats and Stressor Factors

GLBA staff identified two common threats and stressors to the underwater soundscape in the park: vessel noise inside the park and vessel noise coming in from outside the park. McKenna et al. (2014) did not differentiate whether the noise came from inside or outside the park, but there is no doubt that vessel noise is a major threat to the natural soundscape of GLBA.

Vessel noise is a threat to the natural underwater soundscape inside the park because it negatively affects a variety of marine wildlife (e.g., larval fish, invertebrates, harbor seals, and humpback whales) in a variety of ways. Marine animals are adapted to rely on sound for everyday activities such as feeding, intraspecific location, intraspecific communication, and detection of predators (Gabriele et al. 2010, Holt et al. 2011, Wale et al. 2013); therefore, vessel traffic noise has the potential to interfere with these activities. A majority of the (e.g., cruise ships, small motorized vessels) traveling through the bay area for visitors to view the park; these vessel trips usually come at the same time frames every day throughout the visitor season (summer) and their numbers are not likely to decline. The park staff has enforced mandatory speed reductions (from 20 knots to 13 knots) in the lower portion of Glacier Bay during the summer season, which reduces the level of noise and intensity of adverse effects on underwater soundscape.

Data Needs/Gaps

Underwater soundscape data have been collected since May 2000 to present day. There are limited data on underwater sounds prior to the installation of the hydrophone in 2000 (Miles and Malme 1982). In addition, this long time series of data only covers lower Glacier Bay, and not the majority of park waters. Analyses have not been conducted on the duration of noise-free intervals. Analysis regarding the masking index of communication space for vocal species is underway but is computationally intense and would need to be re-evaluated periodically. The continuation of underwater sound monitoring is essential to assessing future condition and trends on how anthropogenic sources affect the underwater soundscape in the park. Establishing a baseline on the duration of noise-free intervals and the masking index of communication space for vocal species will also aid park managers in understanding the biological relevance of anthropogenic noise in the Bay to species of interest.

Overall Condition

Underwater Natural Ambient Sound Level (dBa) at Frequencies of Interest

The project team defined the *Significance Level* for underwater natural ambient sound level (dBa) at frequencies of interest as a 3. Both Kipple and Gabriele (2003) and McKenna et al. (2014) documented wind generated noise (natural sound) as being the most prevalent sound, potentially indicative of a healthy soundscape (Gabriele, written communication, 21 May 2015). The studies documented similar average sound level (84 dB, 83.5 dB) and range in sound (67-102 dB; 70-100 dB). According to McKenna et al. (2014), wind noise could be heard 28-38% of the time between May and September.

Rain noise was also documented in both Kipple and Gabriele (2003) and McKenna et al. (2014); Kipple and Gabriele (2003) documented rainfall sound level ranging from 91-110 dB, while

McKenna et al. (2014) documented rainfall sound level ranging from 70-100 dB with an average sound level of 88.8 dB. Even with vessel noise in many of the data samples, natural ambient sounds such as wind, rain, harbor seals, orcas, and humpback whales can be heard frequently; wind can be heard in almost all of the samples. Figure 178 may serve as a reference condition for historic ambient noise in the park. Due to the increase in small and medium vessels resulting from changes in the vessel quotas implemented by the park (McKenna et al. 2015) and the potential for this vessel noise to interfere with natural ambient sounds, this measure is assigned a *Condition Level* of 1.

Proportion of Hourly Samples without Vessel Noise

This measure was assigned a *Significance Level* of 3. Kipple and Gabriele (2003) documented proportions of ambient noises without vessel noise (41-89%). Winter months had the highest proportions of samples without vessel noise, and summer months had the lowest proportion. The higher proportion of vessel noise in the summer is due to the summer season having more vessel traffic. Gabriele et al. (2010) documented decreased proportions of samples without vessel noise in three seasons. Proportions were not calculated for the winter season in 2007 and 2008.

McKenna et al. (2014) reported the overall occurrence of vessel noise in lower Glacier Bay increased significantly from 51-59% from 2000-2002 and 2007-2008. The majority of hourly vessel noise was present in the park between 0500 and 2100 hours. On average, small vessel (<15 m [<50 ft] long) noise presence increased from <20% of samples (2000-2001) to approximately 30% of the samples (2007-2008). Medium vessel (15 m–30 m [50 ft–100 ft] long) noise was present in the highest proportion of acoustic samples (40-50% during the day). Higher proportions (87-90%) of hourly samples containing vessel noise occurred between 0500 and 2100 hours for both study periods. There was an increase in proportions of hourly samples with vessel noises between the 2000–2002 study and the 2007–2008 study (Figure 182). The increase in hourly samples with vessel noise causes stress and an added vulnerability to the marine mammals in the Bay. More vessel noise throughout the day means more marine mammal activity (e.g., communication, intraspecific location, and detection of predators) may be masked. As a result, this measure was assigned a *Condition Level* of 2, or of moderate concern.

Duration of Noise-Free Intervals

The project team defined the *Significance Level* for this measure as a 3. There were no data on the duration of noise-free intervals in the park. McKenna et al. (2014) described an increase in the proportion of samples with vessel noise, but did not record the duration of noise-free intervals. The increase in proportion of samples with vessel noise may mean that the duration of noise-free intervals has decreased from 2000 to 2008, which is less than the historic reference condition. A *Condition Level* could not be assigned for this measure due to the lack of data pertaining to the duration of noise-free intervals.

Masking Index of Communication Space for Vocal Species

A *Significance Level* of 2 was assigned for this measure. Gabriele et al. (2010) determined the mask index for humpback whale songs and calls in GLBA. Less than 20% of whale songs are lost due to vessel noise, but 40-95% of whale calls (whups) are lost. This may be the result of calls being shorter and in one frequency, while whale songs are long and range in frequency. A higher percentage of songs can be heard to interpret. The masking index was only determined for one vocal species between 0100 and 1300 hours. There were no other data on the masking index of communication space for vocal species; however, the rise of low frequency ambient noise does cause concern that masking is impacting the biological environment. As a result, a *Condition Level* of 2 was assigned to this measure, which indicates moderate concern.

Weighted Condition Score

This component was assigned a *Weighted Condition Score* of 0.54, which indicates there are signs of limited and isolated degradation of the resource. There has been an increase in underwater vessel noise which seems to convey a declining trend in the condition of the underwater soundscape in GLBA.

Underwater Soundscape			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.54**
Underwater Natural Ambient Sound Level (dBa) at Frequencies of Interest	3	1	
Proportion of Hourly Samples without Vessel Noise	3	2	
Duration of Noise-Free Intervals	3	n/a	
Masking Index of Communication Space for Vocal Species	2	2	

4.23.6. Sources of Expertise

- Christine Gabriele, GLBA Wildlife Biologist, Humpback Whale Monitoring Program

4.23.7. Literature Cited

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4.24. Wilderness

4.24.1. Description

Congress passed the Wilderness Act of 1964 to protect natural lands from threat of "expanding settlement and growing mechanization" (U.S. Congress 1964). This Act established the National Wilderness Preservation System (NWPS) to protect wilderness areas and preserve their wilderness character (U.S. Congress 1964). Each designated wilderness is managed by one or more federal agencies: the BLM, NPS, USFWS, or USFS; approximately 40% of the lands in the NWPS are administered by the NPS (NWSC 2013, Landres et al. 2014). Congress and legal scholars have confirmed that the primary legal mandate of the Wilderness Act is to preserve the wilderness character of all areas designated as wilderness (Carver et al. 2013). The policies of the agencies that manage wilderness also state that they are to preserve wilderness character in all areas that are designated as wilderness (Tricker et al. 2012). Specifically, NPS Management Policies 2006, Chapter 6, "Wilderness Preservation and Management," states that, "The purpose of wilderness in the national parks includes the preservation of wilderness character" (Landres et al. 2014, p. 7).

The Glacier Bay Wilderness was established on 2 December 1980, with the passage of ANILCA (NPS 1989). ANILCA also expanded the park boundaries, created the adjacent National Preserve, and provided additional NPS legislative directions for the backcountry management of park resources (NPS 1989). ANILCA originally designated over 1 million ha (~2.6 million ac) of GLBA's nearly 1.3 million ha (~3.2 million ac) as wilderness (NPS 1989, 2010). The designated wilderness area has increased in size since 1980 due to a variety of factors ranging from land exchanges, isostatic rebound, and adoption of modern digital mapping and calculation methodologies, and today GLBA has nearly 1.1 million ha (~ 2.7 million ac) designated as wilderness (NPS 2010). This makes the Glacier Bay Wilderness one of the largest in the NPS (Photo 58) (NPS 2010).

Photo 58. Toyatte Glacier at GLBA (NPS photo).

The Glacier Bay Wilderness encompasses nearly all the land within the national park portion of GLBA, with over 99% of the park land as either designated or eligible wilderness (Figure 183). In addition, approximately 13% of the marine waters of GBP (not including the designated portion of

Dundas Bay) are designated or eligible wilderness. The boundary of the wilderness area extends along the Gulf of Alaska to the mouth of the glacial-fed Alsek Lake and areas surrounding the Chilkat and Fairweather Mountain Ranges. Except for the road that runs from Gustavus to the park headquarters in Bartlett Cove, no roads exist within the park. With limited access, large areas of GLBA are rarely visited. Five marine areas are included in the designated wilderness; Rendu, Hugh Miller, and Adams Inlets; the north and west arms of Dundas Bay, and the waters surrounding the Beardslee Islands (NPS 1989).

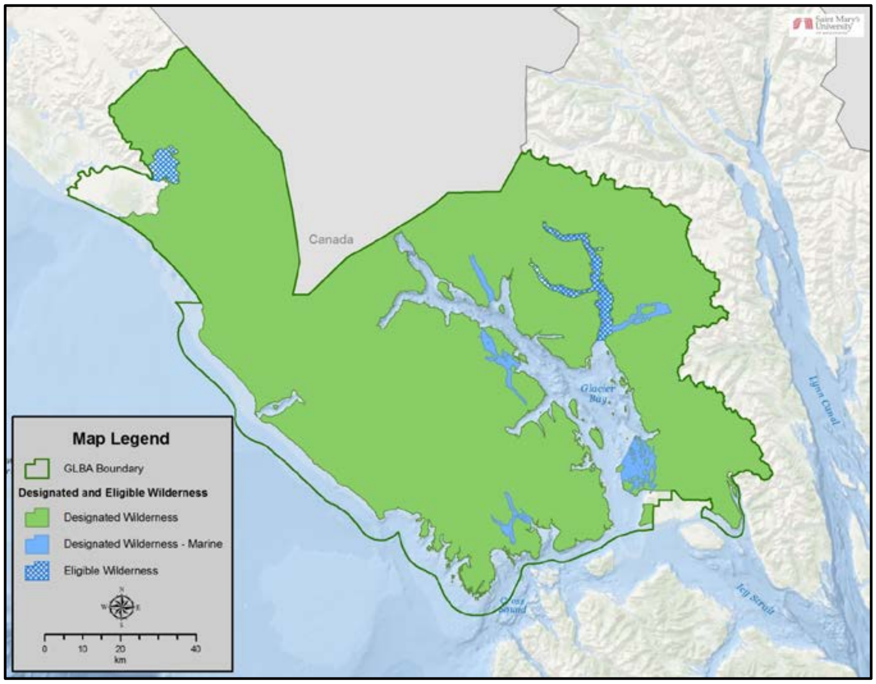


Figure 183. Location of designated and eligible wilderness areas at GLBA.

GLBA's wilderness is managed in accordance with the mandates of the Wilderness Act of 1964 and NPS Management Policies; there are also special provisions in ANILCA for managing wilderness in Alaska (NPS 1989). GLBA has a wilderness visitor use management plan that covers visitor use and activities within the designated wilderness areas (NPS 1989). Plans like these fulfill the NPS policy on wilderness preservation and management by developing and maintaining a wilderness stewardship plan to guide the preservation, management, and use of wilderness resources (NPS 1989).

4.24.2. Measures

- Natural Quality
- Undeveloped Quality
- Untrammeled Quality
- Solitude or Primitive and Unconfined Recreation Opportunity

4.24.3. Reference Conditions/Values

For each measure in this component, a baseline condition was based on either real data or professional judgment from 5 years prior, and the current condition was similarly based on real data or professional judgment.

4.24.4. Data and Methods

Because of the unique nature of this component, the identified measures were further subdivided in order to better assess the current condition of this resource. Sources of data used to describe and define the condition of the identified measures will either have been introduced previously in this NRCA, or will rely on the best professional judgment of GLBA and NPS resource managers. Listed below is how each of the four measures was sub-divided in this component.

Natural Quality

In this assessment, the current condition of the Natural Quality measure was determined by analyzing the following sub-measures:

- Plant and Animal Species: focused on the intact natural ecosystems of GLBA (natural biodiversity, abundance, and distribution), the number of ESA-listed species, the number of invasive plant species, and the number of invasive animal species.
- Physical Resources: included discussions on GLBA's freshwater quality (including contaminants), marine water quality (including contaminants but not including OA), OA, air quality, and the natural glacial areal extent and volume.
- Biophysical Processes: summarized by discussions of natural ecosystems and community development processes.

Undeveloped Quality

For this assessment, the Undeveloped Quality measure was assessed by analyzing the following sub-measures:

- Non-recreational Structures and Installations: focused on the number of administrative communications installations, aids to navigation, or non-historic or unauthorized user-created structures; the number of long-term climate monitoring stations; and the number of non-recreational instruments/markers.
- Useoff Motor Vehicles, Motorized Equipment, and Mechanical Transport: utilized the number of motor vehicles, mechanical transports (motorized vehicle entries, helicopter landings), or uses of motorized equipment in order to assess its current condition.
- Inholdings: focused exclusively on the number of private inholdings to assess current condition in the park.

Untrammeled Quality

For this assessment, the Untrammeled Quality measure was assessed using the following sub-measures:

- Actions Authorized by Federal Land Manager: analyzed by looking at the number of authorized actions in GLBA.

- Actions Not Authorized by Federal Land Manager: analyzed by looking at the number of unauthorized actions in GLBA.

Solitude or Primitive and Unconfined Quality

For this assessment, the Solitude or Primitive and Unconfined Quality was assessed by the following sub-measures:

- Remoteness from Sights and Sounds of People Inside the Wilderness: analyzed by looking at the number of overnight backcountry campers in GBP during the summer visitor season, the number of backcountry days in GBP during the summer visitor season, the number of prohibited human technologies, and the Bartlett River Trail count.
- Remoteness from Occupied and Modified Areas Outside the Wilderness: described by summarizing disturbance by (number of) permitted motorized vessels during the summer visitor season, disturbance by (number of) administrative vessels during the summer visitor season, and disturbance by (number of) aircraft overflights during the summer visitor season.
- Facilities that Decrease Self-reliant Recreation: looked only at the number of facilities that decreased self-reliant recreation.
- Management Restrictions on Visitor Behavior: summarized the number of required camping permits for GBP issued, and the number of camping/foot traffic restrictions.

4.24.5. Current Condition and Trend

Natural Quality

Natural Quality is defined by Landres et al. (2008) as wilderness ecosystems that are substantially free from the effects of modern civilization. Degradations to this measure are typically observed through the effects and changes that modern civilizations and people have had on the natural ecological systems after the time of wilderness designation (Landres et al. 2008). Many of the sub-measures of Natural Quality have been previously discussed in detail in this NRCA.

Plant and Animal Species

Intact natural ecosystems of GLBA (natural biodiversity, abundance, and distribution)

The natural ecosystems of GLBA are largely considered nearly pristine by informed scientists and park staff, and there is little evidence to suggest the contrary. The natural biodiversity of the area is about what should be expected, and many areas remain free from human disturbances or alterations. New species to science have been discovered in GLBA in recent years (Spribille and Fryday 2012, Spribille *In prep*), and scientists continue to work to fully understand the complex successional patterns that are ongoing in the park. The majority of the biotic components highlighted in this NRCA exhibited good current condition (see discussion in Chapter 5 of this NRCA), and the abundance and distribution trends for these species were almost all either stable or increasing. Species such as sea otters, humpback whales, and Steller sea lions have all exhibited increasing trends in recent years, and are excellent representations of the intact natural ecosystems of the park.

Number of ESA-listed species

The number of ESA-listed species in GLBA has been similar for many years, as there are two ESA-listed species currently in GLBA: the humpback whale (endangered) and the Steller sea lion (threatened). The eastern DPS of Steller sea lions was de-listed in 2013 (50 CFR 223 and 224), while the western DPS is still listed under the ESA. Both of these species are well studied in the park, and current condition for both species was determined to be of good condition with increasing trends (of this NRCA: see Chapter 4.12 for humpback whales; see Chapter 4.15 for Steller sea lions).

Number of invasive plant species

The number and distribution of non-native invasive plant species park-wide is discussed in-depth in Chapter 4.19 of this NRCA. As of 2014, there were five invasive plant species that were established within the wilderness of the park: common dandelion, perennial sow thistle, chickweed (*Stellaria media*), mouse-ear chickweed, and reed canary grass. An oxeye daisy (*Leucanthemum vulgare*) infestation was observed in Reid Inlet between 2004 and 2008, but this infestation was hand-pulled annually during this time and is presumed to be eradicated.

The majority of invasive plant species have been found in developed (non-wilderness) areas of the park, particularly in Bartlett Cove and Dry Bay. Common dandelion appears to be the only species that has become established in the backcountry (Fisk 2011). Given the expansive size of GLBA's wilderness, there is a low diversity of invasive species with very few populations. Anecdotal observations by GLBA managers do not indicate any rapid diversification or spread of invasive species in the park's wilderness (NPS unpublished data). The populations of invasive plants that do exist in the park's wilderness are restricted both in size and extent. While there are no indications that the number of invasive plants is increasing, there still exists a moderate concern regarding the potential impact that these species could have on native plant community and ecosystems.

Number of invasive animal species

GLBA managers have indicated and observed that invasive animals have had a difficult time establishing in the park's wilderness. Currently, only one invasive animal species has been documented in the park's wilderness (Atlantic salmon). However, this species has not been documented in the park since its original observation in 2002. Intermittent monitoring for both marine and terrestrial invasive animals exists, and to date no additional species have been documented in GLBA's wilderness. Eurasian collared-doves have been documented in non-wilderness areas of GLBA since around 2010. Additional invasive animal species that could establish in GLBA's wilderness include: marine tunicates (e.g., *Didemnum vexillum, Botrylloides violaceous, Botryllus schlosseri*), European green crabs, gypsy moths, and European black slugs (*Arion ater*).

Physical Resources

Freshwater water quality

Approximately 445,155 ha (1.1 million ac) of freshwater are contained within GLBA, primarily in the form of snow and ice, rivers, streams, lakes, and ponds (NPS 2008). The watersheds within the park are relatively small, and most streams are short (<20 km [12 mi]) with steep gradients (5-20%) (Eckert et al. 2006). Freshwater input into these streams is provided predominantly by snow and glacial meltwaters. Glacial retreat within GLBA continually creates new landforms, including streams and ponds (Eckert et al. 2006).

Chapter 4.22 of this assessment discusses park-wide water quality in depth, with the bulk of the freshwater discussion focusing on contaminants such as mercury, methylmercury, and POPs. Current mercury concentrations in sampled GLBA streams were three to four orders of magnitude below the EPA thresholds for aquatic organisms and human health (Nagorski et al. 2011). In GLBA, wetlands play an important role in the conversion of inorganic mercury to the more toxic methylmercury (Eckert et al. 2006, Wiener et al. 2006). While the prevalence of wetlands in some areas of GLBA is a concern with regard to MeHg, the overall levels of Hg in park samples were well below EPA levels of concern for human health or for aquatic organisms. Similarly, the overall levels of POPs detected in GLBA were low and fell one to three orders of magnitude below the criteria recommended for the protection of piscivorous wildlife (Newell et al. 2000, Nagorski et al. 2011). While the threat of contamination of freshwater in the park's wilderness areas continues to be a concern for managers, it appears that the condition of freshwater in the park is currently good (see Chapter 4.22).

Marine water quality

Roughly 232,695 ha (575,000 ac) of either saltwater or brackish water exists in GLBA (NPS 2008). Long-term monitoring of general oceanographic parameters (since 1993) and marine contaminants indicate generally pristine and unchanging marine water quality. While the current condition of this resource appears to be of low concern, water quality remains a critically important resource due to the tremendous number of marine-related resources present in the park.

Chapter 4.22 discussed water quality park-wide, but also focused on the presence of metals, POPs, and PAHs for the marine ecosystems. Mussel samples from GLBA intertidal zones showed very low levels of the four metals analyzed, particularly in comparison to other locations in the U.S. (Kimbrough et al. 2008, Tallmon 2012). No POP levels exceeded the criteria for the protection of human health or wildlife in GLBA (Nagorski et al. 2011), and PAHs have been detected in GLBA at very low levels (Tallmon 2012).

Ocean acidification

One additional threat that is of concern to park managers is OA. If global emissions of CO_2 continue to increase based on current trends, the average pH of the oceans could fall by as much as 0.5 units by the year 2100 (Raven et al. 2005). The pH of the oceans has already decreased from 8.17-8.09 over the past 200 years (Haufler et al. 2010). Species that are dependent upon calcium carbonate availability are directly impacted by OA, as the amount of calcium carbonate available to organisms decreases as the oceans become more acidic. Reisdorph and Mathis (2014) modeled a likely future scenario concerning OA within GLBA, showing that OA could become severe enough in 100-150 years that aragonite (a biologically important form of calcium carbonate) would be more likely to dissolve than to form shells. Declines in calcium carbonate-dependent species (e.g., shellfish, mollusks, crustaceans, pteropods) could have indirect impacts on anadromous fish communities, Steller sea lions, forage fish communities, sea otters, and a suite of other marine predators in the park (PMEL 2008, Haufler et al. 2010, NRC 2010, IWGOA 2011).

While atmospheric CO_2 levels certainly are the dominant driver of increasing ocean acidification worldwide, it has been shown that OA in the park is being additionally mediated by contributions of glacial meltwater to the ocean water, and that the effects vary based on location and seasonality (Reisdorph and Mathis 2014). Because much of GBP is surrounded by glaciers and snow/ice fields, the meltwater directly enters the bay in several locations. The meltwater that enters the bay is typically low in total alkalinity, which enhances the bay's vulnerability to reductions in pH levels (Reisdorph and Mathis 2014).

Reisdorph and Mathis (2014, 2015) have confirmed that OA is occurring within the park. However, the total potential and present impacts that this process is having on GLBA's ecosystems is still poorly understood at this time. While additional research is needed regarding this topic, it remains a metric that is of moderate concern to the park at this time.

Air quality

The air quality of Southeast Alaska is generally believed to be among the most pristine in the world due to the area's low population density, lack of large-scale industrial development, and vast wildland areas (Dillman et al. 2007, Schirokauer et al. 2014). However, the air quality is currently threatened by global industrial pollution and local sources such as cruise ships. Air quality is of high value to SEAN park managers, and contaminants from global sources are a growing concern due to their possible impacts on a wide variety of SEAN Vital Signs (Moynahan et al. 2008).

While additional air quality-related data are needed to accurately and confidently assess trend and current condition, there does not appear to be any major cause for concern at present (see Chapter 4.21). Recently-installed climate monitoring stations will allow for a more accurate assessment of this resource's current condition. Recent data related to airborne contaminants (e.g., Schirokauer et al. 2014) appear to indicate generally high levels of air quality with some slightly elevated mercury deposition occurring in the region.

Natural glacial areal extent and volume

As discussed in Chapter 4.1, GLBA has significant glacial coverage and is one of nine NPS units in Alaska that have glacial coverage. Glacial dynamics are naturally variable over long timescales, but cycles of glacial advancement and retreat in GLBA have occurred rapidly in geologic terms. While some glaciers in the park are currently advancing, many of the park's remaining glaciers continue to retreat, as the total extent of glacial coverage in the park has decreased approximately 15% since 1950 (Loso et al. 2014). All of the 16 glaciers within GLBA that underwent surface elevation analysis from 1995-2011 showed declines (i.e., thinning) over that time period (Loso et al. 2014). Mass balance estimates based on surface elevation changes showed that thinning generally ranged from 0.1-1.5 m/yr w.e. Only Margerie Glacier experienced increases in surface elevations in some areas which offset declines in other areas, contributing to a positive mass balance between 2009 and 2011 (Loso et al. 2014). The recent (1995-2011) general trend of shrinkage and thinning observed in many of the park's glaciers is likely driven in part by global climate change, and is a significant concern to park managers.

Biophysical Processes

Natural ecosystems and community development processes

Glacial retreat and isostatic rebound continually create and remove habitat throughout the park; because of this, biotic and abiotic communities are constantly adjusting and moving. While these ecological communities in the park are transitioning due to succession, this is by no means to be viewed as a detriment to the overall health of the ecosystem. GLBA is widely recognized as one of the best documented examples of terrestrial primary succession following deglaciation (Chapin et al. 1993). The successional process continues today, and the many years of this phenomenon recorded within GLBA represents perhaps the most studied area of ecological succession in the world (Cooper 1923, 1931, 1939; Lawrence 1958; Reiners et al. 1971; Walker 1999).

As the Glacier Bay Ice Sheet receded, salmon began to colonize the many freshwater streams that were formed. These processes of glacial recession, stream succession, and colonization are ongoing in Glacier Bay. Milner et al. (2011) found that the succession of the fish community in GBP streams is strongly related to stream age, and further found that salmonid abundance and distribution may be dependent on habitat complexity and stability. Additional research has demonstrated that the flow buffering effects of lakes in watersheds accelerates successional processes and, in turn, salmon colonization (Milner 1987, Milner and Bailey 1989). Similarly, the park's bear and moose communities continue to develop and expand with successional processes. The park's moose population appears to be expanding (although additional research is needed to confirm this), and vegetation in early to mid-successional stages (e.g., shrubs) provides critical forage for moose populations (Stephenson et al. 2006). Later successional stages, where trees are more dominant, support fewer moose browse species, but do provide valuable cover in the winter and during calving season (Stephenson et al. 2006). The distribution of bears in GLBA is likely related to food resource availability in different vegetation and stream successional stages, and bear distribution can vary between years in response to food availability (Publicover 1985, Lewis 2012). Black bears could expand their range north as the landscape in northern GBP matures from early, open successional stages to closed scrub and forest (Lewis 2012).

Biological successional processes and associated physical landscape evolution are generally considered by informed scientists and managers to be proceeding in a natural and healthy fashion, with little to no verifiable evidence to the contrary (NPS unpublished data).

Undeveloped Quality

One of the core ideas or principles of wilderness is that it encompasses an area that is undeveloped (U.S. Congress 1964, Landres et al. 2008). As described in Section 2(c) of the 1964 Wilderness Act (U.S. Congress 1964), wilderness is "…an area of *undeveloped* [emphasis added] federal land retaining its primeval character and influence, without permanent improvements or human habituation." Degradations to this measure are typically exemplified by the presence of some sort of man-made, non-natural structure. Examples of these degradations may include installations, habitations, or any other evidence of human presence of occupation (Landres et al. 2008).

Non-recreational Structures and Installations

Number of administrative communications installations, aids to navigation, or non-historic or unauthorized user-created structures

Currently, there are two radio repeaters, four aids to navigation, no non-historic structures, and no known unauthorized user-created structures in GLBA's wilderness. One unauthorized user-created structure was removed from GLBA's wilderness in 2011 (NPS unpublished data). The current quality of this sub-measure is deemed to be of low concern, with an improving trend due to the recent removal of a structure.

Number of long-term climate monitoring stations

Since 2010, 27 defunct or poorly performing climate monitoring stations have been removed from the park. Twenty-four of the monitoring stations that were removed were small stations, and the remaining three stations were much larger. Two modern remote automated weather stations (RAWS) were installed in GLBA in August of 2015, and these stations represent the only long-term climate monitoring stations currently in GLBA (NPS unpublished data). The overall level of concern for this sub-measure is low, and the condition has improved in recent years due to the removal of a large number of stations throughout the park.

Number of non-recreational instruments/markers

The exact number of non-recreational instruments and markers is not known with certainty, but park managers estimate that the current number is seven (NPS unpublished data). It is believed that this number (or estimate) has remained relatively stable over the past several years. These instruments are small and relatively unobtrusive, and considering the size of the park's wilderness it is likely that these instruments have minimal impact on the Undeveloped Quality.

Number of motor vehicles, mechanical transports (motorized vehicle entries, helicopter landings), or uses of motorized equipment.

Based on incomplete data, the number of motorized transport/equipment uses in GLBA wilderness is poorly known. The best information available to park managers suggests that the current (2012-present) number of motor vehicles, transports, or uses of motorized equipment in the park's wilderness averages about two to three instances per year. Within the past decade, there does not appear to be evidence that the number of motorized transport or equipment usage was any higher than present averages, and it may have been lower than the current two to three instances per year estimate (NPS unpublished data).

According to park managers, the number of entries by motorized vessels into designated wilderness waters appears to be low, suggesting minimal impact on the park's wilderness considering the sheer magnitude of the wilderness area; the level of confidence is low, however. The number of motorized vessel entries into eligible wilderness waters (and wilderness waters open to motorized access) is not known.

Number of private inholdings

There are eight private inholdings in GLBA's wilderness: six native allotments that have been formally conveyed, and two that are approved for conveyance. One allotment (not included in the total previously mentioned) was purchased by the NPS in 2007. There is a single valid subsurface mining claim that has been present in the park for many years.

For the size of the park's wilderness, there are remarkably few private inholdings threatening the Undeveloped Quality of the park's wilderness (NPS unpublished data). The inholdings that do exist are typically very small, and to the best knowledge of GLBA managers none of the existing inholdings has seen any development outside of boundary markings.

Untrammeled Quality

Untrammeled Quality refers to a wilderness that is wild and un-manipulated by man. As described by Congress (1964, Section 2(c)), wilderness is "…recognized as an area where the earth and its community of life are untrammeled by man." Untrammeled Quality of wilderness is degraded through intentional, man-made manipulations of ecosystems or ecological processes (Landres et al. 2008). Examples of degradations to a wilderness' Untrammeled Quality include fire suppression, tree removal, the installation of a dam, or the removal of biotic species (Landres et al. 2008).

Number of Actions Authorized by the Federal Land Manager in GLBA

Since 2002, there have been zero to two authorized actions per year in GLBA's wilderness, and managers have noticed no discernable trend. Most of the authorized actions have been very spatially restricted, and have involved either targeted manual removal or herbicide treatment of discrete, single-species invasive plant infestations. Given the large size of the park's wilderness, and the small spatial scale of these restricted actions, the current number of authorized actions in GLBA's wilderness is of low concern and still allows for a good condition of Untrammeled Quality in the park.

Number of Actions Not Authorized by the Federal land Manager in GLBA

To the best knowledge of park managers, there have been no known unauthorized actions in GLBA's wilderness areas.

Solitude or Primitive and Unconfined Recreation Opportunity Quality

Wilderness is to have "…outstanding opportunities for solitude or a primitive and unconfined type of recreation," as defined by the Wilderness Act (U.S. Congress 1964, Section 2 (c)). The interpretation of solitude varies by source and time period, but Landres et al. (2008, p. 26) described solitude as being

> *…viewed holistically, encompassing attributes such as separation from people and civilization, inspiration (an awakening of the senses, connection with the beauty of nature and the larger community of life), and a sense of timelessness (allowing one to let go of day-to-day obligations, go at one's own pace, and spend time reflecting).*

Similarly, the interpretation of primitive and unconfined recreation has also been debated. According to Landres et al. (2008, p. 27):

> *Primitive recreation has largely been interpreted as travel by non-motorized and non-mechanical means (for example by horse, foot, or canoe) than reinforce the connection to our*

> *ancestors and our American heritage. However, primitive recreation also encompasses reliance on personal skills to travel and camp in an area, rather than reliance on facilities or outside help (Roggenbuck 2004). Unconfined encompasses attributes such as self-discovery, exploration, and freedom from societal or managerial controls (Lucas 1983, Nash 1996, Hendee and Dawson 2002).*

Solitude or primitive and unconfined recreation opportunities can be degraded by any reduction of a person's opportunity to experience true wilderness. Examples of these degradations may include encountering other visitors in the park's wilderness, or any instance or facility that may reduce a visitor's self-reliance (Landres et al. 2008).

Remoteness from Sights and Sounds of People inside the Wilderness

Number of overnight backcountry campers in GBP during summer visitor season

From 2007-2014, the average number of overnight backcountry campers per year was 886 (NPS unpublished data). Park managers note that no discernable trend was observed during that time period. The majority of the backcountry camping that occurs in GLBA is concentrated in GBP during the summer visitor season, with most campers utilizing the marine shorelines. Overall, the total number of campers is relatively low, and the opportunities to experience solitude and unconfined recreation remain excellent in the park. There are unparalleled opportunities for visitors to experience remoteness from sights and sounds in areas outside of GBP or inland within GBP.

Number of backcountry days in GBP during the summer visitor season

From 2007-2014, the average number of backcountry days in GBP during the summer season was 3,973 per year (NPS unpublished data). Park managers note that no discernable trend was observed during that time. Much like the number of overnight campers discussion, GLBA offers an unparalleled opportunity for visitors (whether overnight or just day-use) to experience solitude and unconfined recreation.

Number of prohibited human technologies

Section 4(c) of the Wilderness Act states that "…there shall be no temporary road, no use of motor vehicles, motorized equipment or motorboats, no landing of aircraft, no other form of mechanical transport, and no structure of installation within any such area [wilderness]" (U.S. Congress 1964, Section 4(c)). The number of prohibited human technologies allowed to occur (or unauthorized uses) is generally very low in GLBA wilderness areas. Recently, there have been around two to three administrative exceptions to Section 4(c) of the Wilderness Act per year. The exceptions that are allowed to occur are almost exclusively for scientific research.

Bartlett River Trail count

The Bartlett River Trail is one of only two maintained trails in GLBA's wilderness; the trail starts in the park's front country non-wilderness and is used primarily by sport anglers for access to the most accessible freshwater fishing opportunity in the park. Since 2012, the number of round-trip hikers on this trail has averaged about 1,820 users per year. The number of users increased by 29% from 2012-2013, but then deceased by 10% in 2014 (NPS unpublished data). The level of use of this trail likely has little direct relevance to the wilderness-wide condition of Solitude or Primitive and Unconfined

Quality, due mostly to the fact that this trail encompasses a very small portion of the park's vast wilderness area.

Remoteness from Occupied and Modified Areas outside the Wilderness

Disturbance by (number of) permitted motorized vessels during the summer visitor season

There are no objective and quantitative data available to quantify the level of disturbance by this source to wilderness visitors. The numbers of permitted vessels are tightly controlled, and have remained basically unchanged for many years. In the professional judgment of park managers, this source of disturbance does not substantially diminish the condition of this indicator.

Despite some reduction of the feeling of remoteness caused by permitted vessels in adjacent non-wilderness waters within GB proper, the character of "remoteness" regarding the larger Solitude or Primitive and Unconfined Quality is very high throughout the park's wilderness.

Disturbance by (number of) administrative vessels during the summer visitor season

There are no objective and quantitative data available to quantify the level of disturbance by this source to wilderness visitors. The numbers of permitted vessels are tightly controlled, and have remained basically unchanged for many years. In the professional judgment of park managers, this source of disturbance does not substantially diminish the condition of this indicator.

Despite some reduction of the feeling of remoteness caused by administrative vessels in adjacent non-wilderness waters within GBP (especially to wilderness visitors experiencing the GBP watershed only from the shoreline) there remain vast areas of the park wilderness for which administrative vessel use likely has no impact. Consequently, the character of "remoteness" regarding the larger Solitude or Primitive and Unconfined Quality is very high throughout the park's wilderness as a whole.

Disturbance by (number of) aircraft overflights during the summer visitor season

There are no objective and quantitative data available to quantify the level of disturbance by this source to wilderness visitors. The numbers of permitted vessels are tightly controlled, and have remained basically unchanged for many years. In the professional judgment of park managers, this source of disturbance does not substantially diminish the condition of this indicator.

This potential source of disturbance has long been flagged as one of potential concern by park managers, yet there is not a complete understanding regarding the magnitude of its effect on wilderness visitors.

Facilities that Decrease Self-Reliant Recreation

Number of facilities that decrease self-reliant recreation

There are two floating ranger rafts in GBP that are in non-wilderness waters but are located immediately adjacent to wilderness lands. These two facilities are the only such facilities in the park, and have been present for many years; park managers believe these facilities are rarely used by visitors. Because of the low number of facilities and the low usage of these structures, park staff do not believe that they significantly diminish self-reliance of wilderness visitors.

Management Restrictions on Visitor Behavior

Number of required camping permits issued for GBP

Each group that plans to camp in the backcountry wilderness of GBP during the summer visitor season is required to obtain a permit in the front country and to attend a camper orientation prior to entering the wilderness. From 2011-2015, the average number of camper permits issued per year was 510 (NPS unpublished data). There has been remarkably low variation between years, with a maximum number of permits being 559 in 2011, and a minimum number of permits being 448 in 2012 (NPS unpublished data).

Number of camping/foot traffic restrictions

The average number of camping/foot traffic restrictions per year from 2011-2015 was 19-20, with no discernable trend across years (NPS unpublished data). A detailed description of the restrictions from 2011-2015 is provided below:

- 2011 – 14 closed islands in 10 areas (36 CFR 13.1178); four compendium island closures; one compendium camping closure; one emergency closure at the mouth of Geikie Inlet. A total of 20 restrictions.
- 2012 – 14 closed islands in 10 areas(36 CFR 13.1178); four compendium island closures, one compendium camping closure, one emergency closure at Vivid Lake outlet. A total of 20 restrictions
- 2013 – 14 closed islands in 10 areas (36 CFR 13.1178); four compendium island closures, one compendium camping closure. A total of 19 restrictions.
- 2014 – 14 closed islands in 10 areas (36 CFR 13.1178); four compendium island closures, one compendium camping closure, and one emergency closure at South Leland Island. A total of 20 restrictions.
- 2015 – 14 closed islands in 10 areas (36 CFR 13.1178); four temporary island closures, including one that added more closed land (the south ¼) to the previously closed northern ¾ of Leland Island, one emergency camping closure at Vivid Lake outlet. A total of 19 restrictions.

Threats and Stressor Factors

Historically, invasive species were not widespread in Alaska (NPCA 2008). The harsh climate, the remote and relatively undisturbed landscape, and the lack of pathways of introduction are commonly cited reasons why the impact of invasive species has been minimal in Alaska as compared to the rest of the United States (NPCA 2008). Currently the Glacier Bay Wilderness has relatively few known invasive species infestations; however, perennial sow thistle has been found on Strawberry Island (EPMT 2013). Other invasive species present in the park include the common dandelion, reed canary grass, and creeping buttercup (EPMT 2013).

Preservation of natural resources and improving visitor experience are key components of wilderness character (Landres et al. 2014). While activities such as boating, flightseeing, trail building, and other recreational facility development can enhance the visitor experience, they can also have negative impacts on wilderness character. Perhaps the most common activity that affects wilderness character involves the use of motor vehicles, motorized equipment, or mechanized transport (Landres et al.

2014). Most visitors to GLBA are never fully removed from the sounds or sight of aircraft or motorized boats (Mills and Bruno 2015). While the use of this mechanized transportation is permitted within the park, it is the primary threat to the Undeveloped Quality (Mills and Bruno 2015). To help offset some of this impact, most of the designated wilderness waters, and some non-wilderness waters, have seasonal restrictions on motorized vessel use (Mills and Bruno 2015). The daily numbers of cruise ships and private and commercial vessels are also regulated throughout Glacier Bay (Mills and Bruno 2015). Impacts to wilderness character from administrative vessel use will always be present (Landres et al. 2014). This use of motorized vessels and aircraft for administrative purposes has a negative impact on the Untrammeled Quality, the Undeveloped Quality, and the Solitude Quality (Landres et al. 2014). These impacts can be somewhat offset through their judicious use only when the benefits to wilderness stewardship outweigh the impacts (Landres et al. 2012). Recreational developments help protect the park's resources and improve the Natural Quality; they also negatively impact the opportunity for Solitude and Primitive or Unconfined Recreation Quality (Mills and Bruno 2015). While the presence of trails, shelters, signs, and bridges improve visitor safety, they also reduce the opportunity for Primitive Recreation Quality (Tricker et al. 2012). The presence of facility management, maintenance, or trail crews also has a negative impact on the opportunity for Solitude Quality (Tricker et al. 2012).

Changes in visitor use and visitor access to the GLBA wilderness area have the potential to affect all the wilderness qualities. Currently, visitor access to the wilderness is provided primarily by tour boat drop-offs, charter and private vessels, and aircraft (NPS 1989). Tour boat drop-off/pick-up is the most frequently used means of accessing the wilderness (NCPA 2008). Many of the tour vessels operating in Glacier Bay are allowed shore excursions by parties of 12 or less. This allows for more than 2,500 visitors to make landfall annually in the Glacier Bay wilderness (NPS unpublished data). An additional 1,000 visitor's kayak in Glacier Bay every year. The drop-off points along with the tidewater glaciers are popular destinations and tend to concentrate use in these areas (NPS unpublished data). Other than the Bartlett River and Bartlett Lake trails, no other man-made trails exist that could concentrate visitor use. Wilderness campers can choose their own campsites, and the sites generally appear to recover between uses (NPS unpublished data). Some hardened sites and trailing do exist in the more popular areas. Nevertheless, except for along the shoreline of Glacier Bay, there are almost no signs of people.

Data Needs/Gaps

Data needs and gaps have been identified throughout the foregoing sections. These include information about the park's natural resources (the Natural and Untrammeled Qualities of wilderness character), as well as information about human activities/developments and visitor experience (the Undeveloped and Solitude/Primitive/Unconfined Opportunity Quality). Park managers generally consider the latter "social" Qualities to be more vulnerable to degradation, compared to biophysical impacts to the Natural/Untrammeled Qualities by human activities. In 2018, GLBA will begin developing a Wilderness Stewardship Plan, and focused efforts to define and understand the park's detailed wilderness objectives have already started. Research will be undertaken to learn more about what GLBA wilderness visitors want, what they expect, and what they actually experience. The park will learn more about where park wilderness visitors go, how they use the wilderness, and what impacts they have on wilderness resources, as well as on their and other visitors' wilderness experiences.

Overall Condition

Natural Quality

During project scoping, the Natural Quality of wilderness was assigned a *Significance Level* of 3. Invasive plants and OA represent the most likely sources of significant impact to the Natural Quality component of GLBA's wilderness, but the magnitude of this impact is still poorly understood. The presence of invasive plants in GLBA is primarily limited to beaches in lower GBP and the immediately adjacent uplands. The impact of invasive plants in GLBA's wilderness is very spatially limited and affects only a small proportion of the wilderness.

There are additional impacts and changes to the Natural Quality of wilderness that are driven by human activity; any changes caused by human activities are deemed to be negative impacts to this measure. Examples of such anthropogenic changes in GLBA include warming temperatures and changing precipitation patterns (driven by anthropogenic climate change factors), contaminant introduction from distant sources, and the potential establishment of invasive animal species.

There is limited information currently available regarding both the potential threats to this measure, and the impacts that they may have. However, the general information that GLBA managers do have appear to suggest a current Natural Quality condition that is of good condition; a *Condition Level* of 1 was assigned to this measure.

Undeveloped Quality

The Undeveloped Quality of wilderness was assigned a *Significance Level* of 3 during project scoping. GLBA has recently taken steps to remove defunct installations in the park's wilderness. While some additional installations were installed to better inform the Natural Quality measure of wilderness (i.e., climate monitoring stations), the total number of installations and the overall impact from these stations is still reduced compared to where it was 10 years ago. Collectively, the installations have a minimal spatial, visual, and auditory footprint and minimally affect the Undeveloped Quality of GLBA's wilderness. Because of this, this measure was assigned a *Condition Level* of 0, indicating no current concern.

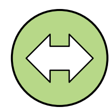
Untrammeled Quality

During project scoping, the Untrammeled Quality of GLBA's wilderness was assigned a *Significance Level* of 3. Untrammeled Quality continues to be near pristine in GLBA's wilderness, and instances of trammeling appear to be very minimal and spatially limited. For this reason, a *Condition Level* of 0 was assigned to this measure.

Solitude or Primitive and Unconfined Recreation Opportunity

Solitude or Primitive and Unconfined Recreation Opportunity in GLBA's wilderness was assigned a *Significance Level* of 3 during project scoping. GLBA's wilderness is vast, and the presence of humans, their works and manipulations, or their activities are generally very sparse, of low magnitude, and short in duration. The majority of human use in GLBA is concentrated along a thin strip of marine shoreline approximately 100-200 m (328-984 ft) wide. Additionally, most camper use of the park is concentrated near glacial attractions or near tour boat drop-offs. GLBA managers possess limited information regarding visitor expectations compared to their actual experiences in the park's wilderness (i.e., the experiential quality of wilderness character). Consequently, this measure was assigned a *Condition Level* of 1, indicating low concern at present.

<u>Weighted Condition Score</u>
Wilderness in GLBA was assigned a WCS of 0.17, indicating good current condition. The overall trend appears to be unchanging overall, but additional information regarding all the focal measures would provide managers with a more accurate current trend designation.

Wilderness			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.17**
Natural Quality	3	1	
Undeveloped Quality	3	0	
Untrammeled Quality	3	0	
Solitude or Primitive and Unconfined Recreation Opportunity	3	1	

4.24.6. Sources of Expertise
This component was completed with significant input and writing from many of GLBA's natural resource staff. Many of the sections in this component were either written by GLBA staff or were summarized by SMUMN based on existing GLBA notes and workbooks that were previously completed as part of another wilderness exercise ongoing at the park. The sources of expertise identified below represent the primary sources of input and writing in the sections above.

- Lewis Sharman, GLBA Ecologist
- Barbara Miranda, GLBA Wilderness Recreation Planner
- Lisa Etherington, GLBA Chief of Resource Management

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Chapter 5. Discussion

Chapter 5 provides an opportunity to summarize assessment findings and discuss the overarching themes or common threads that have emerged for the featured components. The data gaps and needs identified for each component are summarized and the role these play in the designation of current condition is discussed. Also addressed is how condition analysis relates to the overall natural resource management issues of the park.

5.1. Component Data Gaps

The identification of key data and information gaps is an important objective of NRCAs. Data gaps or needs are those pieces of information that are currently unavailable, but are needed to help inform the status or overall condition of a key resource component in the park. Data gaps exist for most key resource components assessed in this NRCA. Table 82 provides a detailed list of the key data gaps by component. Each data gap or need is discussed in further detail in the individual component assessments (Chapter 4).

Table 82. Identified data gaps or needs for the featured components.

Components	Data Gaps/Needs
Glaciers	• The continuation of laser altimetry analyses are needed to determine accurate rates of glacial thinning/advancement. • A lack of consistent monitoring from weather stations in GLBA prevent correlations of glacial change and precipitation/temperature changes from the 1950s to 2010.
Marine Shoreline	• Recent data regarding the distribution and abundance of marine debris in GBP are not available and debris surveys are needed. • Continued mussel contaminant sampling is needed to identify any potential long term trends. • Monitoring, or continued monitoring, of the visitor impacts (i.e., camping) and the current structure and velocity in GBP (and developing a water circulation model) are needed.
Harvested Marine Invertebrates	• The habitat status for both Tanner crabs and weathervane scallops is unknown. • Little is known about the impact that the weathervane scallop fishery gear has on benthic habitat for scallops and the overall benthic community. • Information regarding the occurrence and prevalence of WMS in District 16 is needed. Additionally, information regarding the incidence of BCS in Tanner crab stock in GBP is needed.
Murrelets	• Continuation of SEAN monitoring efforts are needed to track long-term trends in the area. • Murrelet diets and preferred prey species are poorly understood. • Continued research is needed regarding vessel disturbance and habitat use within GLBA. The nesting habitat of these species is cryptic, and areas of high nesting importance should be identified to ensure continued protection from disturbance.

Table 82 (continued). Identified data gaps or needs for the featured components.

Components	Data Gaps/Needs
Sea Ducks	• There has not been active monitoring of sea ducks in GLBA since 2001 and additional data regarding abundance trends in the GLBA region are needed. • Spatial and temporal variations in prey availability are poorly understood and additional research is needed to fully understand these fluctuations. • No data exist regarding winter age ratio of priority duck species or foraging effort of molting and wintering ducks • The effect of human disturbances during the molting period is poorly understood and requires additional research.
Breeding Landbirds	• There is currently no annual landbird monitoring in the park that documents presence, abundance, trends, or demography of breeding landbirds in GLBA.
Glaucous-winged Gulls	• Continued annual monitoring of glaucous-winged gull nesting colonies is needed, especially with legal gull egg harvest likely occurring in the park soon. This monitoring requirement is mandated in the LEIS ROD. • Continued monitoring of gull nesting habitat sites is needed, as nesting colony locations are likely to change as ecological succession progresses in the bay. • Annual harvest summaries are needed following the opening of the legal gull egg harvest period in GLBA.
Moose	• No distribution, abundance, population size, or population composition data exist for moose specific for the Dry Bay Preserve area – much of these data are only available for the Gustavus population. • Data specific to the four areas of interest are limited to 2012 surveys, and are in need of additional surveys.
Bears	• Little to no information exists regarding the brown and black bear populations of GLBA. Information regarding the distribution, home range size, population size, and movements of both species in the park is needed.
Mountain Goat	• Consistent, long-term monitoring is needed to better assess the current condition of this resource and to determine any apparent trends. • Winter kid survival data are needed, which would likely be best obtained during biannual surveys in the park during the fall and again in the late winter. • Ground surveys would provide a better assessment of age/sex ratios of mountain goats.
Harbor Seals	• Research into GLBA seal reproductive rates, survival rates, and the extent to which emigration may occur is needed to provide a better understanding of harbor seal population ecology in GLBA.
Humpback Whales	• There are no available data on prey abundance and distribution, nor is there information regarding the resilience of prey species to climate change and ocean acidification.
Sea Otters	• Data regarding foraging effort, prey diversity, and prey abundance and size are in need of expansion. • There are no available data regarding morphometrics, and limited data exist in regards to kelp abundance in GLBA. • Continuation of sea otter surveys and studies of intertidal benthic communities is needed.

Table 82 (continued). Identified data gaps or needs for the featured components.

Components	Data Gaps/Needs
Harbor Porpoise	• Continued yearly monitoring efforts are needed, and increased use of GPS to display specific locations of porpoises in the bay will allow for more accurate locations and approximations of preferred habitat.
Steller Sea Lions	• Information regarding age-specific survival rates and spatial distribution within GLBA is limited and in need of expansion. • Additional research is needed looking into the impacts that potential threats and stressors may have on Southeast Alaskan sea lions. • Further study of Steller sea lion diet in the GLBA region could be helpful in understanding the species' seasonal distribution.
Anadromous Fish	• Salmonid distribution and abundance data in GLBA represents a substantial data gap for most species. • The establishment of an anadromous fish monitoring program is needed to obtain abundance estimates. While these values are likely to be variable, monitoring of small index streams could be a less intensive option for monitoring abundance over time. • A more detailed look at terminal fisheries may provide a more accurate picture of the total number of removals in GLBA.
Mid-trophic Level Marine Forage Community	• No long-term studies have taken place that documents any trends in this community. Data regarding this community's biomass, density, and distribution are needed.
Pacific Halibut	• Data pertaining to halibut removals, size-at-age, and spawning stock biomass in GLBA waters are limited. The enormous amount of data that exists for this species cannot be narrowed exclusively to GLBA waters as reporting areas do not coincide with GLBA boundaries. • A more complete understanding of halibut movement and usage of GBP is needed. While some recent research has approached this topic, continued and expanded surveys are needed. • Research into what kind of ecological shifts may occur following the cessation of commercial fishing in GBP will likely be needed in the future. • Studies of site fidelity in GBP would allow for a more complete understanding of the species' diet in the bay and what their influence may be in the benthic community.
Plants	• Additional surveys are needed to expand knowledge regarding species richness and invasive species distribution within the park. Particularly at Excursion Ridge/Chilkat Range, Cape Spencer fjords, high elevations of the Dry Bay forelands and Alsek River corridor, and nunataks in the Fairweather Range. • Monitoring of spruce forests is needed as these stands come of an age where they are vulnerable to spruce beetle infestation. • Knowledge regarding shore pine and biological threats to the species is limited, particularly in regards to foliar and canker pathogens and pathogenic insects and fungi.

Table 82 (continued). Identified data gaps or needs for the featured components.

Components	Data Gaps/Needs
Western Toad	• Annual monitoring of western toads, especially in regards to abundance and distribution, is needed in the GLBA region. • Continued sampling of toads for chytrid fungus will allow managers to determine the current spread of the fungus in the area. • Research into the effects of climate change on toads is needed, especially in regards to shifts in temperature and precipitation patterns.
Air Quality	• At the time of writing, there are no active air quality monitors within GLBA, and there are no long-term data regarding the selected air quality metrics. • The establishment of long-term, consistent monitoring of lichen, atmospheric deposition of nitrogen, particulate matter, and visibility is needed. • SEAN is currently developing a monitoring plan for airborne contaminants that is slated for completion in 2018.
Water Quality	• Continuation of annual SEAN monitoring is recommended in order to identify any changes or trends in contaminant levels. • Further study of the potential threats to water quality from contaminants stored in GLBA's glaciers may help managers understand the likely impacts of a warming climate. • A repetition of the Reisdorph and Mathis (2014) study of aragonite saturation states in several years could provide insight into the rate of severity of ocean acidification in GBP.
Underwater Soundscape	• Existing data only covers the lower bay and not the majority of park waters. • No analysis has been done specific to the duration of noise-free intervals in the park's waters. • Analysis regarding the masking index of communication space for vocal species in underway, but is computationally intense and would need to be re-evaluated periodically. • The continuation of long-term monitoring is essential to assessing future condition and trends on how anthropogenic sources affect the underwater soundscape in the park. • Establishment of a baseline for the duration of noise-free intervals and the masking index of communication space for vocal species is needed.
Wilderness	• Additional information is needed regarding the park's Natural and Untrammeled Qualities, as well as information about human activities/developments and visitor experience (Undeveloped and Solitude/Primitive Unconfined Quality). • Completion of a Wilderness Stewardship Plan (scheduled for 2018). • Information regarding visitor use and experience in the GLBA's wilderness.

Several of the park's data needs involve the continuation or expansion of existing monitoring programs to accumulate enough data for identification of trends over time (e.g., glaciers, water quality, and sea otters). Many of the identified data gaps relate to expanding existing monitoring efforts to include broader parameters that may include how a component's prey community is structured (e.g., humpback whales, murrelets) or how the component may interact with its ecosystem (e.g., Pacific halibut, harvested marine invertebrates). Other components, such as breeding landbirds, sea ducks, the mid-trophic level marine forage community, and air quality, are in need of an established monitoring program. These components were not assigned condition in this NRCA due to a lack of recent data.

5.2. Component Condition Designations

Table 83 displays the conditions assigned to each resource component presented in Chapter 4 (definitions of condition graphics are located in Figure 184 following Table 83). It is important to remember that the graphics represented are simple symbols for the overall condition and trend assigned to each component. Because the assigned condition of a component (as represented by the symbols in Table 83) is based on a number of factors and an assessment of multiple literature and data sources, it is strongly recommended that the reader refer back to each specific component assessment in Chapter 4 for a detailed explanation and justification of the assigned condition. Condition designations for some components are supported by existing datasets and monitoring information and/or the expertise of NPS staff, while other components lack historic data, a clear understanding of reference conditions (i.e., what is considered desirable or natural) for some measures, or even current information. Condition could not be determined for four of the 24 selected components: sea ducks, breeding landbirds, mid-trophic level marine forage community, and air quality.

For featured components with available data and fewer information gaps, assigned conditions varied. Thirteen components are considered in good condition (Table 83); however, the marine shoreline, murrelets, glaucous-winged gulls, moose, anadromous fish, and water quality scores were at the edge of the good condition range and any small decline in the communities could shift them into the moderate concern range. Five components were of moderate concern (harvested marine invertebrates –Tanner crabs, Pacific halibut, western toad, plants, and underwater soundscape), and three components were considered to be of significant concern (glaciers, harvested marine invertebrates – weathervane scallops, and harbor seals) (Table 83).

Table 83. Summary of current condition and condition trend for featured NRCA components.

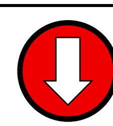
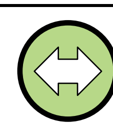
Component	WCS	Condition
Ecosystem Extent and Function		
Landcover		
Glaciers	0.87	
Marine Shoreline	0.33	

Table 83 (continued). Summary of current condition and condition trend for featured NRCA components.

Component	WCS	Condition
Biotic Composition		
Harvested Marine Invertebrates		
Tanner Crabs	0.46	
Weathervane Scallops	0.88	
Birds		
Murrelets	0.33	
Sea Ducks	N/A	
Breeding Landbirds	N/A	
Glaucous-winged Gulls	0.33	
Mammals		
Moose	0.50	
Bears	0.27	
Mountain Goat	0.10	
Harbor Seals	0.71	
Humpback Whales	0.29	

Table 83 (continued). Summary of current condition and condition trend for featured NRCA components.

Component	WCS	Condition
Biotic Composition (continued)		
Sea Otters	0.04	
Harbor Porpoise	0.21	
Steller Sea Lions	0.00	
Fishes		
Anadromous Fish	0.33	
Mid-trophic Level Marine Forage Community	N/A	
Pacific Halibut	0.54	
Herptiles		
Western Toad	0.50	
Plant Communities		
Plants	0.42	
Environmental Quality		
Air Quality	N/A	
Water Quality	0.33	

Table 83 (continued). Summary of current condition and condition trend for featured NRCA components.

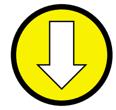
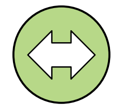
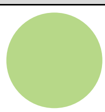
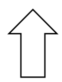
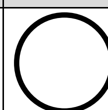
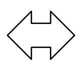
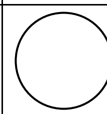
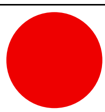
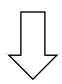
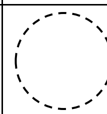
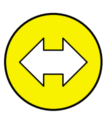
Component	WCS	Condition
Environmental Quality (continued)		
Underwater Soundscape	0.54	
Physical Characteristics		
Geologic & Hydrologic		
Wilderness	0.17	

Table 84. Description of symbology used for individual component assessments.

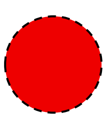
Condition Status		Trend in Condition		Confidence in Assessment	
	Resource is in Good Condition		Condition is Improving		High
	Resource warrants Moderate Concern		Condition is Unchanging		Medium
	Resource warrants Significant Concern		Condition is Deteriorating		Low

Examples of how the symbols should be interpreted:

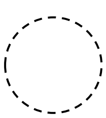
Resource is in good condition; its condition is improving; high confidence in the assessment.



Condition of resource warrants moderate concern; condition is unchanging; medium confidence in the assessment.



Condition of resource warrants significant concern; trend in condition is unknown or not applicable; low confidence in the assessment.



Current condition is unknown or indeterminate due to inadequate data, lack of reference value(s) for comparative purposes, and/or insufficient expert knowledge to reach a more specific condition determination; trend in condition is unknown or not applicable; low confidence in the assessment.

5.3. Park-wide Condition Observations

Despite the large expanse of land that GLBA covers, many of the resources discussed in this report are interrelated and share similar management concerns (e.g., data gaps, threats from outside the park and preserve, critical communities).

5.3.1. Climate Change

Climate change represents one of the largest and most significant threats and stressors to the priority resources within GLBA. It also represents an area that needs significant research in the future; this will be particularly useful as managers attempt to more fully understand how it may be affecting the priority resources in the GLBA area. In Alaska, climate is constantly fluctuating on multiple temporal scales, including several natural cycles such as the PDO. There is a scientific consensus that human activities, particularly those that produce greenhouse gasses, have contributed to a general warming trend in global climate (IPCC 2007). Current warming has accelerated natural processes that release greenhouse gases into the atmosphere, such as permafrost thawing and ebullition (methane bubbling) from northern lakes, further contributing to global warming (Anisimov 2007, Walter et al. 2007). The average annual temperature within GLBA is 1.9°C (35.4°F), and this average is expected to rise to 5.3°C (41.5°F) by 2080, an increase of about 0.6°C (1°F) per decade. Mean winter temperatures could rise more dramatically, by as much as 3.9°C (7°F) by 2080. Glacial expansion and subsequent deglaciation in the GLBA area has occurred relatively rapidly in geologic terms. However, the predicted temperature increases would even further accelerate glacial thinning and retreat within GLBA and possibly lead to the extirpation of several glaciers within the park (SNAP et al. 2009). In the last several decades, Glacier Bay's tidewater glaciers have been receding rapidly (Loso et al. 2014). Muir Glacier, for example, retreated over 7 km (4.3 mi) between 1973 and 1986; the glacier "grounded" and stopped producing icebergs in 1993 (Hall et al. 1995). In the late 1970s and early 1980s, 12 glaciers were calving icebergs into Glacier Bay (Molnia 2007). By 2008, only five glaciers were actively calving into the bay (Womble et al. 2010).

The loss of glacial ice in the park would affect a variety of other ecological communities as well. Glacial melting could introduce pollutants stored in glaciers into GLBA's fresh and marine waters (Bogdal et al. 2009), potentially affecting all organisms that rely on the park's water. Species that are linked to ice-associated habitats during a specific stage of their life history are typically the first to respond to climate change (Walther et al. 2002, Root et al. 2003, Kuletz et al. 2003). In GLBA, rapid glacial recession and changes in the number of tidewater glaciers may positively and/or negatively impact breeding and foraging habitats for murrelet species, especially during the summer (Kuletz et al. 2003, Piatt et al. 2011), but existing data are unable to resolve these questions. In addition, the acceleration of glacial recession will likely make more land available for succession and shoreline erosion (e.g., loss of land area), thus altering available habitat for ecological communities (NPS 2015b).

Diminishing glacial ice in the park, through both recession and thinning, is a potential threat to harbor seals' glacial haul out sites. At these glacial sites, seals rely on icebergs from tidewater glaciers for calving, molting, resting, and protection from predators (Calambokidis et al. 1987, Mathews and Pendleton 2006, Womble et al. 2010). Alaska's tidewater glaciers are predicted to continue thinning and/or receding, largely due to global climate change (Larsen et al. 2007). The loss of glacial haul out sites in GLBA may cause harbor seals to use terrestrial sites for breeding (increasing exposure to land-based predators) or to seek glacial sites in other areas outside of Glacier Bay (Womble et al. 2010, Allen et al. 2011).

One additional threat that is of great concern to park managers is OA. If global emissions of CO_2 continue to increase based on current trends, the average pH of the oceans could fall by as much as 0.5 units by the year 2100 (Raven et al. 2005). The pH of the oceans has already decreased from 8.17 to 8.09 over the past 200 years (Haufler et al. 2010). Species that are dependent upon calcium carbonate availability are directly impacted by OA, as the amount of calcium carbonate available to organisms decreases as the oceans become more acidic. Declines in these calcium carbonate dependent species (e.g., shellfish, mollusks, crustaceans, pteropods) could have indirect impacts on anadromous fish communities, Steller sea lions, forage fish communities, sea otters, and a suite of other marine predators in the park (PMEL 2008, Haufler et al. 2010, NRC 2010, OCB and EPOCA 2010, IWGOA 2011).

While anthropogenic CO_2 levels certainly contribute to ocean acidification in GLBA, it has been shown that OA in the park has been driven primarily by dissolved inorganic carbon and total alkalinity; the effects of these driving factors vary based on location and seasonality (Reisdorph and Mathis 2014). Because much of GBP is surrounded by glaciers and snow/ice fields, the meltwater directly enters the bay in several locations. The meltwater that enters the bay is typically low in total alkalinity, which enhances the bay’s vulnerability to reductions in pH levels (Reisdorph and Mathis 2014).

Changes in water and air temperature, as a result of global climate change, also represents a shared threat to the priority resource communities in GLBA. Predicted temperature increases in Southeast Alaska, especially in the winter months when precipitation could change from snowfall to rain, could have dramatic landscape-wide effects on watersheds (Hodgson and Quinn 2002, Mote et al. 2003, Bryant 2009). Salmon are affected by hydroclimatic factors at every life stage (Shanley and Albert 2014), and an alteration in temperature or flow regime due to climate change could have dramatic impacts on salmon in Southeast Alaska. Salmon fry and juveniles require appropriate stream flows to maintain high quality habitat to mature in (Quinn 2005); and at maturity, global ocean circulation patterns, prey availability, and appropriate stream flows and temperatures are required for successful rearing, migration and spawning (Quinn 2005). Additionally, changes in the normal temperature and precipitation within the park could have both direct and indirect effects on the mid-trophic level forage community of GLBA. Examples of direct impacts could include shifts in the timing and magnitude of freshwater input into GBP due to alterations in the timing and magnitude of snow, sea ice, and glacial melt. Change in the timing and magnitude of freshwater input and subsequent changes in circulation patterns within the bay (Etherington et al. 2007, Hill et al. 2009) would likely alter the timing, magnitude, and spatial patterns of primary production, thus altering all marine trophic levels in GBP (Robards 2014).

Climate is a primary driver of vegetation distribution and abundance (Hennon et al. 2012). Increased temperatures are likely to lengthen the growing season, impact plant phenology, and influence soil water availability (SNAP et al. 2009). While precipitation is projected to increase, evapotranspiration will also likely increase due to warmer temperatures and a longer growing season. This means water will be used by plants or will evaporate back into the atmosphere faster and will not be stored in the soil or on its surface as long. As a result, the area will seem drier, particularly in summer and fall. This could increase wildfire risk and contribute to wetland drying (SNAP et al. 2009). Changes in vegetation as a result of these climate shifts will also impact wildlife distribution and habitat use (Moynahan et al. 2008).

Coastal Alaskan forests are predicted to experience the largest increase in frost-free days this century of any location in North America (Meehl et al. 2004). Winter temperatures will likely increase from an average just below freezing during the twentieth century to an average above freezing, which would reduce snow accumulation (Hennon et al. 2012). As a result, yellow-cedar decline will likely expand into new areas where snow cover is insufficient and may eventually impact stands further upslope (Beier et al. 2008). If yellow-cedar stands are impacted in GLBA, this may also influence wildlife due to changes in forage and habitat (NPS 2013b, Oakes et al. 2015).

Climate also influences the diseases and insect pests that impact many tree species in Southeast Alaska. Weather plays a key role in bark beetle population dynamics, which favor warm, dry weather, particularly in spring. The life cycle of the spruce beetle is typically 2 years, but warmer and longer seasons can allow the beetles to complete their life cycles in 1 year (Holsten 1999). Projected increased temperature or other changes that prolong needle wetness during the growing season could also favor pathogens that affect shore pine, such as western gall rust and Dothistroma needle blight (VanLeuven 2014; Mulvey, written communication, 7 May 2015), as needle blight tends to be driven by temperature and moisture. Severe outbreaks in British Columbia have been linked to local increases in summer precipitation by one study (Woods et al. 2005) and to increased August minimum temperatures and increased spring precipitation in another (Welsh et al. 2014).

Pathogen outbreaks due to warming from climate change is not limited to plant species, however, as warmer summers and shorter winters could favor the spread of parasites such as ticks and the diseases they carry to moose (Sinnott 2013). While parasite outbreaks are uncommon in Southeast Alaska, moose populations in the lower 48 states have experienced an alarming decline in recent years; while the exact cause is unknown, two of the leading suspects are increased parasite loads and heat stress, both linked to shorter and warmer winters (Robbins 2013, Sinnott 2013). The western toad is also likely to be affected by temperature-driven pathogen outbreaks. Kiesecker et al. (2001) linked amphibian pathogen outbreaks to climate-induced changes in UV-B exposure. The decreased water depths from precipitation pattern shifts were exposing egg masses to the harmful UV-B rays; this left the eggs susceptible to disease (Kiesecker et al. 2001). *Saprolegina farax* infection has caused large number of western toad egg mortalities in shallow lakes and ponds of western North America (Pounds 2001).

5.3.2. Vessel Traffic

The remote location of GLBA necessitates visitors to use either plane or watercraft to access the park. The majority of visitors in GLBA arrive via cruise ship, which represents the primary source of anthropogenic disturbance in the park and a threat to many of the priority resources discussed in this

NRCA. While the maximum daily number of cruise ships allowed to enter GBP during the peak season (June-August) is two, potential pollution generated by these vessels, disturbance to biotic communities and wilderness character, and vessel strikes on wildlife are three of the primary threats presented by vessel traffic in the park. Vessels are potential sources of contaminants through chemical spills (e.g., fuel, lubricating oils, or hydraulic fluid), wastewater discharge, or ballast water release (NPS 2003, Wuebben et al. 2000). Many fuels and other oils are persistent chemicals, meaning the contamination could be transported long distances by wind and/or currents (Eley 2000). The NPS maintains a vessel fueling and underground petroleum storage facility at Bartlett Cove near the public dock, where fuel spills could occur and threaten the park's water quality (Wuebben et al. 2000). The potential air quality pollution from vessels is a concern as well, but has been poorly studied and documented to date. Mölders et al. (2013) identified that cruise ships emitted an average of $2.5 \mu g/m^2/s$ PM in the bay. The PM emissions and visibility varied depending on cruise ship speed, with emissions tending to reduce when cruise ships travelled at a constant 6.69 m/s in the bay as opposed to travelling at slow or varying speeds (Mölders et al. 2013). Additionally, Pirhalla et al. (2014) found that strong inversion events in the area tended to trap pollutants from cruise ships and increased PM_{10} concentrations by almost 43% (compared to non-inversion concentrations). Inversion events during Pirhalla et al. (2014) occurred on just under half (42%) of all cruise ship entry days.

Vessel traffic also has the potential to disturb several ecological communities and priority species in the park. Many avian species, such as breeding and non-breeding murrelets, are disturbed by vessels. Agness et al. (2013) found that Kittlitz's murrelets were more likely to fly away from vessels which results in significant energy expansion. Added energy requirements resulting from repeated disturbances could impact the population dynamics of murrelets by lowering reproductive success and or survival. Marcella (2014) found that up to 72% of murrelets passing within 850 m (2788 ft) of a cruise ship were disturbed (dove or flew in response) although murrelets were also disturbed when approached by ships at distances >1000 m (3280 ft). Additionally, vessel traffic has been shown to alter behavior of adult sea ducks and lower survival of ducklings in nesting sea ducks species (Åhlund and Götmark 1989, Keller 1991). Further, vessel traffic has the ability to alter a duck's behavior, habitat selection, and energy expenditure (Tuite et al. 1983, Bell and Austin 1985, Galicia and Baldassarre 1997).

Mammalian species in GLBA are also affected by vessel traffic, with the potential for vessel disturbance most likely greatest among marine mammals. The park has enacted several efforts to minimize the potential for disturbance (e.g., closure areas, approach distance requirements, speed limits), yet the threat persists. Vessel disturbance can cause hauled-out seals to flush into the water, which may result in increased energy expenditure or increase the risk of mother/pup separation (Lewis and Mathews 2000, Young et al. 2014). Vessel disturbance at haul out sites is also a concern for Steller sea lions in GLBA. Repeated disturbance while at haul outs or rookeries may reduce the amount of time sea lions spend resting, disrupt social interactions, and temporarily separate mothers from dependent pups (Mathews 2000). Other priority marine species affected by vessel disturbance include humpback whales, sea otters, and harbor porpoise. Vessel traffic is also known to disturb terrestrial mammals. For example, Lewis (2010) found that vessels approaching within 100 m (328 ft) of a bear significantly increased the frequency of stress behaviors exhibited by the bear. A majority of bears that were approached within this distance fled short distances and some completely left the beach (Lewis 2010, NPS 2013a).

Vessel traffic also has detrimental effects on the park's soundscape (both underwater and atmospheric) and wilderness character. Disturbance from underwater vessel noise can cause stress to humpback whales in GLBA, which may influence whale distribution in the park. Humpback whales are adapted to rely on sound for everyday activities such as feeding, intraspecific location, intraspecific communication, and detection of predators (Gabriele et al. 2010, Holt et al. 2011); vessel traffic noise interferes with these activities. Atmospherically, motorized vessels and aircraft have a negative impact on the untrammeled quality, the undeveloped quality, and the solitude quality of the park (Landres et al. 2014).

Vessel strikes, whether lethal or non-lethal, are another threat to many aquatic species in GLBA. Vessel strikes of humpback whales are a concern in GLBA, and Neilson et al. (2012) identified the mouth of Glacier Bay as a "hotspot" with elevated risk of whale-vessel collisions. While whale strikes occur from vessels of all sizes (i.e., not exclusively from cruise ships); strikes from small vessels appear to be the most common (Neilson et al. 2012). In addition, trauma from boat collisions and lacerations from propellers could result in mortality or increased vulnerability of sea otters due to injury (Riedman and Estes 1990). Other priority species at risk from vessel strikes include avian species such as murrelets, sea ducks, or even landbirds that may collide with windows/lights on vessels.

5.3.3. Plant Succession

Isostatic rebound and vegetation succession continue to alter the landscape of GLBA, and while this represents a natural and ongoing process, it will affect many of the priority resource communities in the park as they continually adapt to changing habitat conditions (Chapin et al. 1993). Breeding landbirds are faced with constantly shifting preferred nesting habitats, especially species that prefer early successional vegetation communities. Similarly, the ongoing vegetation succession and isostatic rebound continually creates and removes nesting habitat for the gulls. Ecological succession occurring on the islands in the park have resulted in many historic nesting locations transitioning from the gull-preferred open meadows and rocky habitats (early successional stage) to closed conifer forests (late-successional stage), which are not suitable for nesting colonies. Transitions like this are to be expected at many of the islands in the park and will eventually preclude gull nesting at historic locations (NPS 2010b).

Vegetation in early to mid-successional stages (e.g., shrubs) provides critical forage for moose populations (Stephenson et al. 2006). Later successional stages, where trees are more dominant, support fewer moose browse species, but do provide valuable cover in the winter and during calving season (Stephenson et al. 2006). If the successional stages that provide forage were lost, the moose population would be negatively impacted. The distribution of bears in GLBA is likely related to food resource availability in different vegetation and stream successional stages, and bear distribution can vary between years in response to food availability (Publicover 1985, Lewis 2012). Black bears could expand their range north as the landscape in northern GBP matures from early, open successional stages to closed scrub and forest (Lewis 2012). Lewis (2012) noted that brown bear use of recently deglaciated (50-150 years) areas was high, perhaps due to the presence of important plant food sources. As plant succession progresses, these species will be replaced by others, creating habitat that may not be as suitable for brown bears.

5.3.4. Overall Conclusions

GLBA represents one of the largest expanses of wilderness and untouched landscapes in the U.S. and possesses incredible scientific significance in regards to glacial retreat/advancement rates, the ongoing ecological succession following deglaciation, and the tremendous diversity of aquatic and terrestrial wildlife. While several park-wide threats and stressors are certainly present, and the continued threat of climate change looms, many of GLBA’s priority resources remain in good or stable condition. Continued monitoring of these priority resources, combined with management efforts directed at minimizing threats will aide many of these communities and maintain their presence and integrity within the park. As described originally in the park’s enabling legislation, GLBA possesses tremendous opportunity for scientific study, and continuing advancements in the scientific understanding of GLBA’s resources will not only benefit the park, but also other locales with shared resources.

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Appendices

Appendix A. Annual rates of elevation change in several glaciers found in GLBA.

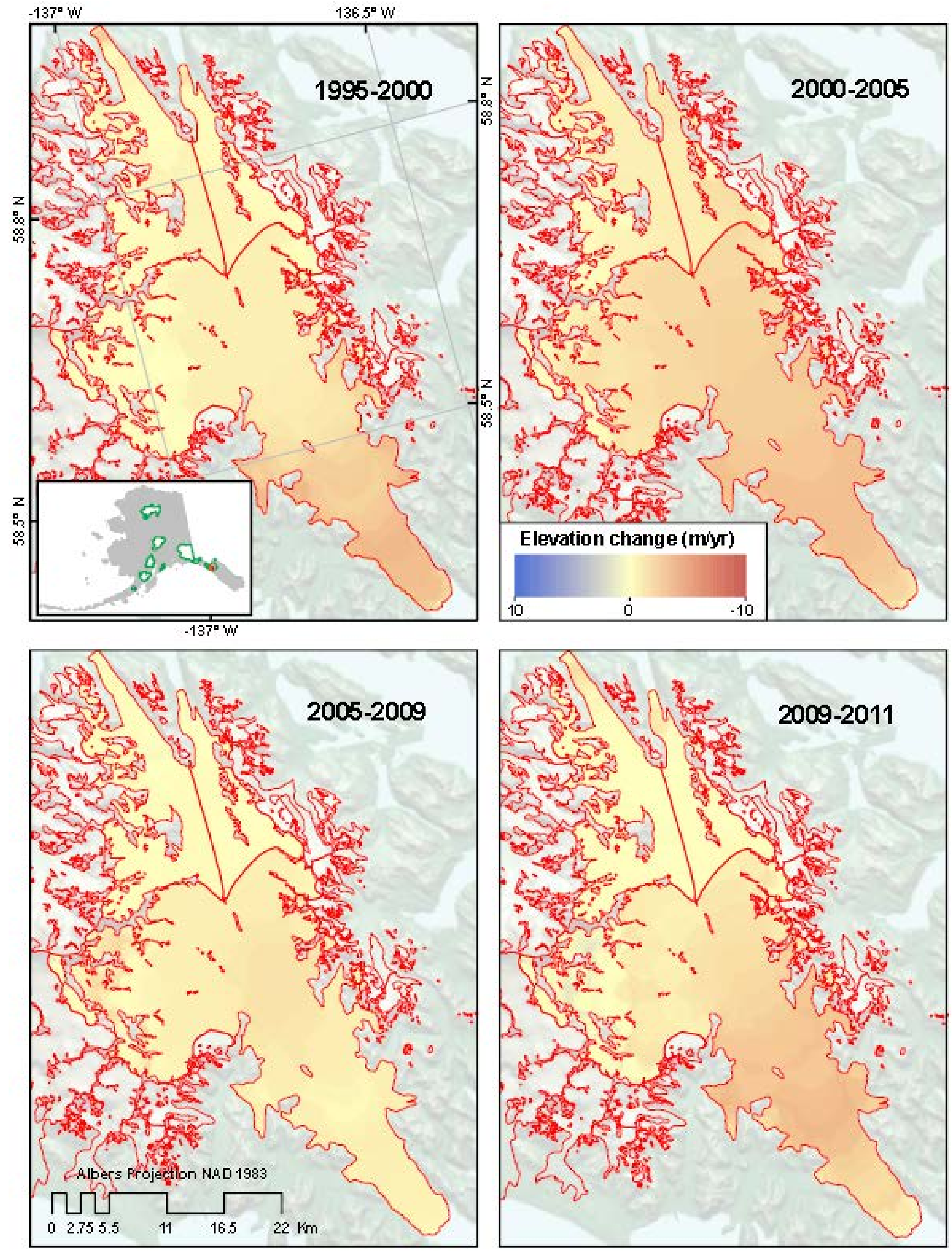

Figure A-1. Annual rates of elevation change for Brady, Lamplugh, and Reid Glaciers in GLBA. Values are averages over the indicated time intervals (Loso et al. 2014).

Appendix A (continued). Annual rates of elevation change in several glaciers found in GLBA.

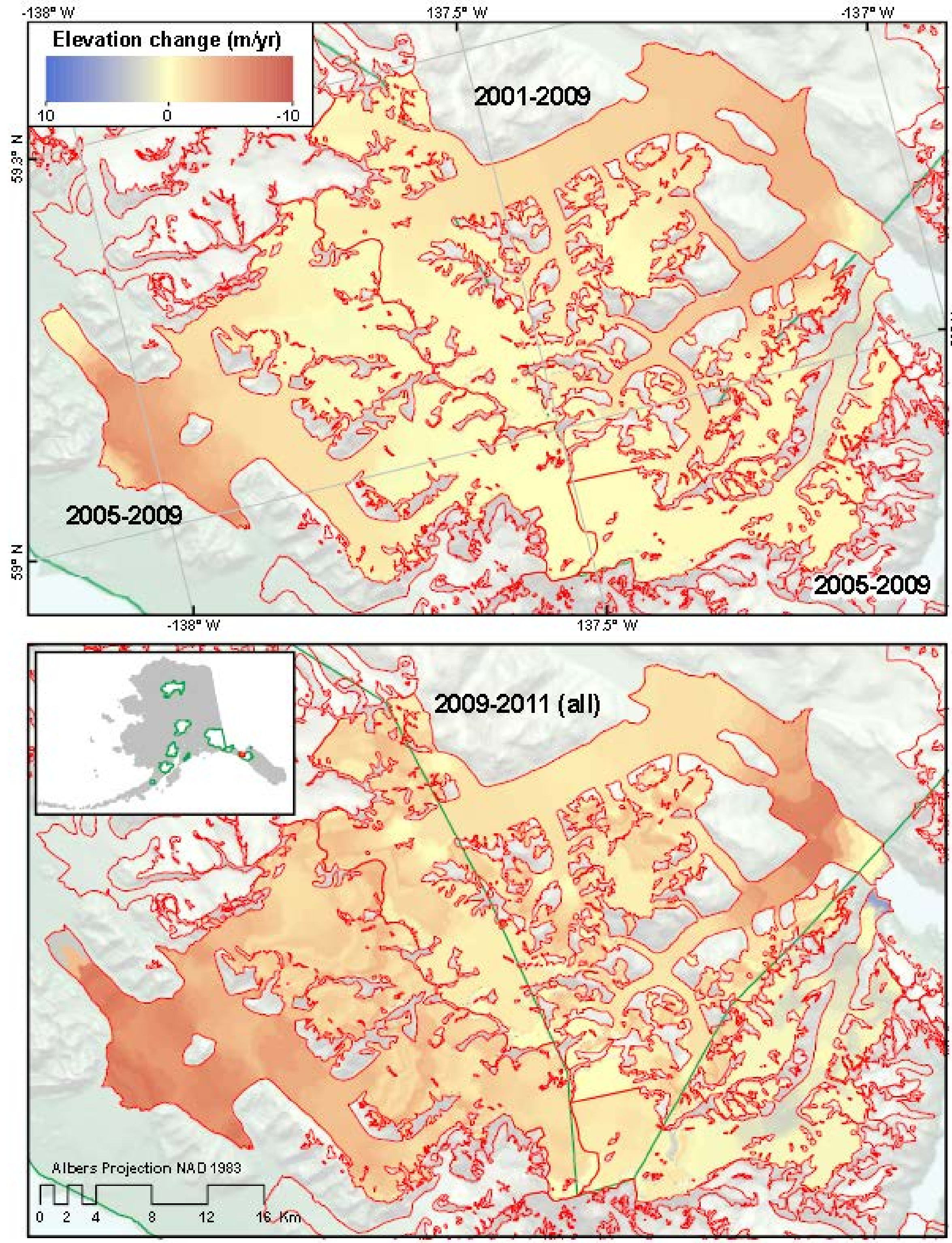

Figure A-2. Annual rates of elevation change for Margerie, Grand Plateau, Konamoxt, Melbern and Grand Pacific Glaciers in GLBA. Values are averages over the indicated time intervals (Loso et al. 2014).

Appendix A (continued). Annual rates of elevation change in several glaciers found in GLBA.

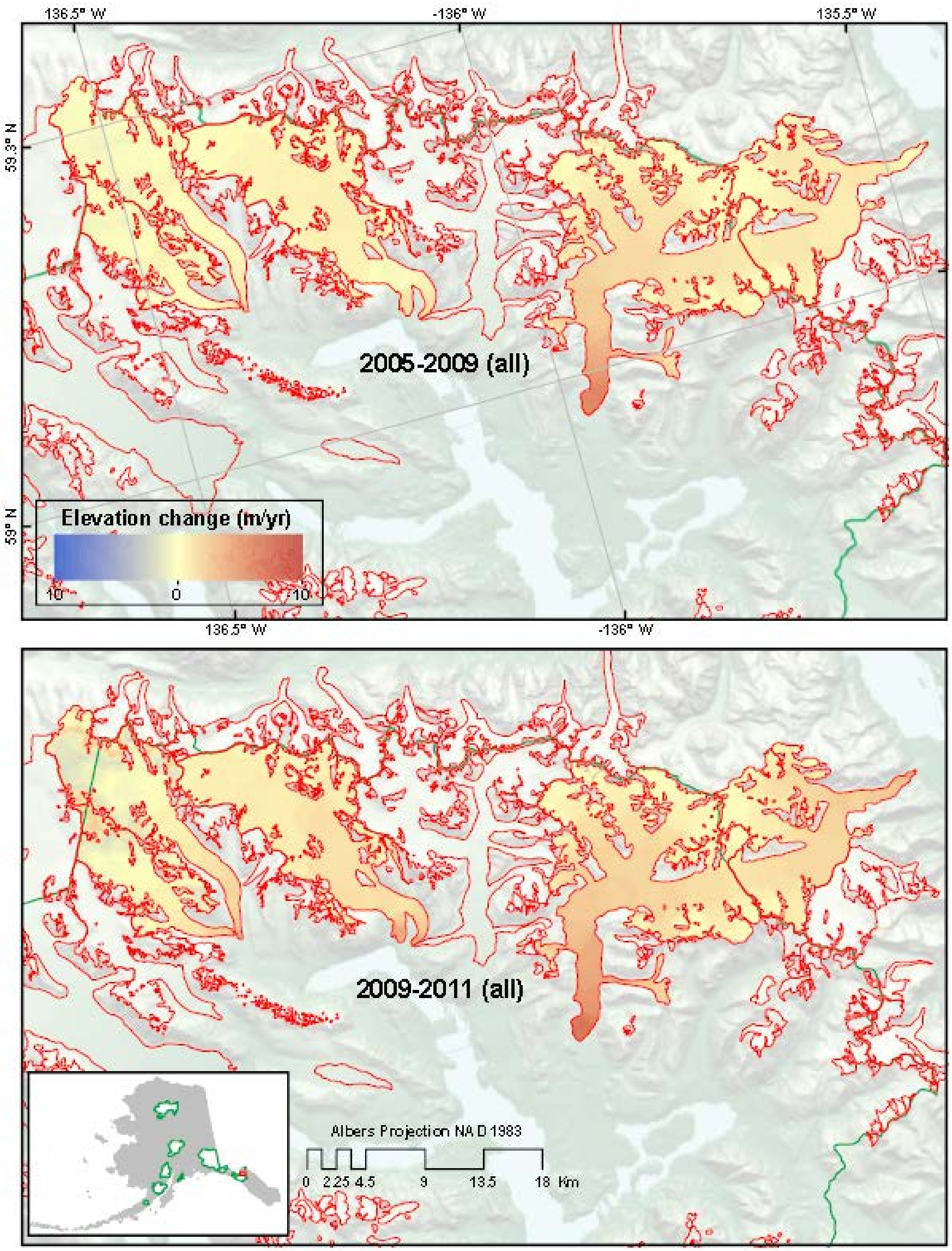

Figure A-3. Annual rates of elevation change for Muir, Riggs, Carroll, Casement, and Davidson Glaciers in GLBA. Values are averages over the indicated time intervals (Loso et al. 2014).

Appendix B. Survey designs and methods used to estimate populations size or trend of Kittlitz's murrelet within GLBA from 1987-2010 (Kirchhoff 2011).

Years	Month(s)	Method(s)	Strip Width (m)	Offshore Transect Orientation(s)	Shoreline Transects (m to shore)	Flying Bird Count Method(s)	Forward Window (W x L, m)	Source
1987 1989 1991	May-Aug	Strip	200	–	Variable	Continuous	200 x 50	Duncan and Climo (1991)
1991	June-July	Strip	200	Crossing	100	Continuous	200 x 200	Drew and Piatt (2008)
1993	June	Strip	300	Zigzag	–	Continuous	300 x 150	Lindell (2005)
1999-2003	June	Strip	200, 300	Crossib	100, 150	Continuous	200 x 200, 300 x 300	Drew et al. (2008)
2007	July	Line & Strip	200	Perpendicular	–	–	–	Kirchhoff (2008)
2008	June	Line	-	Crossing	100	Continuous	200 x 100	Arimitsu and Piatt (unpub. Data)
2009	July	Strip	300	Zigzag	–	Continuous	300 x 150	Kirchhoff et al. (2010)
2012	July	Line & Strip	300	Zigzag	–	Snapshot & Continuous	300 x 150	Kirchhoff and Lindell (2011)
2009-2010	July	Line	–	Perpendicular & Zigzag	–	Continuous	400 x 200	Hoekman et al. (2011a, b)

Appendix C. Breeding landbird species observed in GLBA from 1965-2004. Studies denoted with an * indicate a species checklist where no on the ground surveys took place.

Common Names	Scientific Name	NPS (2014)*	Trautman (1966)	Wik (1967)*	Muldoon (1987)	Willson and Nichols (1997)	Willson and Gende (1998)	Saracco and Gende (2004)
northern goshawk	*Accipiter gentilis*	X		X				
sharp-shinned hawk	*Accipiter striatus*	X		X				
red-tailed hawk	*Buteo jamaicensis*	X		X				
northern harrier	*Circus cyaneus*	X		X				
bald eagle	*Haliaeetus leucocephalus*	X	X	X	X			X
rufous hummingbird	*Selasphorus rufus*	X	X	X			X	X
Eurasian collared-dove	*Streptopelia decaocto*	X						
belted kingfisher	*Ceryle alcyon*	X	X	X				
peregrine falcon	*Falco peregrinus*	X		X				
merlin	*Falco columbarius*	X		X				
sooty grouse (blue grouse)	*Dendragapus fuliginosus*	X	X	X				X
willow ptarmigan	*Lagopus*	X	X	X				
white-tailed ptarmigan	*Lagopus leucura*	X		X				
rock ptarmigan	*Lagopus muta*	X	X	X	X			
Cedar Waxwing	*Bombycilla cedrorum*	X						
brown creeper	*Certhia americana*	X		X				X
American dipper	*Cinclus mexicanus*	X		X				
northwestern crow	*Corvus caurinus*	X	X	X	X	X		X
common raven	*Corvus corax*	X	X	X	X	X		X
Steller's jay	*Cyanocitta stelleri*	X		X				
dark-eyed junco	*Junco hyemalis*	X	X	X		X	X	X
Lincoln's sparrow	*Melospiza lincolnii*	X	X	X	X	X		X
song sparrow	*Melospiza melodia*	X	X	X				

Appendix C (continued). Breeding landbird species observed in GLBA from 1965-2004. Studies denoted with an * indicate a species checklist where no on the ground surveys took place.

Common Names	**Scientific Name**	**NPS (2014)***	**Trautman (1966)**	**Wik (1967)***	**Muldoon (1987)**	**Willson and Nichols (1997)**	**Willson and Gende (1998)**	**Saracco and Gende (2004)**
savannah sparrow	*Passerculus sandwichensis*	X	X	X	X	X		
fox sparrow	*Passerella iliaca*	X	X	X	X	X	X	X
snow bunting	*Plectrophenax nivalis*	X	X	X	X			
golden-crowned sparrow	*Zonotrichia atricapilla*	X	X	X			X	
common redpoll	*Carduelis flammea*	X	X	X	X	X		X
pine siskin	*Carduelis pinus*	X	X	X		X	X	X
gray-crowned rosy-finch	*Leucosticte tephrocotis*	X	X	X	X			
White-winged crossbill	*Loxia leucoptera*	X		X				
red crossbill	*Loxia curvirostra*	X		X			X	X
pine grosbeak	*Pinicola enucleator*	X	X	X			X	X
barn swallow	*Hirundo rustica*	X	X	X	X			
American cliff Swallow, cliff swallow	*Petrochelidon pyrrhonota*	X						
bank swallow	*Riparia riparia*	X	X	X	X			
tree swallow	*Tachycineta bicolor*	X	X	X	X		X	
violet-green swallow	*Tachycineta thalassina*	X	X	X				
red-winged blackbird	*Agelaius phoeniceus*	X		X				
chestnut-backed chickadee	*Poecile rufescens*	X	X	X		X	X	X
American pipit, water pipit	*Anthus rubescens*	X	X	X	X	X		
yellow-rumped warbler	*Dendroica coronata*	X	X	X		X	X	X
yellow warbler	*Dendroica petechia*	X	X	X	X	X	X	X
Townsend's warbler	*Dendroica townsendi*	X	X	X		X	X	X
common yellowthroat	*Geothlypis trichas*	X	X	X				
orange-crowned warbler	*Vermivora celata*	X	X	X	X	X	X	X

Appendix C (continued). Breeding landbird species observed in GLBA from 1965-2004. Studies denoted with an * indicate a species checklist where no on the ground surveys took place.

Common Names	**Scientific Name**	**NPS (2014)***	**Trautman (1966)**	**Wik (1967)***	**Muldoon (1987)**	**Willson and Nichols (1997)**	**Willson and Gende (1998)**	**Saracco and Gende (2004)**
Wilson's warbler	*Wilsonia pusilla*	X	X	X	X	X	X	X
ruby-crowned kinglet	*Regulus calendula*	X	X	X		X	X	X
golden-crowned kinglet	*Regulus satrapa*	X	X	X		X		X
red-breasted nuthatch	*Sitta canadensis*	X		X				
winter wren (pacific wren)	*Troglodytes troglodytes*	X	X	X		X	X	X
hermit thrush	*Catharus guttatus*	X	X	X	X	X	X	X
gray-cheeked thrush	*Catharus minimus*	X	X	X		X	X	X
Swainson's thrush	*Catharus ustulatus*	X	X	X			X	X
varied thrush	*Ixoreus naevius*	X	X	X		X	X	X
American robin	*Turdus migratorius*	X	X	X		X	X	X
alder flycatcher	*Empidonax alnorum*	X						
olive-sided flycatcher	*Contopus cooperi*	X	X	X				
Pacific-slope flycatcher	*Empidonax difficilis*	X		X		X	X	X
northern flicker	*Colaptes auratus*	X						
American three-toed woodpecker	*Picoides dorsalis*	X		X			X	
downy woodpecker	*Picoides pubescens*	X	X	X				X
hairy woodpecker	*Picoides villosus*	X		X			X	X
red-breasted sapsucker	*Sphyrapicus ruber*	X		X				X
great horned owl	*Bubo virginianus*	X	X	X				
northern saw-whet owl	*Aegolius acadicus*	X						
short-eared owl	*Asio flammeus*	X	X	X				
	Total # of Species	**67**	**43**	**61**	**18**	**23**	**24**	**31**

Appendix D. Breeding landbird species richness and abundance values, by survey site, observed during Willson and Nichols (1997) survey of the east arm of Glacier Bay.

Common Name	Barlett Cover	Stump Cover	Wachusett Inlet 1	Wachusett Inlet 2	Wachusett Inlet 3	Wachusett Inlet 4	Hunter Cove 1	Hunter Cove 2	Hunter Cove 3	Hunter Cove 4	Adams Inlet 1	Ad In
yellow-rumped warbler	4	–	–	–	–	–	–	4	1	1	–	
yellow warbler	–	7	–	3	3	–	2	–	–	4	–	
orange-crowned warbler	–	9	4	–	2	1	3	–	4	5	2	
Wilson's warbler	1	6	2	1	3	1	10	1	4	7	1	
Townsend's warbler	3	–	–	–	–	–	–	–	–	–	–	
American robin	3	–	–	–	–	–	–	–	–	–	5	
gray-cheeked thrush	–	1	–	–	–	–	5	–	1	—	–	
hermit thrush	8	2	–	2	3	–	2	7	2	3	5	
varied thrush	8	–	–	–	–	–	1	1	1	–	–	
fox sparrow	–	10	11	9	11	13	6	–	3	3	1	
savannah sparrow	–	–	6	–	–	1	–	–	–	–	4	
Lincoln's sparrow	–	1	–	–	–	–	–	–	–	–	–	
golden-crowned kinglet	3	–	–	–	–	–	–	–	1	–	–	
ruby-crowned kinglet	3	1	–	–	–	–	–	6	5	–	–	
chestnut-backed chickadee	1	–	–	–	–	–	–	2	–	–	–	
pine sisken	2	–	–	–	1	–	2	3	–	–	–	
common redpoll	–	2	–	1	8	3	–	–	–	–	2	
dark-eyed junco	6	–	–	–	–	–	–	2	–	–	1	
winter wren	8	–	–	–	–	–	–	1	3	–	–	
western flycatcher	2	–	–	–	–	–	–	3	4	–	–	
American pipit	–	–	–	–	–	1	–	–	–	–	–	
common raven	2	–	–	–	–	–	–	–	–	–	–	
northwestern raven	–	–	–	–	–	–	–	2	–	–	2	

* numbers were rounded to the nearest whole number.

Appendix D (continued). Breeding landbird species richness and abundance values, by survey site, observed during Willson and Nichols (1997) survey of the east arm of Glacier Bay.

Common Name	Barlett Cover	Stump Cover	Wachusett Inlet 1	Wachusett Inlet 2	Wachusett Inlet 3	Wachusett Inlet 4	Hunter Cove 1	Hunter Cove 2	Hunter Cove 3	Hunter Cove 4	Adams Inlet 1	Ad Inl
Total Number of Individuals	**54**	**39**	**23**	**16**	**31**	**20**	**31**	**32**	**29**	**23**	**23**	**3**
Total Number of Species	**14**	**9**	**4**	**5**	**7**	**6**	**8**	**11**	**11**	**6**	**9**	**1**

* numbers were rounded to the nearest whole number.

Appendix E. Breeding landbird species and number of individuals observed in GLBA at Delta Station during Trautman (1966).

Common Name	June																				July				
	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	1	2	3	4	5
	Delta Station*																								
bald eagle	5	2	–	1	–	1	2	1	1	2	3	–	4	1	3	1	3	1	1	3	–	–	–	–	2
blue grouse (sooty grouse)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
willow ptarmigan	–	–	–	–	–	–	–	2	1	1	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–
rock ptarmigan	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	2	1	–	–	–	–	–	–	–	–
great-horned owl	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
short-eared owl	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
rufous hummingbird	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
belted kingfisher	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
downy woodpecker	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–
olive-sided flycatcher	–	–	–	–	–	–	–	1	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–
violet-green swallow	–	–	–	–	–	–	–	2	–	4	–	–	–	–	1	–	–	–	1	–	–	–	–	–	–
tree swallow	5	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	2
bank swallow	–	–	–	–	–	–	–	–	–	–	–	12	–	–	–	–	–	–	–	–	–	–	–	–	–
barn swallow	16	2	2	10	8	3	2	8	2	2	2	1	3	2	3	1	2	–	4	1	5	6	2	1	2
common raven	7	3	3	2	1	2	5	3	2	4	5	3	2	5	4	5	6	–	5	6	5	2	11	3	3
northwestern crow	3	–	–	–	–	–	–	–	–	–	23	–	–	2	–	–	–	–	–	–	–	–	1	–	15
chestnut-backed chickadee	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	4	–	–	–	–	–	–	–	–	–
winter wren	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
robin	1	–	2	2	2	2	2	2	2	2	2	–	1	2	2	2	–	–	–	–	–	–	–	–	–
varied thrush	1	–	–	3	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	3
hermit thrush	–	–	–	12	12	1	–	5	–	3	5	1	3	10	1	–	1	2	–	–	1	–	–	–	12
Swainson's thrush	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
gray-cheeked thrush	–	–	4	25	3	–	–	10	–	2	4	–	1	12	–	–	2	1	–	–	1	–	–	1	2

* Beside Muir Inlet beach, 1.5 km north of the base of Klotz Hill. Area within 6 km of camp was investigated.

Appendix E (continued). Breeding landbird species and number of individuals observed in GLBA at Delta Station during Trautman (1966).

Common Name	June																				July				
	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	1	2	3	4	5
	Delta Station*																								
golden-crowned kinglet	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
ruby-crowned kinglet	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
water pipit (American pipit)	–	–	2	8	2	1	–	20	–	15	8	10	–	10	8	5	8	15	2	10	1	20	–	3	–
orange-crowned warbler	–	–	3	10	4	–	1	18	1	20	15	16	5	10	1	1	7	10	4	10	2	8	–	3	2
yellow warbler	–	–	8	40	20	–	2	5	–	–	3	4	3	–	2	–	4	4	–	1	2	5	–	2	2
Yellow-rumped warbler	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Townsend's warbler	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
common yellowthroat	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Wilson's warbler	–	–	2	10	15	–	–	15	1	4	18	10	8	8	5	2	5	3	–	–	3	–	–	4	5
pine grosbeak	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	3
gray-crowned rosy finch	–	–	6	3	12	1	8	10	3	10	5	–	3	12	10	15	12	15	2	48	8	10	–	30	15
common redpoll	1	3	30	30	15	15	20	35	10	15	15	25	15	30	20	25	16	50	4	10	8	20	–	20	10
pine siskin	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
savannah sparrow	4	1	5	8	12	3	2	30	3	8	4	8	2	10	3	2	8	10	10	12	10	–	2	10	8
Oregon junco (dark-eyed junco)	1	–	1	–	–	–	–	1	–	2	–	–	–	2	–	–	1	–	–	–	–	–	–	–	2
golden-crowned sparrow	–	–	–	–	–	–	–	–	–	–	–	1	–	4	–	–	–	–	–	–	–	–	–	–	–
fox sparrow	–	1	10	40	60	2	15	50	10	25	25	20	10	15	20	12	15	25	10	18	19	15	2	23	15
Lincoln's sparrow	–	–	–	–	–	–	1	–	–	–	–	–	–	3	–	–	–	–	–	–	–	1	–	10	–
song sparrow	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
snow bunting	–	–	–	2	–	–	–	27	–	24	–	–	–	–	–	4	–	–	–	–	–	28	–	–	–

* Beside Muir Inlet beach, 1.5 km north of the base of Klotz Hill. Area within 6 km of camp was investigated.

Appendix F. Breeding landbird species and number of individuals observed in GLBA at Muir Cabin, Nunatak Station, Bartlett Cove, and an unspecified location during Trautman (1966).

Common Name	July																									
	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
	Muir Cabin[A]							Nunatak Station[B]							Unspecified Location									Bartlett Cove[C]		
bald eagle	2	2	4	4	1	1	–	2	2	1	1	–	–	–	–	–	–	–	1	–	–	5	–	1	4	8
blue grouse (sooty grouse)	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
willow ptarmigan	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	28	–	–	7	12	–	–	–
rock ptarmigan	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
great-horned owl	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–
short-eared owl	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
rufous hummingbird	–	–	–	–	1	3	–	–	–	–	–	–	1	1	2	1	4	1	3	8	7	1	–	1	2	1
belted kingfisher	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	1	–
downy woodpecker	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
olive-sided flycatcher	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
violet-green swallow	–	–	–	–	–	–	–	–	–	–	–	–	2	–	–	–	–	–	–	–	–	–	–	–	–	–
tree swallow	–	3	–	–	2	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2	–
bank swallow	–	–	–	–	–	–	–	–	–	–	–	–	–	–	4	–	8	–	6	–	–	–	–	–	–	–
barn swallow	–	1	1	6	–	–	3	6	4	4	4	8	8	4	3	–	4	2	8	–	4	2	4	2	6	8
common raven	5	3	5	3	4	5	7	2	2	4	8	3	2	4	1	4	4	4	4	2	2	4	5	3	3	4
northwestern crow	15	50	40	15	35	–	–	–	–	–	–	–	–	18	–	–	2	105	25	–	–	4	–	4	50	15

[A] Beside Muir Inlet Beach, 0.8 km south of Muir Point and the mouth of Adams Inlet. Area investigated within 6km of camp.

[B] Beside Nunatak Cove, area investigated within 8 km of camp.

[C] Area investigated within 3 km of camp in Bartlett Cove.

Appendix F (continued). Breeding landbird species and number of individuals observed in GLBA at Muir Cabin, Nunatak Station, Bartlett Cove, and an unspecified location during Trautman (1966).

Common Name	July																									
	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
	Muir Cabin[A]							Nunatak Station[B]							Unspecified Location									Bartlett Cove[C]		
chestnut-backed chickadee	–	2	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	5
winter wren	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
robin	–	–	–	1	1	2	–	–	–	–	–	–	30	–	–	–	–	–	–	–	–	–	2	4	12	15
varied thrush	20	5	–	–	2	11	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	1	2	5
hermit thrush	10	4	5	6	4	8	–	3	–	–	–	2	2	1	8	1	–	–	1	–	5	5	10	2	10	12
Swainson's thrush	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1	–
gray-cheeked thrush	2	1	4	–	1	1	–	1	–	–	1	1	1	–	–	–	–	–	–	–	1	–	–	–	–	–
golden-crowned kinglet	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	1	–	–	–	–	–	–	1	–
ruby-crowned kinglet	4	4	–	1	1	4	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	10	8
water pipit (American pipit)	–	–	–	–	–	–	24	–	2	8	10	40	10	18	20	4	2	6	1	–	–	1	–	–	–	–
orange-crowned warbler	10	6	8	7	10	10	–	–	4	–	2	4	10	2	18	2	–	–	3	–	7	–	4	–	3	–
yellow warbler	12	12	15	3	6	5	–	–	3	–	–	1	–	–	2	1	–	–	1	–	–	–	1	–	1	–
Yellow-rumped warbler	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–
Townsend's warbler	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–
common yellowthroat	–	–	–	–	–	–	–	–	–	–	–	1	1	–	1	–	1	–	–	–	–	–	–	–	–	–

[A] Beside Muir Inlet Beach, 0.8 km south of Muir Point and the mouth of Adams Inlet. Area investigated within 6km of camp.

[B] Beside Nunatak Cove, area investigated within 8 km of camp.

[C] Area investigated within 3 km of camp in Bartlett Cove.

Appendix F (continued). Breeding landbird species and number of individuals observed in GLBA at Muir Cabin, Nunatak Station, Bartlett Cove, and an unspecified location during Trautman (1966).

	July																									
	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
Common Name	Muir Cabin[A]							Nunatak Station[B]							Unspecified Location									Bartlett Cove[C]		
Wilson's warbler	20	30	–	2	2	8	–	–	4	2	18	5	12	–	10	3	1	–	4	2	3	–	1	–	30	5
pine grosbeak	–	1	–	2	2	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–
gray-crowned rosy finch	–	–	20	–	–	–	4	5	2	8	10	8	3	2	–	–	–	–	–	–	50	–	–	–	–	–
common redpoll	20	–	–	–	–	5	–	3	15	10	10	10	30	24	50	10	10	20	10	30	100	20	110	20	3	10
pine siskin	12	2	13	–	8	10	–	–	3	–	–	–	3	–	–	3	–	–	–	5	5	–	2	–	–	–
savannah sparrow	9	12	15	12	18	15	–	1	10	3	20	8	12	10	15	5	3	1	17	3	6	–	3	–	40	30
dark-eyed junco	4	4	3	1	3	4	–	–	–	–	–	–	4	–	–	–	–	–	–	–	10	–	–	–	9	40
golden-crowned sparrow	–	–	2	–	–	–	–	–	1	–	–	–	40	–	–	–	–	–	–	–	–	–	–	–	–	–
fox sparrow	10	12	25	18	15	10	4	–	8	2	12	2	30	30	40	12	8	15	35	15	15	5	42	10	15	15
Lincoln's sparrow	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
song sparrow	–	–	–	–	–	–	3	–	–	–	–	–	–	2	–	–	–	–	–	–	–	–	–	–	3	–
snow bunting	–	–	–	–	–	–	–	–	–	–	–	3	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Number of Individuals	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	**5439**
Number of Species	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	**43**

[A] Beside Muir Inlet Beach, 0.8 km south of Muir Point and the mouth of Adams Inlet. Area investigated within 6km of camp.

[B] Beside Nunatak Cove, area investigated within 8 km of camp.

[C] Area investigated within 3 km of camp in Bartlett Cove.

Appendix G. Breeding landbirds of conservation concern observed in GLBA between 1966 and 2004.

Common Name	USFWS (2008); BCR 5	Rich et al. (2004)	Berlanga et al. (2010)	Federal Listed	State Listings	BLM	Audubon Alaska Watchlist
northern goshawk	X	–	–	–	–	–	Yellow List
bald eagle	X	LPR*	–	–	–	–	–
rufous hummingbird	X	M*	STEEP DECLINE †	–	–	–	–
belted kingfisher	–	–	STEEP DECLINE	–	–	–	–
peregrine falcon	X	–	–	DL	SSOC	Sensitive	
sooty grouse	–	–	STEEP DECLINE †	–	–	–	–
Steller's jay	–	LPR*	–	–	–	–	–
fox sparrow	–	LPR*	–	–	–	–	–
snow bunting	–	–	STEEP DECLINE	–	–	–	–
golden-crowned sparrow	–	LPR*	–	–	–	–	–
pine siskin	–	–	STEEP DECLINE	–	–	–	–
bank swallow	–	–	STEEP DECLINE	–	–	–	–
chestnut-backed chickadee	–	LPR*	–	–	–	–	–

M = continued active management is the recommended conservation action (Rich et al. 2004)

LPR = long-term planning and responsibility is the recommended conservation action (Rich et al. 2004)

*= Stewardship species (SS) with >75% of population found in BCRs 5, 15, and 32

Tri-National = Temperate breeders of high tri-national concern (Berlanga et al. 2010)

Steep Decline = percent (%) population loss based on BBS or CBC trend since mid-1960s, or on PT score (>50%) if no reliable trend data

† = species endemic to the Tri-National area (Berlanga et al. 2010)

BCC = Bird of Conservation Concern, Federal listing category

DL = Delisted, but being monitored, Federal listing category

SSOC = State Species of Concern, State listing category

Appendix G (continued). Breeding landbirds of conservation concern observed in GLBA between 1966 and 2004.

Common Name	USFWS (2008); BCR 5	Rich et al. (2004)	Berlanga et al. (2010)	Federal Listed	State Listings	BLM	Audubon Alaska Watchlist
Townsend's warbler	–	–	–	–	SSOC	Sensitive	–
Wilson's warbler	–	–	STEEP DECLINE	–	–	–	–
winter wren (pacific wren)	–	LPR*	–	–	–	–	–
gray-cheeked thrush	–	–	–	–	SSOC	Sensitive	–
varied thrush	–	–	–	–	–	–	Red List
olive-sided flycatcher	X	M	Tri-National	–	SSOC	Sensitive	Red List
Pacific-slope flycatcher (western flycatcher)	–	LPR*	–	–	–	–	–
northern flicker	–	–	STEEP DECLINE	–	–	–	–
short-eared owl	–	–	STEEP DECLINE	BCC	–	–	–
TOTAL:	**5**	**9**	**10**	**2**	**4**	**4**	**3**
M/SS:	–	**2/1**	–	–	–	–	–
LPR/SS:	–	**7/7**	–	–	–	–	–
STEEP DECLINE/ ENDEMIC:	–	–	**9/2**	–	–	–	–
TRI-NATIONAL:	–	–	**1**	–	–	–	–

M = continued active management is the recommended conservation action (Rich et al. 2004)

LPR = long-term planning and responsibility is the recommended conservation action (Rich et al. 2004)

*= Stewardship species (SS) with >75% of population found in BCRs 5, 15, and 32

Tri-National = Temperate breeders of high tri-national concern (Berlanga et al. 2010)

Steep Decline = percent (%) population loss based on BBS or CBC trend since mid-1960s, or on PT score (>50%) if no reliable trend data

† = species endemic to the Tri-National area (Berlanga et al. 2010)

BCC = Bird of Conservation Concern, Federal listing category

DL = Delisted, but being monitored, Federal listing category

SSOC = State Species of Concern, State listing category

Appendix H. Sea otter prey items during observed foraging bouts between 1993 and 2004 (Bodkin et al. 2006), and between 1993 and 2011 (Weitzman 2013).

Prey Items	Bodkin et al. (2006)	Weitzman (2013)
Bivalves - Clams		
Clinocardium nuttallii	X	X
Entodesma navicular	X	X
Gari californica	X	
Leukoma staminea		X
Mactromeris polynyma	X	X
Macoma spp	X	X
Mya arenaria	X	
Mya truncata	X	
Mya spp.		X
Protothaca staminea	X	
Saxidomus gigantea	X	X
Serripes groenlandicus	X	X
Unidentified clam		X
Bivalves – Mussels		
Modiolus modiolus	X	X
Mytilus trossulus	X	X
Unidentified mussel		X
Bivalves – Others		
Chlamys rubidis		X
Pododesmus macrochisma	X	
Unidentified scallop	X	X
Unidentified bivalve		X
Gastropods		
Euspira lewisii		X
Fusitriton oregonensis	X	X
Neptunea spp.	X	X
Limpet	X	
Unidentified snail		X
Mollusks – Others		
Cryptochiton stelleri	X	X
Octopus dofleini	X	X
Unidentified chiton		X
Echinoderms – Stars		
Crossaster papposuss		X
Gorgonocephalus caryi	X	X

Appendix H (continued). Sea otter prey items during observed foraging bouts between 1993 and 2004 (Bodkin et al. 2006), and between 1993 and 2011 (Weitzman 2013).

Prey Items	**Bodkin et al. (2006)**	**Weitzman (2013)**
Ophiuroid sp.	X	X
Pteraster tesselatus		X
Pycnopodia helianthoides	X	X
Solaster spp.	X	X
Unidentified sea star		X
Echinoderms –Urchins		
Strongylocentrotus droebachiensis	X	X
Echinoderms – Other		
Cucumaria fallax	X	X
Cucumaria miniata		X
Cucumaria spp.		X
Crustaceans		
Cancer magister	X	X
Cancer productus		X
Chionoecetes bairdi	X	X
Paralithodes camtschatica	X	X
Paguridae spp.	X	
Pugettia gracilis		X
Pugettia spp.	X	X
Telmessus cheiragomus	X	X
Pandalus spp.	X	X
Barnacle spp.	X	X
Unidentified crab		X
Other		
Chordate: fish	X	X
Porifera: Sponge	X	
Worm: *Echiurus* spp.	X	X
Unidentified worm		X

Appendix I. Reported sea otter strandings in GLBA, Icy Strait, and Gustavus area between 2006 and 2013 (NOAA 2014). The K/U column states whether cause of death was known (K) or unknown (U).

Date	Place	Latitude	Longitude	Necropsy	K/U	Cause	Comments
30-May-06	Beardslee Islands	58.57194	-135.96173	No	U	–	Unable to collect - found on Beardslee Island #6
7-Jun-06	Pt Carolus	58.38365	-136.03536	Yes	K	Pulmonary edema	Cardiovascular collapse - Infection/septic
10-Jun-06	Bartlett Cove	58.4618	-135.88737	Yes	U	–	Not trauma and not Valvular Endocarditis
11-Aug-06	Gustavus	58.39183	-135.72354	Yes	K	Valvular endocarditis - Tested positive for Strep bovis	Infection/pulmonary conjest.edema/acathocephoild infestation (mild) and gastric mucosal ulceration
29-Aug-06	Strawberry Island	58.51521	-136.047113	Yes	K	Trauma	Intraspecific - Lumbar region w/massive hemorrhaging
29-Jun-07	Strawberry Island	58.4956	-136.04745	Yes	K	Trauma	Extensive bruising
11-Jun-08	Boulder Island	58.54896	-136.03009	Yes	K	Pulmonary edema	Intraspecific trauma/drown while mating
22-Jun-08	Lester Point			No	U		Rotten with maggots
23-Aug-08	Bartlett Cove	58.45244	-135.89207	Yes	K	Head trauma	Most likely Basisphenoid bone fracture
23-May-09	GST Beach	–	–	Yes	U	–	Possibly previously seen sick animal. Partially burned
17-Jun-09	Pleasant Island	–	–	No	U	–	2 well decomposed animals
10-Aug-09	Bartlett Cove	58.4575	-136.0036	Yes	K	Intestinal torsion	Possibly caused by cardiomyopathy
15-Oct-09	Bartlett Cove	58.4526	-135.88483	Yes	U	–	Possible cardiomyopathy
11-Nov-09	GST Beach	–	–	Yes	K	Predation	Orca predations w/ post mortem boat strike?
28-Jan-10	Gustavus	–	–	No	U	–	Possibly hunted
18-Jun-10	Airport Ditch/GST	58.3955	-135.67007	Yes	K	Pulmonary congestion/edema	Pulmonary congestion/edema and widespread vascular and liver congestion.
19-Jun-10	Airport Ditch/GST	–	–	Yes	U	–	Severely scavenged - Advanced decomposition
16-Jul-10	South Sandy Cove	–	–	Yes	U	–	Advanced decomposition
21-Jul-10	Leland Island	58.63448	-135.98853	Yes	K	Septicemia	Secondary COD Emaciation
22-Jul-10	Sitakaday Narrows	58.43108	-135.9039	Yes	U	–	Advanced decomposition.
25-Jul-10	Sitakaday Narrows	58.5056	-136.0633	No	U	–	–
25-Jul-10	Taylor Bay	58.28981	-136.48148	No	U	–	–

Appendix I (continued). Reported sea otter strandings in GLBA, Icy Strait, and Gustavus area between 2006 and 2013 (NOAA 2014). The K/U column states whether cause of death was known (K) or unknown (U).

Date	Place	Latitude	Longitude	Necropsy	K/U	Cause	Comments
27-Jul-10	Icy Strait	58.31253	-135.85986	Yes	U	–	–
27-Jul-10	Point Adolphus	58.29137	-135.76178	Yes	U	–	Advanced decomposition
7-Aug-10	Beardslee Entrance	–	–	No	U	–	Scavenged and decayed. Beach cast
17-Aug-10	Icy Strait	58.33014	-135.86934	Yes	U	–	Advanced decomposition
18-Aug-10	Beardslee Islands	–	–	Yes	K	Starvation	–
21-Sep-10	Excursion Inlet	58.4742	-135.483716	Yes	K	Gunshot	LE case investigation
6-Jul-11	Beardslee Islands	58.53835	-135.860556	Yes	K	Trauma - Gunshot	Small puncture wounds near the head. Bullet fragments found
18-Jul-11	Dude Creek	–	–	No	U	Possible Gunshot	Decayed - Blood near head w/algae - Gunshot?
13-Aug-11	Gustavus dock	–	–	No	U	–	Injured/Sick just E of the GST dock. Lying on the beach barely looked alive and shaking.
6-Jan-12	Gustavus Beach	–	–	No	U	–	Beach cast about 10 minutes east of the GST dock. Looked fresh eyes missing but no apparent cause of death. Not collected
9-Apr-12	Gustavus Beach	58.394083	-135.716416	Yes	K	Acute Heart Failure	Rotten with no eyes, fur coming out and oozing brown goo. Best guess due to amount of fluid in abdomen and good fat stores died from acute heart failure.
11-Jun-12	Scidmore Cut	58.84605	-136.61636	Yes	K	Possible Stillbirth	Young animal with no obvious injuries or trauma. Eyes were not fully open, may not have been a full term (premature) Possible stillbirth, no milk in system.
25-Jun-12	Boulder Island	58.558254	-136.01948	No	U	–	Partially decomposed
20-Aug-12	Boulder Island	58.56139	-136.05298		U	–	Very bloated
24-Aug-13		–	–	No	U	–	Dead floating
12-Oct-13	Rink Creek	–	–	No	U	–	Appeared to have been dragged up the beach by wolves, eyes missing and some missing fur.
23-May-13	Boulder Island	58.561717	-136.017437	–	U	–	Seven individuals involved. Skeletal remains with hides.

Appendix I (continued). Reported sea otter strandings in GLBA, Icy Strait, and Gustavus area between 2006 and 2013 (NOAA 2014). The K/U column states whether cause of death was known (K) or unknown (U).

Date	Place	Latitude	Longitude	Necropsy	K/U	Cause	Comments
5-Jul-13	Sitakaday Narrows	–	–	No	U	–	Floating dead not collected
6-Jul-13	Cooper's Notch	–	–	No	U	–	Floating dead not collected
17-Jul-13	Composite Island	58.89702	-136.565895	No	U	–	Still intact but appeared mummified according to reporting party.

Appendix J. Invasive species documented within GLBA by the Alaska EPMT (NPS 2013a). Effort are areas sampled in order to document invasives varied from year to year which may have contributed to variation observed annually.

Scientific name	2004	2005	2006	2007	2008	2009	2010	2011	2012
*Achillea ptarmica**						X			
*Aegopodium podagraria**			X						
Alchemilla mollis						X		X	
Allium schoenoprasum		X	X		X	X			
Alopecurus pratensis				X	X	X	X		
Arctium minus			X	X	X				
Brassica rapa							X		
Bromus inermis	X	X			X	X	X		
Campanula medium					X	X	X		
Capsella bursa-pastoris		X	X				X	X	X
Centaurea montana				X	X	X		X	X
Cerastium fontanum	X	X	X	X	X	X	X	X	X
*Cerastium tomentosum**						X			
Chenopodium album				X					
Cirsium arvense^*			X	X		X			
Dactylis glomerata		X	X	X	X	X	X	X	X
*Elymus repens^*		X		X	X	X	X	X	X
Galeopsis tetrahit^*				X					
Geranium robertianum						X	X	X	X
Hieracium aurantiacum^*				X	X			X	
Hordeum jubatum		X	X		X	X			
Hypochaeris radicata		X	X	X	X				
Lamium album		X	X		X	X	X		
Leucanthemum vulgare	X	X	X	X	X	X	X	X	X
*Leucanthemum x superbum**						X			
*Linaria vulgaris**				X		X			
Lolium perenne	X	X	X		X		X	X	
*Lonicera tatarica**								X	
Lupinus polyphyllus	X	X	X	X	X	X	X	X	X
Matricaria discoidea	X	X	X	X	X	X	X	X	X
Meconopsis betonicifolia									X
Mentha sp.			X	X	X				
Myosotis asiatica					X				

* Species documented only in the town of Gustavus but not within official GLBA boundaries.

^ Species classified as prohibited noxious weeds in the state of Alaska (AK DNR 2010).

Appendix J (continued). Invasive species documented within GLBA by the Alaska EPMT (NPS 2013a). Effort are areas sampled in order to document invasives varied from year to year which may have contributed to variation observed annually.

Scientific name	2004	2005	2006	2007	2008	2009	2010	2011	2012
Myosotis scorpioides		X	X		X	X	X	X	
Myosotis sylvatica									X
Phalaris arundinacea	X	X	X	X	X	X		X	X
Phleum pratense	X	X	X	X	X	X	X	X	X
Plantago major	X	X	X	X	X	X	X	X	
Poa annua			X	X	X	X			
Poa pratensis			X		X				
*Polygonum aviculare**				X					
Ranunculus acris		X	X	X	X	X	X	X	X
Ranunculus repens		X	X	X	X	X	X	X	X
Rheum rhabarbarum		X	X	X	X				
Rosa rugosa			X	X		X		X	
Rubus idaeus		X	X	X	X		X	X	X
Rumex acetosella		X	X	X		X	X		
Rumex crispus						X	X		
*Silene chalcedonica**	X								
*Sonchus arvensis^*		X	X	X	X		X	X	X
Sorbus aucuparia		X	X	X	X			X	
Stellaria media		X	X		X	X	X		
Symphytum officinale		X	X	X	X	X	X	X	X
Taraxacum officinale ssp. officinale	X	X	X	X	X	X	X	X	X
Trifolium hybridum		X		X				X	
Trifolium pratense	X	X	X	X	X	X	X		
Trifolium repens	X	X	X	X	X	X	X	X	
*Tripleurospermum inodorum**									X
Triticum aestivum	X	X	X	X	X			X	
Veronica serpyllifolia				X		X	X		
*Veronica spicata**						X			
Viola tricolor				X	X	X	X	X	X
Annual totals	**14**	**31**	**35**	**37**	**38**	**38**	**30**	**29**	**21**
Annual hectares inventoried	**333.3**	**346.2**	**710.8**	**691.5**	**480.4**	**123.9**	**72.9**	**31.6**	**13.0**
Cumulative # of species documented	**14**	**31**	**38**	**47**	**49**	**56**	**57**	**59**	**62**

* Species documented only in the town of Gustavus but not within official GLBA boundaries.

^ Species classified as prohibited noxious weeds in the state of Alaska (AK DNR 2010).

Appendix K. Western toad distribution by habitat type and by life stage as reported in the GLBA area from 1991-2010 (NPS 2015). Observations are the result of volunteers and were recorded opportunistically. Survey effort and location varied annually.

Life Stage	**1991**	**1993**	**1997**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**Total**
Adults															
Forested Area	–	–	–	–	–	–	1	–	–	–	1	–	2	–	**4**
Freshwater Pond/Lake	–	–	–	–	1	–	2	–	–	–	2	3	–	–	**8**
Other	–	–	–	–	2	–	4	–	–	–	–	–	–	–	**6**
River	–	–	–	–	2	–	–	–	–	–	–	–	–	–	**2**
Saltwater/Estuarine	–	–	–	–	–	1	–	7	–	–	–	–	–	–	**8**
Stream	–	–	–	–	2	1	2	–	–	–	3	–	–	–	**8**
Wetland/Bog	–	–	–	–	1	–	–	1	–	–	2	–	–	–	**4**
Not Classified	–	1	3	–	–	–	1	7	2	12	13	3	3	4	**49**
Yearly Total	**0**	**1**	**3**	**0**	**8**	**2**	**10**	**15**	**2**	**12**	**21**	**6**	**5**	**4**	**89**
Subadults															
Forested Area	–	–	–	–	–	–	4	–	–	–	–	–	–	–	**4**
Freshwater Pond/Lake	–	–	–	–	3	1	–	24	–	–	–	–	–	–	**28**
Other	–	–	–	–	3	–	–	–	–	–	–	–	–	–	**3**
Saltwater/Estuarine	–	–	–	–	–	1	–	–	–	–	–	–	–	–	**1**
Stream	–	–	–	–	4	–	–	–	–	–	–	–	–	–	**4**
Wetland/Bog	–	–	–	–	1	–	–	–	–	–	–	–	–	–	**1**
Not Classified	–	–	–	–	–	–	1	–	–	5	10	1	–	–	**17**
Yearly Total	**0**	**0**	**0**	**0**	**11**	**2**	**5**	**24**	**0**	**5**	**10**	**1**	**0**	**0**	**58**
Tadpoles															
Forested Area	–	–	–	–	–	–	40	–	–	–	–	–	–	–	40
Freshwater Pond/Lake	–	–	–	–	–	125	30	3100	1050+	–	–	–	2	–	3257
Other	–	–	–	–	–	1108	–	–	–	–	–	–	–	–	**1108**
River	–	–	–	–	–	–	–	300	–	–	–	–	–	–	**300**
Stream	–	–	–	–	–	–	–	–	–	–	–	–	100	–	**100**

Appendix K (continued). Western toad distribution by habitat type and by life stage as reported in the GLBA area from 1991-2010 (NPS 2015). Observations are the result of volunteers and were recorded opportunistically. Survey effort and location varied annually.

Life Stage	1991	1993	1997	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	Total
Not Classified	3	–	–	100	–	–	200	11500	–	–	300	–	1	–	**12104**
Yearly Total	**3**	**0**	**0**	**100**	**0**	**1,233**	**270**	**14,900**	**1,050**	**0**	**300**	**0**	**103**	**0**	**17,959**
Undefined/Other															
Forested Area	–	–	–	–	–	–	–	–	–	–	–	1	–	–	**1**
Stream	–	–	–	–	1	–	–	–	–	–	–	–	–	–	**1**
Not Classified	–	–	–	–	–	–	–	–	–	14	6	1	–	–	**21**
Yearly Total	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	**0**	**14**	**6**	**2**	**0**	**0**	**23**
Eggs															
Freshwater Pond/Lake	–	–	–	–	–	–	–	–	200+	–	–	–	–	–	**200+**
Yearly Total	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**200***	**0**	**0**	**0**	**0**	**0**	**200+**
Total	**3**	**1**	**3**	**100**	**20**	**1,237**	**285**	**14,939**	**1252**	**31**	**337**	**9**	**108**	**4**	**18,529+**

Appendix L. Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

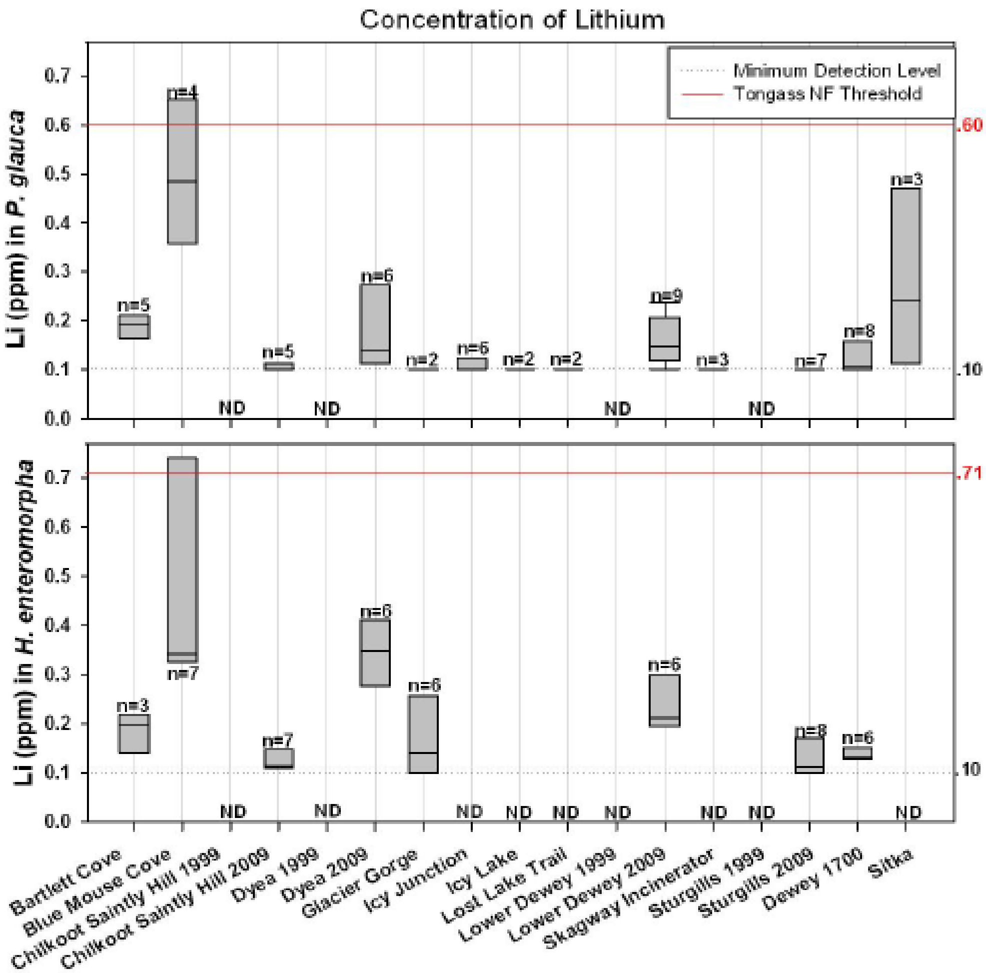

Figure L-1. Lithium concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

Appendix L (continued). Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

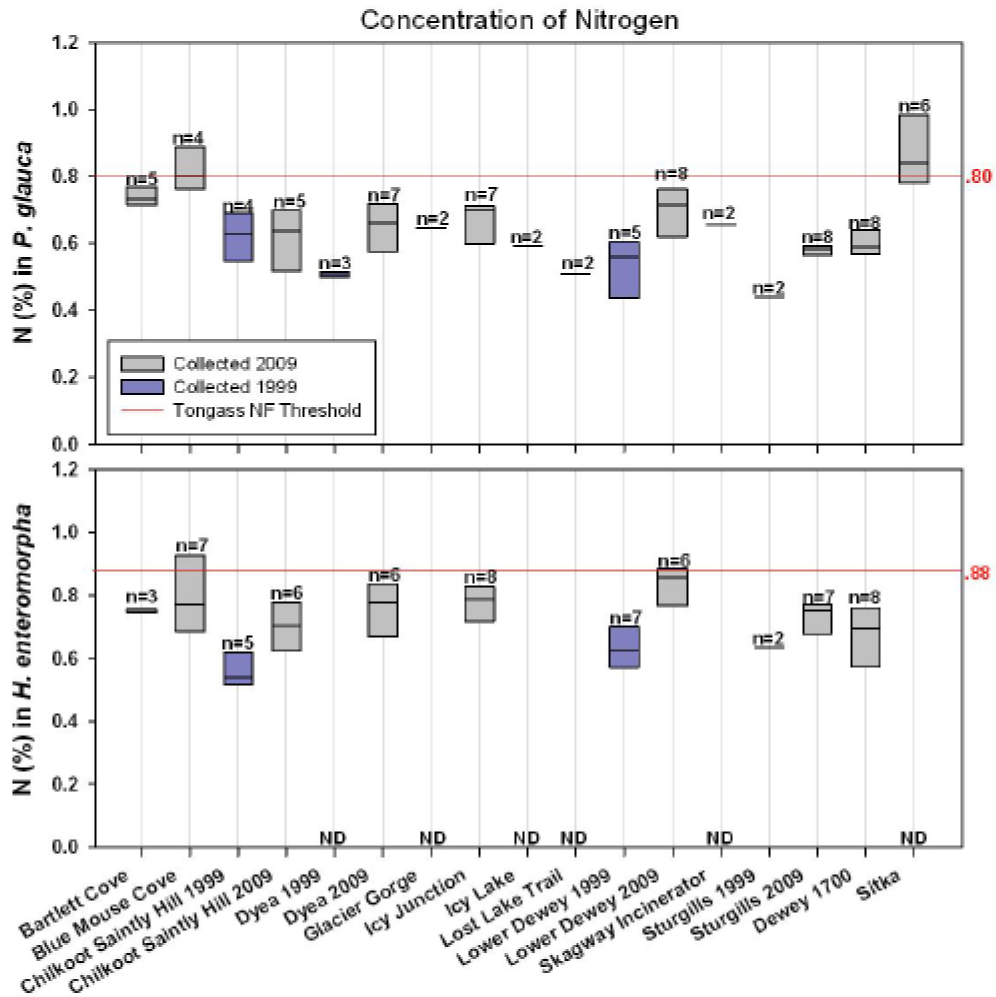

Figure L-2. Nitrogen concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

Appendix L (continued). Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

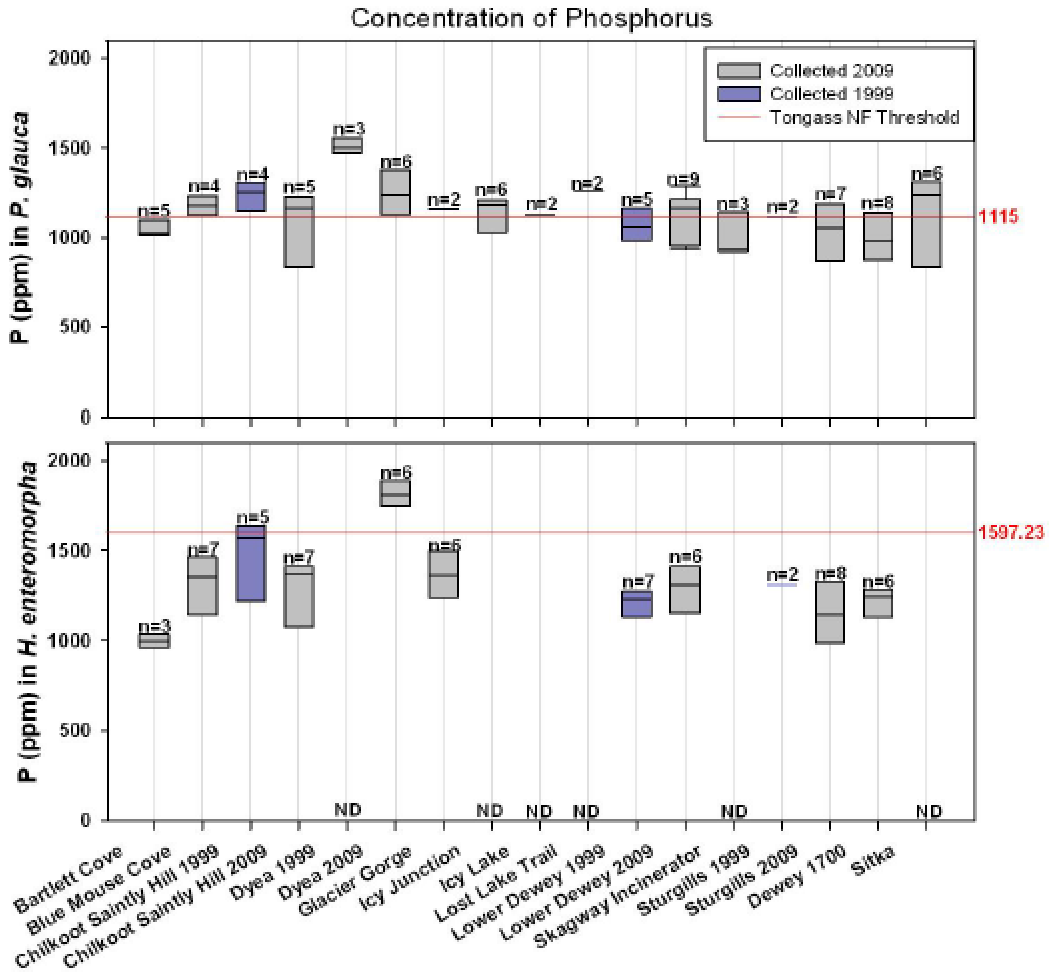

Figure L-3. Phosphorus concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

Appendix L (continued). Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

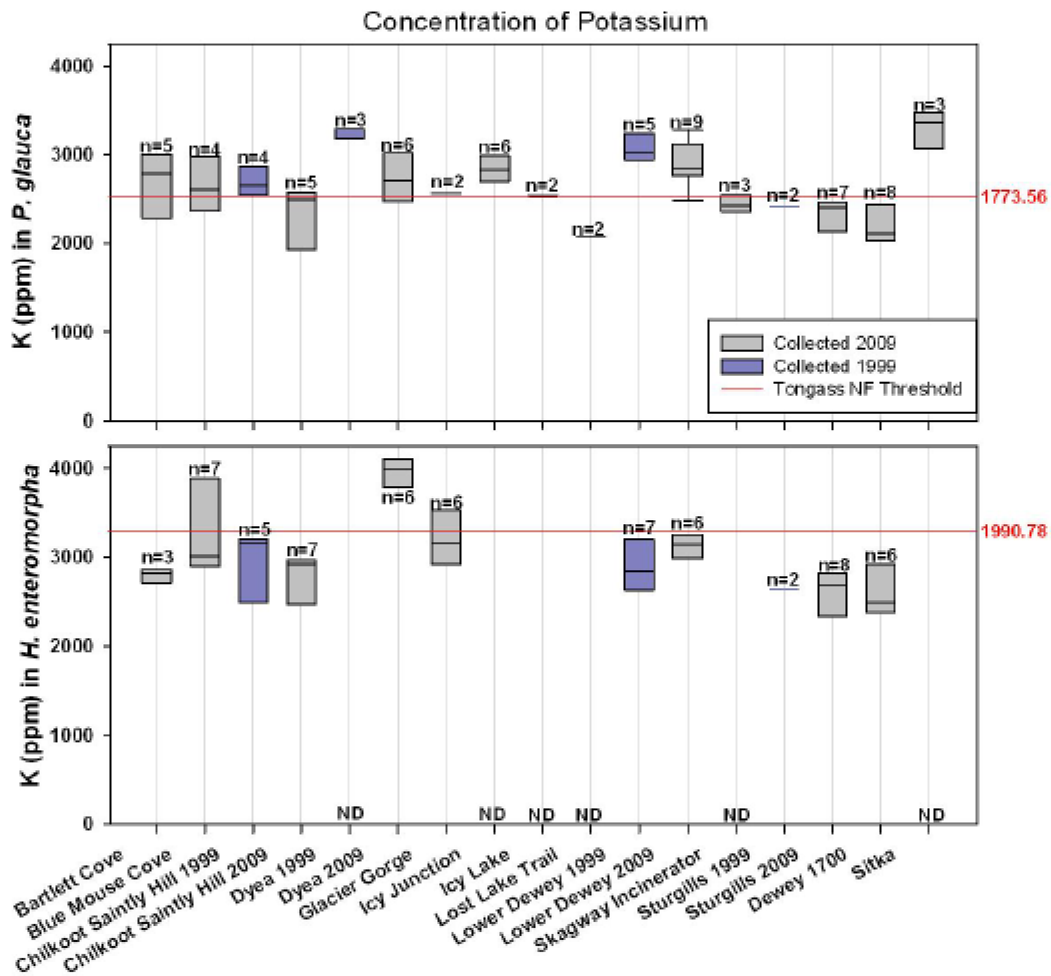

Figure L-4. Potassium concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

Appendix L (continued). Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

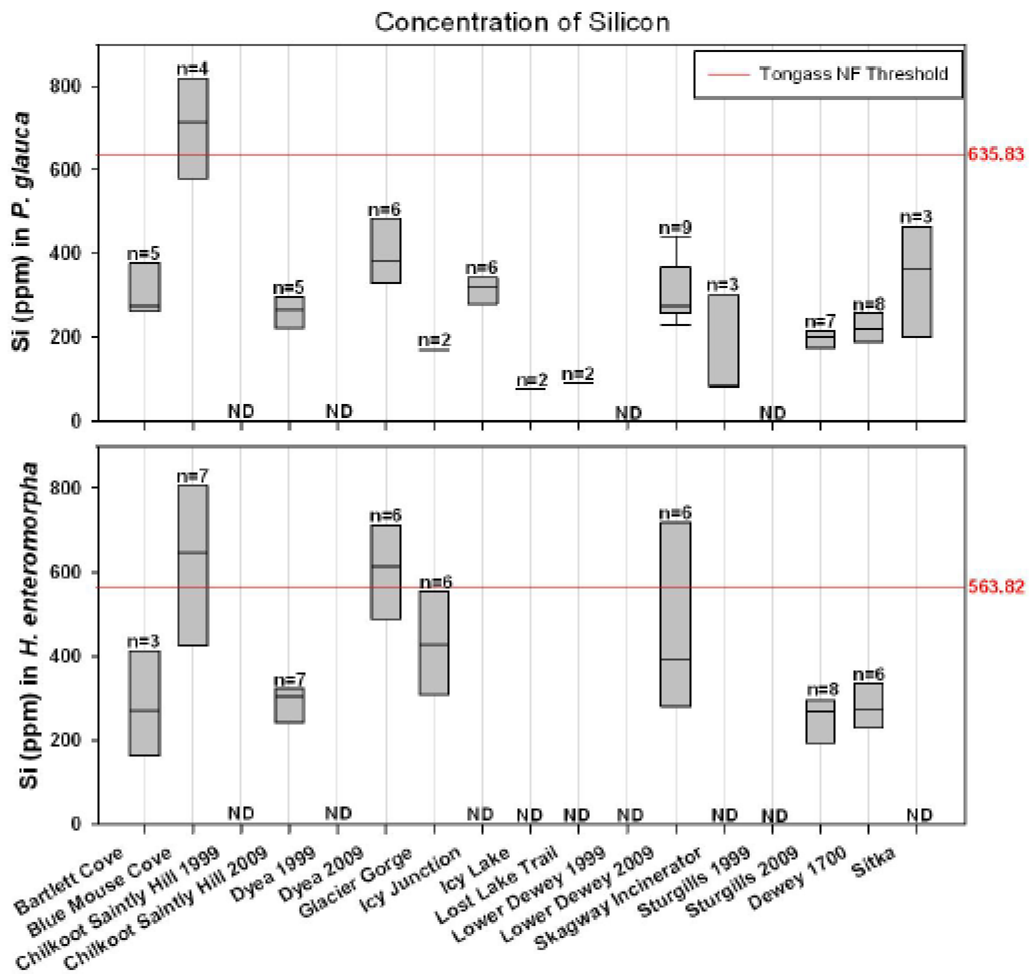

Figure L-5. Silicon concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

Appendix L (continued). Elemental concentrations in lichen tissue (Schirokauer et al. 2014).

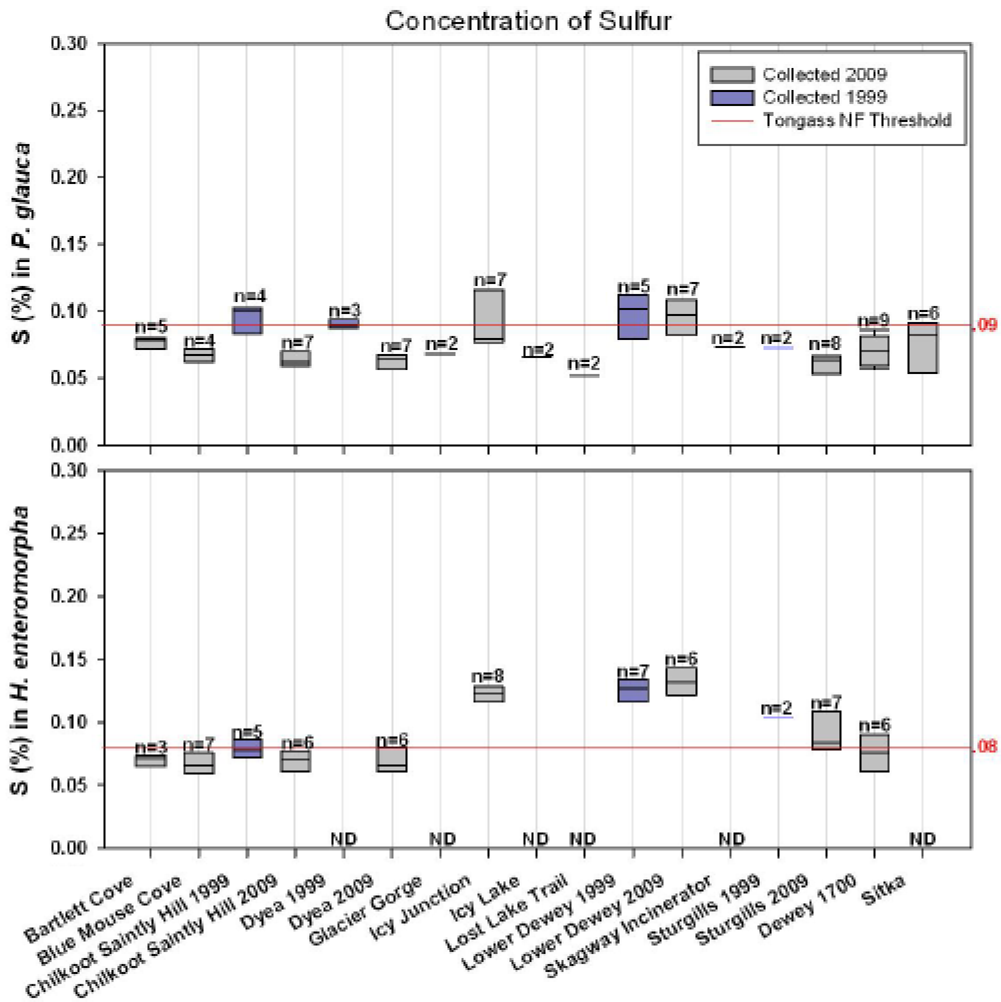

Figure L-6. Sulfur concentrations in two species of lichen sampled in southeastern Alaska by Schirokauer et al. (2014). The two sites in GLBA (Bartlett Cove and Blue Mouse Cove) are on the far left.

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