GWAII HAANAS

NATIONAL PARK RESERVE AND HAIDA HERITAGE SITE

TECHNICAL COMPENDIUM

to the

2007 State of the Protected Area Report

Protected through the cooperation of the
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Gwaii Haanas

National Park Reserve and Haida Heritage Site

Technical Compendium to the 2007 State of the Park Report

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The Haida family crest of Gwaii Haanas acquired through potlatch.

Images of the sea otter (*Enhydra lutris* / in Haida **kuu**) and red sea urchin (*Strongylocentrotus franciscanus* / in Haida **guuding aay**) by the artist Giitsxaa. These animals were chosen for Gwaii Haanas by Haida Elders as they resonate in local history, reminding us of the vulnerability of species and ecosystems. Red sea urchins abound around Haida Gwaii because their predator, the sea otter, was extirpated during the fur trade era.

1. INTRODUCTION

This report is the Technical Compendium (TC) to the first ever State of the Protected Area Report (SoPR) for Gwaii Haanas National Park Reserve and Haida Heritage Site (Gwaii Haanas) completed September, 2007. The SoPR recounts levels of well-being of Ecological Integrity (EI) of indicator ecosystems using individual measures, as well as separate assessments on Visitor Experience, Public Education and Cultural Resources. Adding the latter three components to the EI component within the total SoPR is a new Parks Canada initiative for which some terms of reference are still under development.

This TC reports only on EI and provides a historical and statistical background relating to the measures for the indicator ecosystems featured in Gwaii Haanas' first SoPR. Also included are measures not reported in the SoPR as they are under development and may appear in Gwaii Haanas' next SoPR in 2012. The biodiversity, process and stressor measures reported on in the SoPR are listed according to each indicator ecosystem in Table 1. Outcomes of the status and trend analyses for these measures are summarized in Table 2. This TC complies fully with the guiding principles for monitoring and reporting EI, as outlined in PCA (2005).

Being the first TC ever compiled for Gwaii Haanas, background historical context is provided for each EI measure reported herein. Such context aids initiating what will be an on-going series within the required five-year reporting cycle for Gwaii Haanas hereafter. It is also essential to underscore that Gwaii Haanas' EI monitoring program remains a work-in-progress towards the Parks Canada-wide deadline of complete monitoring programs at Field Units by 2008.

For each EI indicator, we provide details (when available) on status and trend assessments, data quality, monitoring methods and thresholds. The methods used here are consistent with the data analyses section of the EI monitoring guide outlined in PCA (2006). In keeping with Agency guidelines and colour convention, a change in status of an EI indicator or measure represents a shift from a good (green), fair (yellow), or poor (red) state to some other state. Changes in status will comply with the 5-year reporting cycle for SoPRs (and attendant TCs). A change in trend refers to the following two distinct, yet related, events: (1) a change in trend refers to the magnitude and direction of change in

Table 1. Summary of Ecological Integrity measures for Gwaii Haanas according to indicator ecosystem reported in the 2007 State of the Park Report.

Indicator Ecosystem	Biodiversity	Process	Stressor
Forest	• Vascular plants • Marbled Murrelet	• Forest insects and diseases • *Forest structure*[1]	• Non-native plants • Introduced deer • Human footprint
Non-forested	• *Vascular plants*	• Extent of the alpine	• *Non-native plants* • Introduced deer
Lake and Wetland	• *Vascular plants* • Western toad	• *Extent of wetlands/lakes*	• *Non-native plants* • *Introduced deer* • Non-native amphibians
Stream, River and Estuary	• Spawning salmon • *Benthic invertebrates*	• Water quality • *Riparian land cover*	• Not measured
Shoreline	• Black Oystercatcher • Colony-nesting seabirds • Peale's Peregrine Falcon • Steller sea lion • *Bat maternity colony*	• Coastal erosion	• Raccoons on seabird islands • Invasive plants • Visitor effects at campsites
Marine (Intertidal and Subtidal)	• CHAP[2] (Fish Assemblage Assessment) • Spawning Pacific herring	• CHAP (Environmental Assessment) • CHAP (Eelgrass Health Assessment)	• CHAP (Anthropogenic Disturbance Index)
Park-wide	• *Species at risk*	• Not measured	• Non-native mammals

1 measures not assessed in the SoPR are in *italics*, although detailed information is available in this Technical Compendium
2 CHAP (Coastal Health Assessment Program), that uses eelgrass meadows as the biological sentinel, has four measures

Table 2. Roll-up for Gwaii Haanas' Indicator Ecosystems reported in the 2007 State of the Park Report: (A) Status and (B) Trend.

A) Status

Indicator Ecosystem (number of measures)	Number of Measures According to Status				Rolled-up Status
	Good	Fair	Poor	Unknown	
Forest (7)	2	3	2	-	fair
Non-forested (2)	-	-	1	1	unknown
Lake and Wetland (2)	1	-	-	1	good
Stream, River and Estuary (2)	1	1	-	-	good
Shoreline (8)	5	2	1	-	good
Marine (2)	1	-	1	-	fair
Park-wide (1)	-	-	1	-	unknown

B) Trend

Indicator Ecosystem (number of measures)	Number of Measures According to Trend				Rolled-up Trend
	Improving	Stable	Deteriorating	Unknown	
Forest (7)	1	1	1	4	deteriorating
Non-forested (2)	-	-	-	2	unknown
Lake and Wetland (2)	-	-	1	1	unknown
Stream, River and Estuary (2)	-	1	-	1	unknown
Shoreline (8)	2	5	1	-	unknown
Marine (2)	-	2	-	-	stable
Park-wide (1)	-	-	1	-	unknown

an individual monitoring measure, and (2) a change in trend refers to a change in status, e.g., an increase (from yellow to green) or a decrease (from yellow to red). Interpretation of these two types of trend may be in the same or opposite directions. For example, a decrease in a stressor could increase status, whereas a decrease in a biodiversity measure could decrease status.

Thresholds, as described in PCA (2005), refer to explicit levels in a monitoring measure (or indicator) where a change in status is reached. Every measure has the following two thresholds: (1) between green and yellow, and (2) between yellow and red. The Agency has standards on identification methods for these thresholds (PCA 2006). Of course, each park is unique for threshold establishment according to its monitoring history and ecological characteristics. Thresholds will be subject to regular reviews and updating.

We assess data quality, through power analysis, where possible. For a specific statistical test, power analysis quantifies the relationship between the minimum detectable change in a measure, variation, type I error rate or confidence level, type II error rate or power, and sample size. Key is the monitoring program's ability to detect a certain magnitude of change with an acceptable error level given the measure's sample size and variation.

The unique opportunity Gwaii Haanas presents Parks Canada and the nation in conservation management warrants mention. Firstly, key context to understanding Gwaii Haanas is that it was among the first of the national parks to be cooperatively managed between a First Nation and Parks Canada Agency in southern Canada. Since 1993, Gwaii Haanas' lands and fresh (non-tidal) waters have been cooperatively managed by the Government of Canada (represented by Parks Canada Agency) and the Council of the Haida Nation (CHN) through the *Gwaii Haanas Agreement*. This agreement created the Archipelago Management Board (AMB) of two CHN and two Government of Canada representatives (from Gwaii Haanas) that make all management decisions. Within the spirit of cooperative management, both western science and traditional Haida knowledge sources are to be used and respected in aid of management decision-making (AMB 2003 a).

Secondly, there is the prospect of creating Gwaii Haanas National Marine Conservation Area Reserve (NMCAR) surrounding Gwaii Haanas. The conservation continuum envisioned will span alpine to deep-sea ecosystems. Indeed, the footprint of the proposed Gwaii Haanas NMCAR was legally defined in the *South Moresby Agreement* in 1988 along with a commitment to create a "national marine park." Gwaii Haanas' terrestrial management plan (AMB 2003 a) does not include marine waters, but, critically, states that; "… it cannot help but recognize the close relationship that exists between land and sea." Therefore, the inseparability of land and sea towards an integrated management approach over the long term characterizes Gwaii Haanas' management ethic. Certain influential marine issues must, therefore, be considered in monitoring now. Examples are colonies of nesting seabirds on land or the distribution of marine nutrients into riparian (near-stream) forests by spawning salmon.

2. INDICATOR ECOSYSTEMS AND THEIR MEASURES

In alignment with the Pacific Coast Bioregion, and our State of the Park Report (SoPR), we report here on measures within the following indicator ecosystems:

- **Forest –** coniferous rainforest from the sub-alpine to the shoreline;
- **Non-forested –** alpine and sub-alpine highland tundra;
- **Lake and Wetland –** lakes, ponds and freshwater wetlands at all elevations;
- **Stream, River and Estuary –** all water courses and marine-influenced wetlands;
- **Shoreline –** rocky to sedimentary shores at the land-sea interface;
- **Marine –** combines "intertidal" and "subtidal" Bioregional indicators, extending from the high tide line into the deep-sea; and
- **Park-wide –** measurements relevant across all ecosystems including species at risk and introduced species.

Each indicator ecosystem has between one to eight measures (total of 27 at this time) used to assess Gwaii Haanas' EI in our first SoPR. We also discuss 11 measures under development within our evolving monitoring program for the next SoPR. As the total monitoring program is not yet fully operational, we are operating on interim measures. Throughout the text, reference links will be made to relevant sections in the 2007 SoPR.

2.1. FOREST

2.1.1. Measure 1 – Vascular Plants

Monitoring Question

Are there trends in the diversity, cover and structure of the vascular plant community in forest ecosystems of Gwaii Haanas that we would not expect due to random fluctuations?

Context

In many ways, the vegetation on Haida Gwaii is similar to that of comparable areas in southeast Alaska or the adjacent mainland coast. The archipelago's insular biota is relatively impoverished, however, with only 741 recorded vascular plant species (Cheney et al. 2007) compared to ~2,300 on the adjacent mainland (Douglas et al. 1998). As an isolated archipelago, Haida Gwaii is also home to regional endemics and major disjunctions.

Gwaii Haanas' low elevation forests fall within the coastal western hemlock biogeoclimatic zone. Forests on the leeward east side fall into the Wet Hypermaritime Coastal Western Hemlock subzone (CWHwh). This is classic coastal rainforest, dominated by large western hemlock (*Tsuga heterophylla*), Sitka spruce (*Picea sitchensis*) and red cedar (*Thuja plicata*) trees. The windward west coast falls into the Very Wet Hypermaritime Coastal Western Helmolck subzone (CWHvh). Extreme exposure makes these forests, which are dominated by cedar and hemlock, more boggy and stunted (Green and Klinka 1994). Throughout Haida Gwaii, the structure and composition of the forested understorey has been greatly influenced by the browsing of introduced deer (Stockton et al. 2005; Gaston et al. 2007 a,b; see Section 2.1.3 below).

Methods

In 1998, vegetation plots were established in conjunction with the Research Group on Introduced Species (RGIS) project (Section 2.1.3) in the two lower elevation forest types found within Gwaii Haanas: CWHwh and CWHvh. The percent cover of every species was measured for various height strata in 3.6 m radius plots on Kunga Island (CWHwh, N = 40 plots) and at Louscoone Point (CWHvh, N = 20 plots) (Figure 1). Lists of all species present (richness) were also recorded from larger (25 m radius)

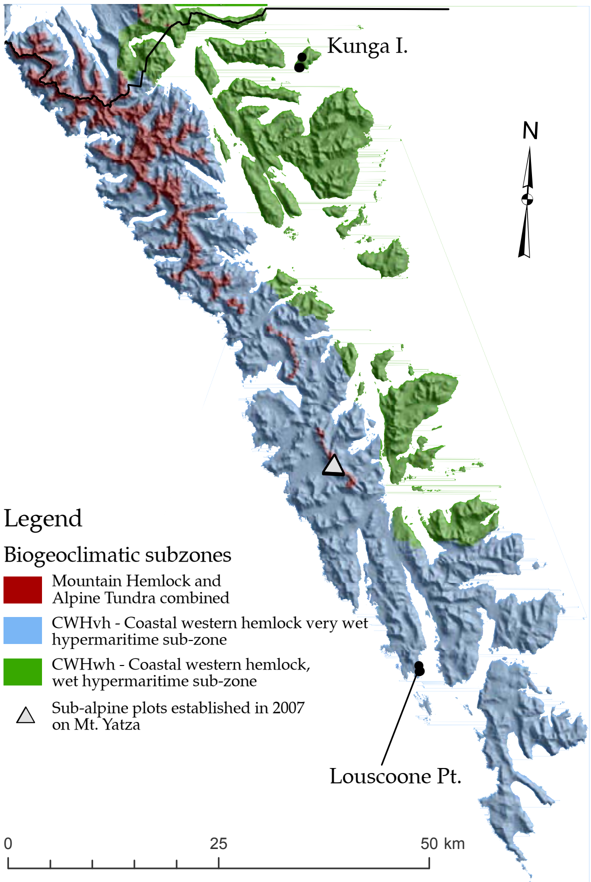


Figure 1. Low elevation forest by biogeoclimatic subzone and the location of permanent vegetation plots. Data from Research Branch, British Columbia Ministry of Forests. <ftp://ftp.elp.gov.bc.ca/dist/arcwhse/wildlife/> (last accessed May 31, 2005).

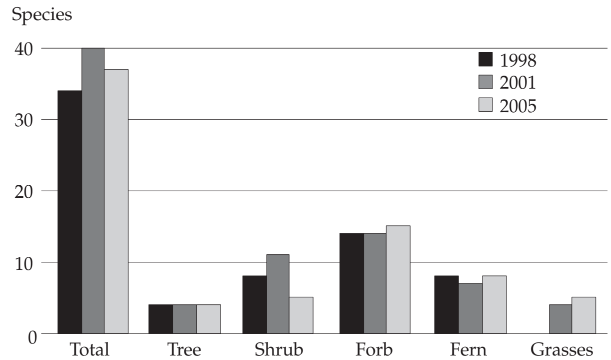


Figure 2. Species richness (total species recorded) in the 25m radius plots (N=10) at Kunga Island.

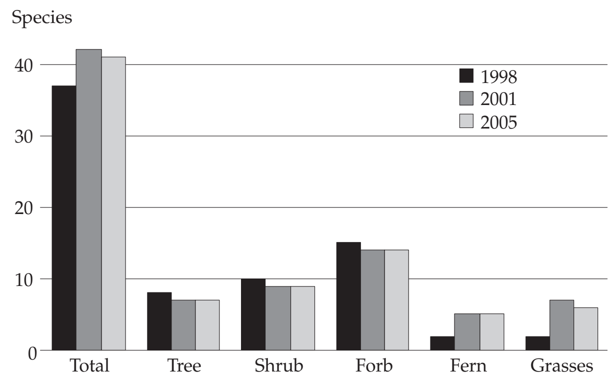


Figure 3. Species richness (total species recorded) in the 25m radius plots (N=10) at Louscoone Point.

plots (N = 20 plots on Kunga and N = 10 plots at Louscoone Point). In 2007, four exclosures with matching control points were established on the sub-alpine (unforested) slopes of Mount Yatza.

Our EI metrics for plants are species richness, percent vegetation cover, and percentage similarity. For our analyses, we use three strata: 0-50 cm, 50-150 cm and 150-400 cm. For each metric we looked for a trend over time from 1998 to 2005. For all power analyses we have used a standard of a = 0.05 and b = 0.20.

Results

Concerning species richness, plant diversity has remained stable since 1998 (Figures 2 and 3). There is no significant trend in species richness (total number of species recorded) in the vegetation plots on Kunga Island (GLM: N = 3 years, R^2 = 0.18, P = 0.72) or at Louscoone Point (GLM: N = 3 years, R^2 = 0.49, P = 0.51).

For percent vegetation cover, we used single group repeated measures to assess change in cover at permanent sample plots. Within each stratum we looked at total percent cover and percent cover by life form (trees, shrubs, forbs, ferns, and grasses). Between 1998 and 2005, we found little significant change in the percent cover in the plots at both Kunga Island and Louscoone Point (Tables 3 and 4). On Kunga Island, we recorded a significant decline in the cover of shrubs in the lowest stratum (0-50 cm, single group repeated measures ANOVA: df = 2, F = 21.62, adjusted P<0.00001 [by Geisser-Greenhouse adjustment]). There was a significant decline from 1998 to 2001, and again from 2001 to 2005 (Scheffe's multiple comparison test: P<0.05). At Louscoone Point, we detected a decline in the cover of ferns in the lowest stratum (0-50 cm: df = 2, F = 4.62, adjusted P<0.042) and the cover of trees in the upper 2 strata (50-150 cm: df = 2, F = 4.27, adjusted P=0.039; 150-400 cm: df = 2, F = 8.91, adjusted P=0.0036). Figures 4 and 5 show that there has been a very slight decline in the mean cover of vegetation in virtually all life forms at all strata. The only

Table 3. Results of a repeated measures analysis of percent vegetation cover on Kunga Island (N = 40 plots).

Lifeform	Mean % cover	Standard Deviation	F	P	Post hoc test Scheffe's multiple comparison	Detectable change in % cover (4 year sampling period)	% detectable change (4 year sampling period)
0-50 cm stratum							
Trees	15.83	27.64	1.64	0.208*		5.6	35.4
Shrubs	0.54	0.85	21.62	0.000001*	1998>2001>2005	0.3	55.8
Forbs	0.16	0.57	2.46	0.114*		0.2	123.1
Ferns	1.66	6.86	1.41	0.243*		2.1	126.3
Grasses	4.40	15.76	2.62	0.106*		1.7	38.6
Total	22.59	31.71	1.20	0.297*		6.0	26.6
50-150 cm stratum							
Trees	10.67	20.02	0.30	0.617*		6.4	60.0
Shrubs	0.18	0.74	0.91	0.364*		0.1	56.3
Total	10.97	20.39	0.37	0.576*		6.5	59.3
150-400 cm stratum							
Trees	13.42	20.92	0.47	0.503*		10.8	80.5
Shrubs	1.81	5.30	0.18	0.688*		1.7	93.8
Total	15.23	23.39	0.53	0.477*		11.0	72.2

* Geisser-Greenhouse adjusted

Table 4. Results of a repeated measures analysis of percent vegetation cover at Louscoone Point (N = 20 plots).

Lifeform	Mean % cover	Standard Deviation	F	P	Post hoc test Scheffe's multiple comparison	Detectable change in % cover (over a 4 year sampling period	% detectable change (over a 4 year sampling period)
0-50 cm stratum							
Trees	2.88	2.77	1.12	0.314*		0.4	14.3
Shrubs	17.98	11.73	0.16	0.801*		4.3	23.9
Forbs	0.005	0.09	2.09	0.165*		0.01	280
Ferns	0.36	0.96	4.62	0.042*	1998>2005 1998=2001 2001=2005	0.2	55.6
Grasses	1.1	3.44	2.61	0.116*		0.5	45.5
Total	22.32	13.17	1.3	0.284		4.4	19.7
50-150 cm stratum							
Trees	3.11	3.59	4.27	0.039*	1998>2005 1998=2001 2001=2005	1	32.2
Shrubs	15.36	20.98	1.44	0.250*		3.8	24.7
Total	18.47	21.58	3.32	0.068*		3.9	21.1
150-400 cm stratum							
Trees	11.93	10.51	8.91	0.004*	1998=2001>2005	0.9	7.5
Shrubs	7.4	11.21	0.16	0.703		8.6	116.2
Total	19.33	14.77	0.09	0.788		8.7	45

* Geisser-Greenhouse adjusted

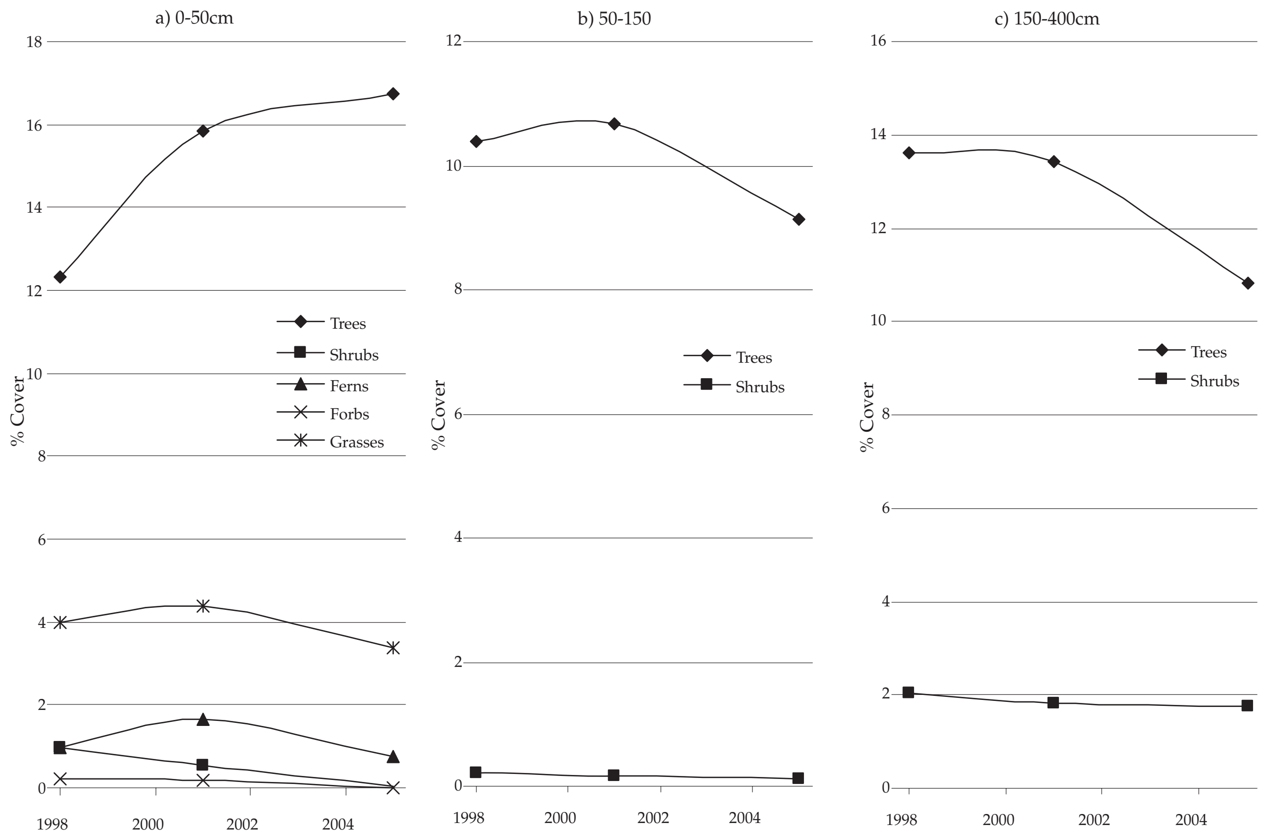


Figure 4. Mean percent cover on Kunga Island of the various life forms (trees, shrubs, forbs, ferns and grasses) in 3 strata: a) 0-50 cm, b) 50-150 cm, and c) 150-400 cm (N=40 plots).

exceptions are an increase in the cover of trees in the lowest stratum at Kunga and a possible increase in shrub cover in the upper stratum at Louscoone. Error lines were not included in Figures 4 and 5 because of the magnitude of the between plot variation (Tables 3 and 4).

Power analysis, based on the most recent four year sampling period (2001 to 2005), indicates that using this analysis we can detect quite small changes in the percent cover of individual life forms and total cover by stratum (shown as "detectable change in % cover" in Tables 3 and 4). However, because the actual percent cover values are so low (e.g. the mean percent cover of shrubs in the lowest stratum on Kunga is only 0.54%), this small actual detectable change in cover translates to a fairly large "percent detectable change" (Tables 3 and 4). In the lowest stratum on Kunga, for example, although we have the power to detect a change of 0.3 percent cover for shrubs, because the mean cover of shrubs is so low (0.54), this translates to a detectable change of 55.8%.

To determine if the makeup of the vegetation communities are changing over time, we used percentage similarity (or Renkonen index). This is a quantitative similarity index, calculated using vegetation cover measurements, which is used to compare communities (Krebs 1999). The index ranges from 0 (no similarity) to 100 (complete similarity). Using the percent cover for all species, we found that the vegetation community is getting more divergent over time in all strata (the percentage similarity was significantly lower from 2001 to 2005 than from 1998 to 2001), both a Kunga and Louscoone (Tables 5 and 6). Power analysis indicates that we can detect a change in percentage similarity between sampling periods of 8% to 28%, depending upon the stratum.

To determine if the structure of the vegetation community is changing over time, we again used percentage similarity. This time we assessed the similarity in cover of the various life forms (trees, shrubs, forbs, ferns, grasses) rather than individual species. By looking at all the strata simultaneously (each life form / strata combination was entered into the analysis as an independent entity), we get an indication of change in vertical structure as well as composition. There has been a significant

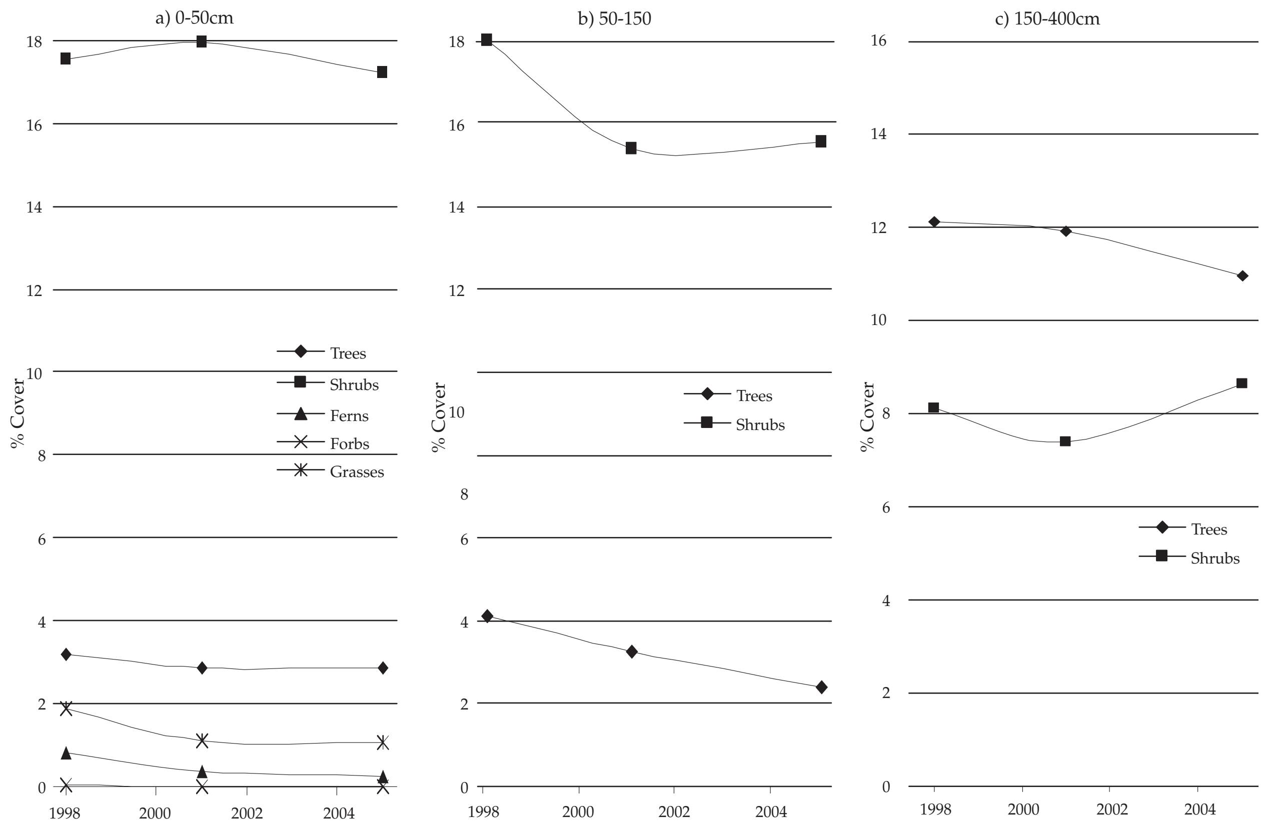


Figure 5. Mean percent cover at Louscoone Point of the various life forms (trees, shrubs, forbs, ferns and grasses) in 3 strata: a) 0-50 cm, b) 50-150 cm, and c) 150-400 cm (N=40 plots).

Table 5. Results of a percentage similarity analysis conducted using percent cover for all species on Kunga Island.

Stratum	Mean percentage similarity 1998-2001	Mean percentage similarity 2001-2005	Standard deviation 1998-2001	Standard deviation 2001-2005	T*	P (2 tailed)	% detectable change (between sampling periods)
0-50 cm	75.49	37.65	32.47	45.52	-4.97	0.00001	28.21
50-150 cm	94.41	62.23	17.29	48.83	-4.55	0.00005	21.08
150-400 cm	99.14	46.254	2.31	48.54	-6.73	< 0.00001	21.59

* paired t-test

Table 6. Results of a percentage similarity analysis conducted using percent cover for all species at Louscoone Point.

Stratum	Mean percentage similarity 1998-2001	Mean percentage similarity 2001-2005	Standard deviation 1998-2001	Standard deviation 2001-2005	Z*	P (2 tailed)	% detectable change (between sampling periods)
0-50 cm	87.61	80.49	14.67	18.45	3.06	0.0022	8.15
50-150 cm	86.15	74.07	24.28	32.11	3.02	0.0025	16.48
150-400 cm	9.19	79.12	5.27	29.30	3.25	0.0012	19.33

* Wilcoxon signed rank test

change in the vegetation structure at both Kunga and Lousecoone (Table 7). The change is very marked at Kunga, with a percentage similarity of only 43% between 2001 and 2005).

Although no thresholds have yet been established for the EI metrics associated with plants in forested ecosystems, we assess the status of the measure to be in "poor" condition (red) because of the effects of browsing by introduced deer (Section 2.1.3.). While species richness and percent vegetation cover are both quite stable, percentage similarity analyses indicate that both the composition and structure of the vegetation community continue to change. These trends are consistent for both the CWHvh and the CWHwh forest plots. Respecting the precautionary approach, the overall trend for plants in forests is "deteriorating," based on the metric with the most cause for concern.

The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots. We will be applying this methodology to the vegetation plots in forested habitat to ensure consistency and sufficient statistical power to detect changes. This will result in some changes to the metrics and protocol detailed here. The new bioregional study design for permanent vegetation plots will be retrofit to the plots previously established on Kunga Island and at Louscoone Point (both inside and outside of exclosures). The new plots should be surveyed in 2008 and every five years thereafter.

2.1.2. Measure 2 - Non-native Plants

Monitoring Question

Are cover and richness of the non-native plant community in forested ecosystems increasing?

Context

Of the 741 vascular plant species recorded on Haida Gwaii, 203 are not native to the islands (Cheney et al. 2007) and new species continue to colonize. Most of these species are early successional plants, adapted to habitats that have been disturbed in some way (e.g., towns, industrial sites or shorelines). Very few non-native plants have become established in undisturbed forested areas. Although only a small proportion of non-native species are considered invasive, rapidly expanding and threatening native flora and fauna through competition and habitat alteration, the presence of any non-native plant changes the floristic make-up of the area it colonizes.

Methods

In 1998, vegetation plots were established in conjunction with the RGIS project (Section 2.1.3. - Deer) in two forest types within Gwaii Haanas: the relatively drier forest characteristic of the east side (CWHwh subzone) and the very wet forest characteristic of the exposed south and west sides (CWHvh). The percent cover of each species, native or non-native, was measured for various height strata in 3.6 m radius plots on Kunga Island (CWHwh, N = 40 plots) and at Louscoone Point (CWHvh, N = 20 plots). Lists of all species present (richness) were also recorded from larger plots (25 m radius, N = 20 at Kunga and N = 10 at Louscoone). Surveys were conducted in 1998, 2001 and 2005.

The EI metrics for non-native plants are percent cover and species richness. By definition, non-native plants do not naturally occur within these ecosystems. The lower threshold (going from good to fair: green-yellow) has, therefore, been set at zero for both non-native species richness and percent cover. The upper threshold (going from fair to poor: yellow-red) has tentatively been set at 10% for percent cover and 20% for species richness. For species richness, this means the status becomes red when >20% of the species found within the plots are not native.

Results

To date, no non-native plants have been recorded in the smaller (3.6 m radius) percent cover plots

Table 7. Results of a percentage similarity analysis conducted using percent cover by life form across three strata (0-50 cm, 50-150 cm and 150-400 cm) on Kunga Island and at Louscoone Point.

Location	Mean percentage similarity 1998-2001	Mean percentage similarity 2001-2005	Standard deviation 1998-2001	Standard deviation 2001-2005	T*	P (2 tailed)	% detectable change (between sampling periods)
Kunga	91.76	42.72	9.61	42.24	7.56	<0.00001	19.83
Louscoone	90.37	82.38	5.34	9.17	3.848	0.0001	6.43

* Paired t-test

at either Kunga or Louscoone. The status based on this measure is therefore assessed as green. At Louscoone, there were also no non-native plants found in the larger (25 m radius) species richness plots. At Kunga, however, two to four non-native plants have been recorded during the surveys conducted in 1998, 2001 and 2005. Five different species are represented, including bull thistle (*Cirsium vulgare*), marsh cudweed (*Gnaphaluim uliginosum*), wall lettuce (*Lactuca muralis*), wood groundsel (*Senecio sylvaticus*) and stinging nettle (*Urtica dioica*). These non-native species represent between 5 to 9% of the total plant species recorded in the survey plots in any year (Table 8). In the most recent survey (2005), 5.1% of the species found were non-natives. The status for this metric is therefore assessed to be yellow. Following the precautionary principle, we assess the overall status for non-native plants in forested ecosystems to be yellow, based on the metric with the poorest status. With only three years of data, trend analyses cannot be run.

The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots. We will be applying this methodology to the vegetation plots in Gwaii Haanas to ensure consistency and sufficient statistical power to detect changes. This will result in some changes to the metrics and protocol detailed here.

Table 8. Species richness found in the 25 m radius plots on Kunga Island (N = 20) in 1998, 2001 and 2005.

Stratum	1998	2001	2005
Number of non-native plants	3	4	2
Total number of plants	37	44	39
% non-native plants	8.1%	9.1%	5.1%

2.1.3. Measure 3 - Introduced Deer

Monitoring Question

Is the presence of introduced deer altering vegetation communities and the associated fauna in forested ecosystems of Gwaii Haanas and are the effects increasing?

Context

Prior to European contact there were no native deer on Haida Gwaii, save a small (now extinct) relict population of Dawson caribou (*Rangifer tarandus dawsoni*) (Burls et al. 2004). Because there were also no key large predators (e.g. wolf (*Canis lupus*), cougar (*Puma concolor*)), when Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) were introduced in the late 19th century, the deer population expanded and spread unchecked until it colonized all but the most remote islands. By 1996, deer densities were estimated at ~32 per km^2 on offshore islands (Stockton et al. 2005). There are no demographic data for total deer population, but dendrochronolgy work indicates that after the initial phase of colonization, deer have maintained a constant and high browsing pressure on the island's vegetation, suggesting a lack of marked fluctuation in deer densities (Vila and Martin 2007).

The presence of deer on Haida Gwaii has greatly influenced the vegetation (Pojar et al. 1980; Daufresne and Martin 1997; Stockton et al. 2005; Gaston et al. 2007 a). In forested habitats, the end result now found over much of the archipelago is the cathedral-like, old growth forest with little or no understorey. This is in stark contrast to the impenetrable forest with dense understorey found on the adjacent mainland coast and reported by early visitors to Haida Gwaii prior to or soon after deer colonization (Pojar and Banner 1984).

In 1995, a group of local and international scientist formed the Research Group on Introduced Species (RGIS) to explore changes in the ecology of Haida Gwaii caused by the introduction of non-native species. To date, the group has primarily focused on the impacts of deer in forested ecosystems. The most noticeable deer-induced modification is a browse line at between 1.1 and 1.5 m (Vila and Martin 2007). This is the maximum height that deer can reach, with most plants below this line showing effects of browsing. To assess the effects of deer on vegetation, Stockton (2007) compared the cover and diversity of undersotrey plants on islands without deer, islands where deer have been present for <20 years, and those where deer have been present for >50 years. His study found that both cover of vegetation and plant diversity declined with increased duration of deer presence. The outcome of deer browsing on plant community composition is a result of the combined effects of diet preference of the deer and differential recovery among the plant species. Deer browsing has thus radically altered the structure of many plant associations, and has greatly reduced or virtually eliminated preferred forage species in many areas. Among the species most affected are deciduous shrubs, ferns and broad-leaved herbs (Pojar et al. 1980; Daufresne and Martin 1997). For example, the deer's heavy browsing on regenerating sprouts has completely stopped stem replacement on preferred shrubs (e.g., *Vaccinium* spp. - Vila and Martin 2007).

The dramatic change in vegetation structure and composition caused by deer browsing has impacted ecosystem function, leading to indirect effects via physical habitat modifications and trophic cascades. By changing vegetation cover and vertical structure, deer destroy important habitat requisites of various species of wildlife, thereby affecting many other animal populations (Rooney and Waller 2003). On Haida Gwaii, this has been documented for both songbirds (Allombert et al. 2005 a) and forest invertebrates (Allombert et al. 2005 b). Deer browsing has cultural impacts as well. Many of the plant species most affected by deer browsing are culturally significant to the Haida for food, medicine, building or ceremonial purposes (e.g., red cedar, berry plants – Turner 2004).

Methods

In 1998, as part of the RGIS project, deer exclosures (measuring 15 m x 15 m) were erected in two forest types in Gwaii Haanas (Figure 1), both in the coastal western hemlock biogeoclimatic zone (Krajina 1965). A set of three exclosures and matching control (unenclosed) plots were established on Kunga Island. This area falls within the relatively drier forest subzone characteristic of the east side of Gwaii Haanas (CWHwh). A second set of three exclosures and matching controls was established at Louscoone Point, representing the very wet forest characteristic of the exposed west and south coasts of Gwaii Haanas (CWHvh).

Another component of the RGIS project involved culling deer from two offshore islands: SGang Gwaay (130 ha, CWHvh, initiated in 1998) and Reef Island, just north of the Gwaii Haanas boundary (249 ha, CWHwh, initiated in 1997). These islands were chosen for their isolation, on the assumption that future deer immigration would be limited. Both islands are large enough to support typical old-growth interior forest and both showed signs of intensive deer browsing. Although the deer have never been entirely eliminated from these islands, by 2001, <5 deer were estimated to remain on both islands (Gaston et al. 2007 b) and the numbers have since been maintained at these very low levels. Beginning in 2002, a cull program has also been established on Hotspring Island (17 ha, CWHwh).

To assess the effects of deer browsing, vegetation response has been monitored at both the exclosure sites and the cull islands. Permanent vegetation plots have been established in which we record species presence (richness) and vegetation cover at different height strata. However, because the history of browsing cannot be eliminated from these areas, removing or decreasing deer using exclosures or culls will not necessarily lead to the recreation of biotic conditions that would occur without deer. Legacy effects, such as the depletion of the species pool and seed bank or the persistent nutrient and soil chemistry changes, may change the trajectory of recovery and resultant vegetation composition and structure (Rooney and Waller 2003). As a result, while comparing vegetation in areas with and without current deer browsing still provides a good indication of whether or not deer are having an effect on the current condition of vegetation in Gwaii Haanas (status), it is quite difficult to assess the trend in deer impacts (i.e., answer the question: are the impacts of deer continuing to increase, or have they stabilized?).

Initially when deer browsing is eliminated or reduced in an area (using exclosures or culls), the changes over time in the vegetation in these areas reflects the recovery from deer browsing. Changes in the vegetation outside of these areas (where deer pressure remains) may reflect a trend in deer impacts, but they may also reflect other environmental trends, such as climate change. Only once the areas without deer browse have finished recovering (i.e. come to a steady composition and structure), can we use vegetation changes to assess trend in deer impacts. Then we can look at the *change* in difference between the vegetation inside and outside these areas to assess the trend in deer impacts. Both the vegetation inside and outside of these areas will be affected by the other environmental factors, but deer browsing will only affect the vegetation outside of the deer excluded areas. Until the vegetation in the exclosure and culled areas has regenerated, the simplest way to assess the trend in deer impacts is to assess the trend in the deer population. We are, therefore, considering incorporating pellet count transects into our vegetation monitoring program. They are an efficient and effective method to detect changes in relative abundance of deer (BC 1998).

Results

Our EI metrics for deer effects on forest vegetation are species richness, percent vegetation cover, and percentage similarity. For our analyses, we compared data collected in 2005 from vegetation plots (15m x 15m) inside versus outside exclosures at Kunga (N = 3 exclosures/control pairs) and Louscoone (N = 3). For all power analyses we have used a standard of $\alpha = 0.05$ and $\beta = 0.20$.

For species richness analysis, we assembled a combined species lists for the three exclosures and control plots in each forest type. In plots subject to browsing by introduced deer, we recorded fewer native species than in plots where deer have been excluded for seven years (1998 to 2005). In 2005 on Kunga Island, we recorded 22 species inside exclosures and 16 species outside. At Louscoone point, 18 species were recorded inside and 13 outside. A reduction in diversity was noted for all life forms except for trees (Figures 6 and 7).

Only two non-native species were recorded in the paired plots. On Kunga Island, both bull thistle (*Cirsium vulgare*) and wall lettuce (*Lactuca muralis*) were found inside exclosures.

For percent cover analysis, we used paired t-tests to compare percent vegetation cover inside the exclosures in 2005 to that outside. We looked at total percent cover in each of three strata: 0 to 50 cm, 50 to 150 cm and 150 to 400 cm. We also compared percent cover by life form (trees, shrubs, forbs, ferns, and grasses) in each stratum. The only significant difference we found was a decrease in the cover of shrubs in the lowest stratum (0 to 50 cm) at Louscoone Point (paired t-test: N = 3 plots, T = 11.43, P = 0.007) (Tables 9 and 10). At least part of the reason we were unable to detect a statistically significant difference in many instances, was the extreme variability between plots. Standard deviations are often as large as the associated means. This is reflected in the low power of the tests (high % detectable

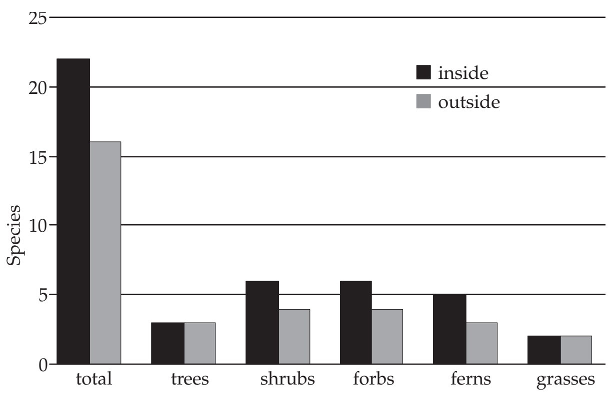


Figure 6. Species richness inside and outside of deer exclosures on Kunga Island.

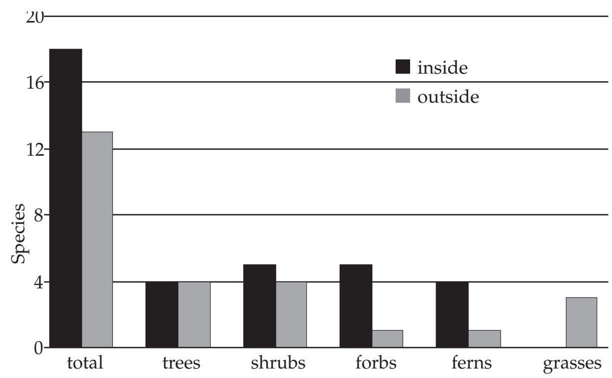


Figure 7. Species richness inside and outside of deer exclosures at Louscoone Point.

Table 9. Analysis comparing percent vegetation cover inside and outside exclosures on Kunga Island in 2005 (N=3 paired pairs).

Lifeform	Mean % cover inside	Mean % cover outside	Standard Deviation inside	Standard Deviation outside	T*	P	% detectable difference
0-50 cm							
Tree	16.67	18.33	17.56	17.56	-1	0.423	28
Shrubs	48.33	0	37.86	0	2.21	0.58	126.7
Forbs	0	0	0	0			
Ferns	0.67	0	1.15	0	1	0.43	280.2
Grasses	0	5	0	8.66	-1	0.43	
Total	65.67	23.33	20.65	20.21	2.15	0.164	83.9
50-150 cm							
Tree	4	0	6.93	0	1	0.423	280.2
Shrubs	38.33	0	37.53	0	1.77	0.219	158.4
Total	42.33	0	38.42	0	1.91	0. 197	146.8
150-400 cm							
Tree	1.67	0	2.89	0	1	0.423	280.2
Shrubs	25	0	25	0	1.73	0.225	161.7
Total	26.67	0	27.54	0	1.68	0.235	167

Table 10. Analysis comparing percent vegetation cover inside and outside exclosures at Louscoone Point in 2005 (N=3 paired pairs).

Lifeform	Mean % cover inside	Mean % cover outside	Standard Deviation inside	Standard Deviation outside	T*	P	% detectable difference
0-50 cm							
Tree	10.33	13.33	13.05	15.28	-0.26	0.816	307.9
Shrubs	79.67	18.33	11.24	15.28	11.43	0.007	18.9
Forbs	0.67	0	1.15	0	1	0.423	280.2
Ferns	0	0	0	0			
Grasses	0	0	0	0			
Total	90.67	31.67	18.15	28.87	3.622	0.068	50.3
50-150 cm							
Tree	1.67	0	2.89	0	1	0.423	280.2
Shrubs	43.33	0	45.09	0	1.66	0.238	168.3
Total	45	0	47.7	0	1.63	0.244	171.4
150-400 cm							
Tree	0.33	0	0.58	0	1	0.423	280.2
Shrubs	0	0	0	0			
Total	0.33	0	0.58	0	1	0.423	280.2

* paired t-test

differences) (Tables 9 and 10). Figures 8 and 9 indicate that, despite the variability, the effects of deer are likely very biologically meaningful. In the lowest strata (0 to 50 cm), the vegetation cover is dramatically lower outside the exclosures, where it is still subject to browsing. On Kunga Island, the mean total cover is 66% inside the exclosure, compared to 23% outside. Shrubs are non-existent in this stratum in the presence of deer, compared to 48% cover inside the exclosures. In the lowest stratum at Louscoone Point, the mean total percent cover is 91% inside the exclosure compared to 32% outside. Again, the shrubs are dramatically reduced (80% inside vs. 18% outside). In the middle stratum (50 to 150 cm), no vegetation cover remains in the presence of deer. Inside the exclosures, the mean vegetation cover is 42% on Kunga and 45% at Louscoone. This is almost entirely composed of shrubs. On Kunga, the situation in the upper stratum (150 to 400 cm), mirrors that in the middle stratum. The vegetation cover, composed mostly of shrubs inside the exclosures (mean total cover = 27%), is totally eliminated in the presence of deer. On Louscoone, there is virtually no vegetation cover in the upper stratum either inside or outside of the exclosures. Although the x-axis on these figures indicates mean percent cover (N = 3 paired plots), error lines are not included in Figures 8 and 9 because the between plot variation is so large (Tables 9 and 10).

To determine if the makeup of the vegetation communities are different inside and outside the exclosures, for a similarity index we used percentage similarity. This is a quantitative similarity index, calculated using vegetation cover measurements (Krebs 1999). The index ranges from 0 (no similarity) to 100 (complete similarity). Using the percent cover for all species, we found that the mean percentage similarity of the vegetation community inside versus outside the exclosures on Kunga Island was 28%, 67% and 33% for the three strata (0 to 50 cm, 50 to 150 cm, and 150 to 400 cm). At Louscoone Point, the mean percentage similarities were 70%, 33% and 67% respectively. T-test analysis found that the indexes were not significantly different from a similarity of 100%. Again, however, this was at least in part due to the large variability between plots means and the resultant low power (Table 11). Power analysis indicates that using this analysis we cannot determine the plots inside and outside of the exclosures to be different until the percentage similarity drops below between 48.3 and 6.6, depending on the stratum (Table 11).

To determine if the structure of the vegetation community is different inside and outside the exclosures, we again used percentage similarity. This time we assessed the similarity in cover of the various life forms (trees, shrubs, forbs, ferns, grasses) rather than individual species. By looking at all the strata simultaneously (each life form / strata combination was entered into the analysis as an independent entity), we get an indication of the similarity in vertical structure as well as composition. We found that

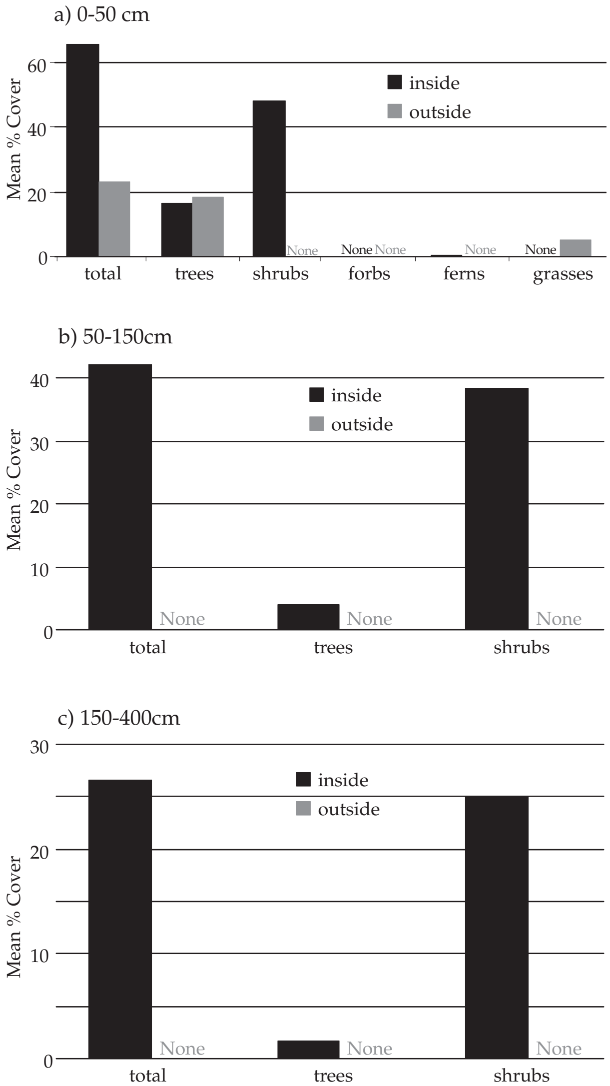


Figure 8. Mean percent cover found inside and outside 3 exclosures on Kunga Island. Percent cover is shown for the various life forms (trees, shrubs, forbs, ferns and grasses) in 3 strata: a) 0-50 cm, b) 50-150 cm, and c) 150-400 cm.

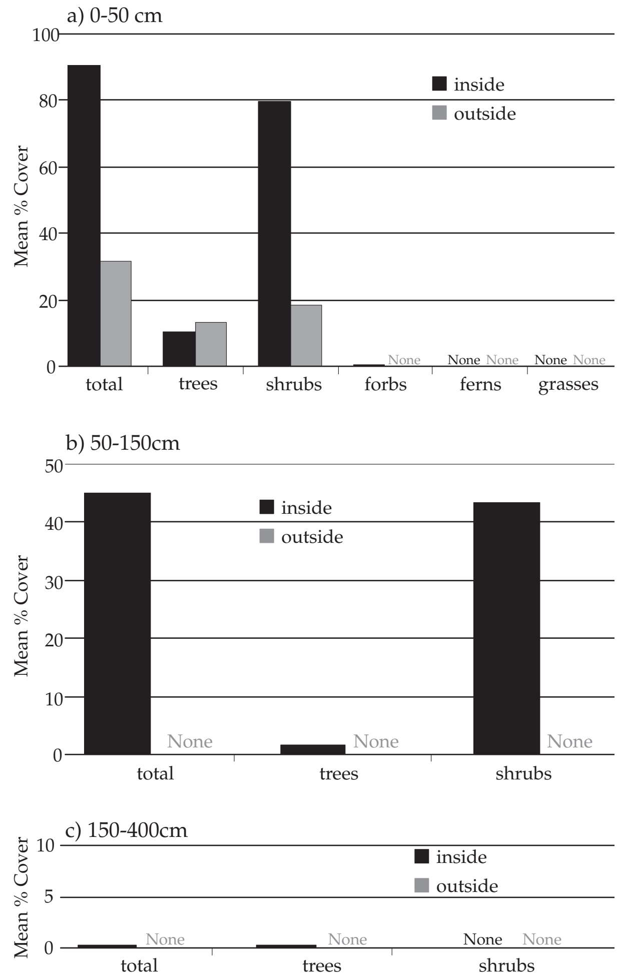


Figure 9. Mean percent cover found inside and outside 3 exclosures at Louscoone Point. Percent cover is shown for the various life forms (trees, shrubs, forbs, ferns and grasses) in 3 strata: a) 0-50 cm, b) 50-150 cm, and c) 150-400 cm.

Table 11. Results of a percentage similarity analysis (using percent cover for all species) comparing vegetation communities inside and outside deer exclosures on Kunga Island and Louscoone Point in 2005.

Strata	Mean similarity inside vs outside	Standard Deviation similarity inside vs outside	T	P	Maximum detectable mean similarity
Kunga					
0-50 cm	27.78	48.11	-2.6	0.061	22.2
50-150 cm	66.67	57.74	-1	0.211	6.6
150-400 cm	33.33	57.74	-2	0.092	6.6
Louscoone					
0-50 cm	70.4	31.974	-1.6	0.125	48.3
50-150 cm	33.33	57.74	-2	0.092	6.6
150-400 cm	66.67	57.74	-1	0.211	6.6

the mean percentage similarity inside versus outside the exclosures was only 30 on Kunga Island and 57 at Louscoone Point (Table 12). Even with the large variation between plots, this is significantly different from 100% similarity at Louscoone, and nearly so at Kunga. Power analysis indicates that using this analysis; the percentage similarity of the plots inside versus outside of the exclosures must be below 74.5 at Louscoone for us to detect a difference. Because of the much larger variation between plots on Kunga (as indicated by the standard deviations), here the percentage similarity must be below 10.6 to be statistically significant (Table 12).

We show that deer dramatically affect the cover, structure and diversity of vegetation in forested ecosystems, both in the wetter CWHvh (Louscoone Point) and in the drier CWHwh (Kunga Island). Because the exclosures were constructed in 1998, when the area had already been subject to heavy browsing for several decades, the difference recorded inside and outside the exclosures document the recovery of vegetation seven years (1998-2005) after deer browsing pressure was removed. These results were mirrored by the findings of Gaston et al. (2007 b), who compared vegetation on Reef Island and SGang Gwaay prior to and five years after the culling of deer from the islands.

Deer are not native to Haida Gwaii. Therefore, EI status would be considered "good" (green) for this measure when no effects of deer are recorded. Although a second threshold (going from fair to poor: yellow-red) has not yet been determined for this measure, we assess the current status to be "poor" (red), based on the abundance of research demonstrating the profound and far-reaching effects of deer in forested ecosystems. The trend is undetermined.

The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots. We will be applying this methodology to the vegetation plots (exclosures, controls and cull islands) in Gwaii Haanas to ensure consistency and sufficient statistical power to detect changes. This will result in some changes to the metrics and protocol detailed here. As a bioregion, we are also assessing protocols to monitor the relative abundance of deer using pellet counts.

2.1.4. Measure 4 - Forest Structure

Monitoring Question

The monitoring question is yet to be determined.

Context

British Columbia Ministry of Forests and Range (MoF) has established Permanent Sample Plots (PSPs) throughout the province's forests, in natural, experimental and silviculturally-treated stands (BC 2003). The purpose of PSPs established in natural stands is to provide long-term, local data on rates of growth, mortality, changes in stand structure, and stand development. Foresters use these data to develop growth and yield models and to validate site index curves. This information is used for management planning and to determine annual allowable cuts.

In the 1960s, 280 PSPs were established in Haida Gwaii, 154 of which are within Gwaii Haanas. Upon establishment, each plot was ecologically classified to the site series level with the soil and forest floor described from a soil pit dug near the plot centre. The plots are remeasured approximately every 10 years following standard PSP methods (BC 2003) under contract to the MoF (so, are there 4 points in the time series ??). The last surveys on Haida Gwaii were completed in 2004 and 2005. Surveys include measurements of tree species composition, diameter, height, age, density, stand structure, damage, pathology and decay indicators. In the most recent round of surveys, data were also collected for standing coarse woody debris.

We are currently investigating the utility of this long-term and on-going dataset for measures that can track changes in forest structure in

Table 12. Results of a percentage similarity analysis (using percent cover by life form across three strata: 0-50 cm, 50-150 cm and 150-400 cm) comparing vegetation communities structure inside and outside deer exclosures on Kunga Island and Louscoone Point in 2005.

	Mean similarity inside vs outside	Standard Deviation similarity inside vs outside	T	P	Maximum detectable mean similarity
Kunga	30.45	45.97	2.6203	0.06	10.6
Louscoone	57.17	6.56	-11.3027	0.004	74.5

Gwaii Haanas. Some of these measures could be included in our EI Monitoring Program and could be reported on in the next SoPR in 2012.

2.1.5. Measure 5 - Forest Insects and Diseases

Monitoring Question

Is the frequency of occurrence and extent of forest insect pest outbreaks changing over time?

Context

Some insects have significant periodic influences on forest ecosystems. Many have larval stages that feed on the foliage of trees or bore into their trunks, often inducing tree death. Populations of these insects are frequently dependent on weather conditions, and, when conditions are favourable, insect epidemics can occur. Such outbreaks can reduced tree growth, cause deformities or kill trees. On Haida Gwaii, where stand-replacing fires are extremely rare, periodic outbreaks of insect defoliators, landslides and mass wind-throw are the major forces influencing forests.

Because such "pest" outbreaks have commercial effects, they have long been monitored. Anecdotal reports of outbreaks of forest insect pests on Haida Gwaii date back to 1931. Since 1949, systematic surveys have been conducted annually by the Canadian Forest Service (CFS) to determine the presence and abundance of insect pests. In 1995, the CFS decided to discontinue the Forest Insect and Disease Survey Program. Funding was obtained from the South Moresby Forest Replacement Account and British Columbia Ministry of Forests, and surveys have continued. In 2005, Gwaii Haanas began providing logistical support for these surveys by deploying pheromone traps for the western blackheaded budworm (*Acleris gloverana*).

The CFS has identified over 40 insect pests on Haida Gwaii (Vallentgoed 2000). While many of these can have potentially significant effects on forest ecosystems when they occur in high densities, most have never reached epidemic proportions. Two however, the western blackheaded budworm and the western hemlock sawfly (*Neodiprion tsugae*), regularly reach epidemic levels. A third pest, the spruce aphid (*Elatobium abietinum*) causes chronic damage to primarily older foliage of Sitka spruce in coastal stands on the east side of the Islands, particularly within Gwaii Haanas. Recently the damage has been sufficiently severe to be mapped during annual aerial surveys. Success of this insect has historically been linked to winter weather, the most severe outbreaks following mild winters. A fourth, yet to be identified pest, causes yellow cedar mortality at higher elevations and has recently been found in sea level stands in Gwaii Haanas. An exotic insect, the Cooley spruce gall (*Adelges cooleyi*), was introduced to Haida Gwaii when ornamental Douglas fir was imported. Active removal of most infested trees has largely contained this pest.

Outbreaks of the western blackheaded budworm have occurred in 1931, 1943-44, 1952-55, 1959, 1972-75, 1985-88 and 1996-2001 and have caused significant defoliation each time for between 1 and 6 years (Vallentgoed 2000; Nealis at al. 2004). This pest eats mostly new foliage of hemlock and to a lesser extent spruce. Severe infestations strip trees entirely of their foliage. Although no defoliation was recorded in 2006, many adults were captured in pheromone traps, suggesting that an outbreak may be imminent.

Most outbreaks are first detected in the southerly part of Gwaii Haanas before being found further north; hence operating pheromone traps in the southern part of Gwaii Haanas should allow for earlier detection of the next budworm outbreak. During the last two blackheaded budworm outbreaks, the western hemlock sawfly, which feeds on old foliage of conifers has simultaneously reached epidemic levels (Wood and Garbutt 1989; Turnquist et al. 2001). These combined outbreaks have resulted in relatively high mortality of defoliated trees, and a significant reduction in tree growth (Vallentgoed 1993). During the last outbreak, damage was concentrated within stands of regenerating western hemlock to a greater extent than in previous outbreaks.

It is not clear why these two species periodically reach epidemic proportions together, although weather conditions may play a significant role in larval survival and development, and dispersal. The collapse of outbreaks is thought to be due to increased parasitism of budworm eggs by other insects. The greater severity of recent outbreaks may thus be due, in part, to recent changes in climate. The extent and duration of these outbreaks, and the increasing prevalence in regenerating stands however, may be a consequence of forest management practices that have resulted in the dominance of young western hemlock stands on Haida Gwaii (Nealis et al. 2004).

Methods

Systematic aerial surveys are flown annually by CFS to get an over-view of forest insect pest outbreaks. These flights can lead to more intense ground surveys where resources permit. When evidence of an outbreak is found, defoliated areas are mapped and the intensity of defoliation (light, moderate, severe) is estimated. Affected areas are checked in the year following the collapse of the outbreak to determine the extent of the "gray" area (those forests that experienced significant mortality or tree top-kill).

Pheromone traps were operated on an experimental basis during 2005 and 2006 to detect the presence of western blackheaded budworm moths and perhaps predict the next initiation of the next outbreak. The plan is to deploy them again in 2007 but it is not certain whether their use will become part of the monitoring protocol.

Results
Since record keeping began, the onset of western blackheaded budworm outbreaks have occurred on average every 10.8 years (range 7 to 13 years) and have lasted an average of 3.1 years (range 1 to 6 years). While the last outbreak began in typical fashion 11 years after the onset of the previous one, its duration of 6 years was the longest yet recorded. Accurate records of the extent of defoliation are only available for the last two outbreaks (SoPR Section 3.1.4.). The extent of defoliation (55,050 ha) was also greater than that of the previous outbreak (44,300 ha), although the grey area was less (1,500 vs. 4,375 ha) during the last outbreak.

The EI metrics proposed for monitoring this measure are the return interval for outbreaks, the duration of outbreaks, the extent of defoliation, and the extent of grey area. As accurate information is only available for the last two outbreaks, it is not possible to establish thresholds or to identify any clear trends.

2.1.6. Measure 6 - Marbled Murrelet

Monitoring Question
What are the trends in estimated numbers of Marbled Murrelet counted from radar surveys at the marine openings of selected watersheds in Gwaii Haanas?

Context
Marbled Murrelet (*Brachyramphus marmoratus* - MAMU) is a high profile species in current land use planning coast-wide, including Haida Gwaii, because the fate of this species is linked directly to commercial forest practices (Burger 2002; Harfenist et al. 2002). MAMU are unique among seabirds in their non-colonial nesting on thick, moss-covered limbs of large, old-growth trees. Pairs lay a single egg and feeding trips (for small fish) for their chick usually occur during darkness or low light conditions.

Primarily because of nesting habitat loss due to old-growth logging, MAMU is federally listed as *Threatened*. The other main threats are depletion of their marine food resources and exposure to contamination by spilled oil. Evidence from other species of related (alcid) seabirds suggests that the Haida Gwaii birds are not experiencing the same food shortages as are those nesting along the southern British Columbia coast. Exposure levels of Gwaii Haanas' MAMU to oil contamination are unknown. Recovery of this species requires the identification and protection of areas of high nesting activity. Further, baseline population and density measurements are needed in Haida Gwaii to monitor species recovery and effectiveness of management actions (Harfenist and Cober 2005). Approximately 40% of prime MAMU nesting habitat on Haida Gwaii has already been logged (Holt 2004).

It has been difficult to census breeding birds due to their cryptic nest locations and nesting behaviour. The application of radar techniques enable counting incoming and outgoing birds at strategic inshore locations - estuarine portals to watersheds that host old-growth forest stands used for nesting. Radar tracking in June-July provides baseline population and density estimates needed to assess species recovery. Radar was used at dawn and dusk in 2004 and 2005 in Haida Gwaii (including Gwaii Haanas) through multi-agency cooperation (including Parks Canada) towards population estimates and the amount and location of coastal forest use (Harfenist and Cober 2005). Initial density estimates for this region were more similar to those from the west coast of Vancouver Island, and higher, compared to those recorded from the mainland coast.

Methods
The federal/provincial/forest industry funded collaboration aims achieve the following results (Harfenist and Cober 2005):

- establish population estimates at watershed and whole archipelago scales against which future trends could be compared;
- provide estimates of total population and density in watersheds for the Haida Gwaii/Queen Charlotte Islands Land Use Planning table, Haida Forest guardians and other land managers;
- compare the detection capabilities of two different radar systems; and

- train local people in radar survey technology.

The standardized coast-wide radar-based protocol, with some adaptations developed for Haida Gwaii, was used. Surveys are done either from a moored vessel or a tent on land - both nearby the estuaries of particular catchment (watershed) areas with old-growth stands (some logged) likely to host incubation and/or nestling birds in June and July. Radar scans are made around dawn and dusk when birds enter/leave the catchment area on provisioning trips for their chicks.

Results

A total of 26 Haida Gwaii watersheds were surveyed in 2004 and 16 of those resurveyed in 2005 were recommended for use as long-term population monitoring locations, including five within Gwaii Haanas (Figure 10). Characteristics of the five Gwaii Haanas locations (all boat-based) are listed in Table 13. These include ratings of watershed topography, habitat quality, distribution and accessibility for each survey location. The coast-wide pattern of more birds recorded at dawn surveys compared to dusk surveys is evident in Table 14. About 98% of the incoming birds were detected before sunrise – considered the best index of Marbled Murrelet populations (Burger 2001). Further, incoming detections tended to exceed outgoing detections during dawn surveys but not during dusk surveys. Windy Bay was one of the archipelago's three top catchments for numbers of bird detections.

Table 13. Characteristics of the long-term Marbled Murrelet monitoring locations in Gwaii Haanas, 2004 and 2005 (from Harfenist and Cober 2005). Boat-based radar was used at all locations.

Location	Flight Path	Catchment Reliability	Inland Habitat Quality	Catchment Size Class
Windy Bay	Fair	Fair	High	Medium
Bigsby	Good	Good	Low	Small
Klunkwoi	Good	Good	Low	Medium
Hutton	Good	Good	Low	Medium
Huston	Good	Good	Medium	Small

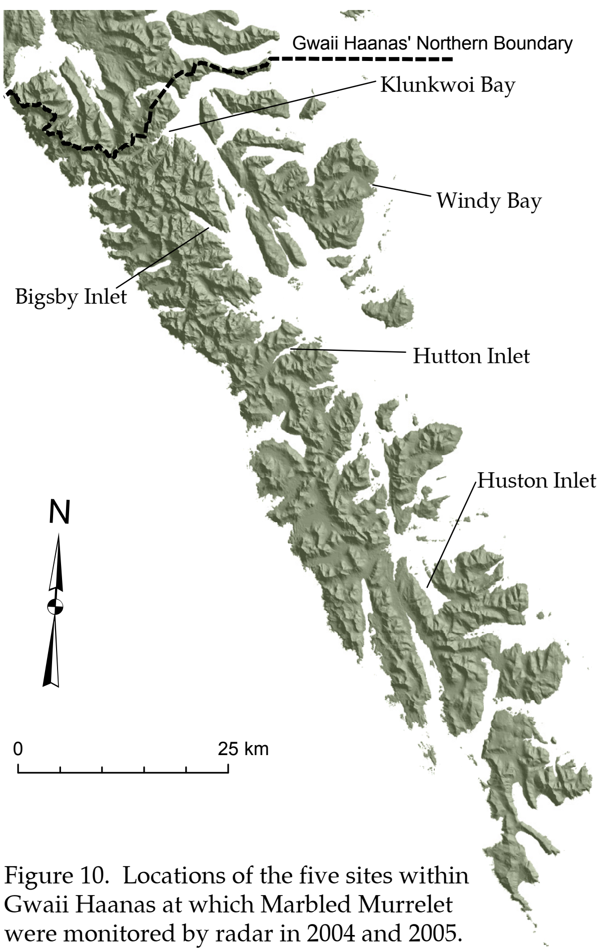


Figure 10. Locations of the five sites within Gwaii Haanas at which Marbled Murrelet were monitored by radar in 2004 and 2005.

Densities are the number of incoming or outgoing birds detected at survey sites per hectare of suitable nesting habitat within the associated catchment areas. Archipelago-wide, densities ranged from 0.313 to 0.060 birds per ha (Harfenist and Cober 2005). The catchment areas were defined in the Haida Gwaii/Queen Charlotte Islands (HG/QCI) Land Use Plan Marbled Murrelet habitat suitability maps. Four of the Gwaii Haanas catchments were among the 13 in which the densities of birds per unit habitat were calculated (Table 15). It must be mentioned, however, that the original mapping was thought to underestimate the amount of nesting habitat. A refinement of habitat suitability mapping, derived from air photo interpretations of the survey watersheds, is due for completion by April 2007. This new inventory will replace the nesting habitat suitability model outcomes.

There are insufficient MAMU population data to determine trends in Gwaii Haanas. However, based on models of habitat change from 1800 to 2000 in the MAMU section of the Environmental Conditions Report (Holt 2004) for the Haida Gwaii Land Use Plan (HG/QCI 2005), MAMU habitat has declined in Gwaii Haanas, primarily in the Lyell Island Landscape Unit. Most of the other Landscape Units within Gwaii Haanas have experienced only limited decreases in suitable habitat. Reasoned speculation is that MAMU populations have declined over the last 200 years at least in the north end of Gwaii Haanas.

Table 14. Numbers of Marbled Murrelets detected at dawn (a) and dusk (b) at monitoring locations in Gwaii Haanas, 2004 and 2005 (from Harfenist and Cober 2005).

(a) Dawn

Location	Date	Incoming	Outgoing
Windy Bay	07-Jul-04	203	148
	14-Jul-04	237	142
	27-Jun-05	192	138
	13-Jul-05	216	177
Bigsby	18-Jul-04	122	149
	27-Jul-04	63	76
	24-Jun-05	84	111
	17-Jul-05	142	109
Klunkwoi	08-Jul-04	45	96
	26-Jul-04	50	61
	22-Jun-05	55	100
	14-Jul-05	49	84
Hutton	09-Jul-04	88	45
	28-Jul-04	58	31
	24-Jun-05	95	62
	15-Jul-05	89	96
Huston	11-Jul-04	122	84
	17-Jul-04	120	99
	26-Jun-05	51	132
	16-Jul-05	113	78

(b) Dusk

Location	Date	Incoming	Outgoing
Windy Bay	06-Jul-04	30	57
	13-Jul-04	46	32
	26-Jun-05	73	74
	12-Jul-05	49	56
Bigsby	17-Jul-04	55	63
	26-Jul-04	3	25
	24-Jun-05	24	53
	16-Jul-05	54	47
Klunkwoi	07-Jul-04	23	86
	25-Jul-04	11	23
	21-Jun-05	11	54
	13-Jul-05	ND	ND
Hutton	08-Jul-04	26	23
	18-Jul-04	ND	ND
	27-Jul-04	1	17
	23-Jun-05	6	13
	14-Jul-05	ND	ND
Huston	10-Jul-04	83	76
	16-Jul-04	32	31
	25-Jun-05	ND	ND
	15-Jul-05	28	20

ND = no data due to interference with the radar by rainfall

Table 15. Marbled Murrelet densities (numbers of birds per hectare of suitable nesting habitat) in watersheds of Gwaii Haanas, 2004 and 2005 (from Harfenist and Cober 2005). There were no data for Windy Bay.

Location	Number of Marbled Murrelets[1]	L.U.P. Suitable Nesting Habitat[2] (ha)	L.U.P. Density[3] (birds per ha)
Bigsby	145.5	464.5	0.313
Klunkwoi	98	812.9	0.121
Hutton	92	326.3	0.282
Huston	127	1,028.8	0.123

1 average maximum radar counts for 2004 and 2005
2 area of Marbled Murrelet nesting habitat suitability classes 1, 2 and 3 as mapped in the Haida Gwaii/Queen Charlotte Islands Land Use Plan
3 number of Marbled Murrelets per hectare of suitable nesting habitat calculated using the Haida Gwaii/Queen Charlotte Islands Land Use Plan maps

Note that there is no evidence coast-wide that MAMU “pack” into remaining habitat when their nesting habitat is removed (i.e., logged). In other words, the number of nesting birds is directly related to the amount of suitable habitat.

At this time, there is no density threshold for nesting MAMU deemed “good” by COSEWIC, Canadian Wildlife Service or any other science-based organization. Preliminary data indicate that nesting densities along the British Columbia coast tend to cluster into two groups with Haida Gwaii and the west coast Vancouver Island supporting higher densities of MAMU (number of birds per ha of suitable habitat) than the north mainland, central, eastern Vancouver Island and southern mainland (“sunshine”) coasts.

2.1.7. Measure 7 – Human Footprint: Past

Monitoring Question

With the exception of the Ikeda/Jedway mine area, monitoring of the footprint of pre-establishment activities is not recommended. Sites receive relatively few impacts from visitors and they are restoring naturally. However, the scale and seriousness of the environmental degradation at the old Jedway / Ikeda mine site (Golder Associates 2003) warrants special attention although this “contaminated” site is in a “Mineral Exclusion” area within Gwaii Haanas (SoPR Section 3.1.6.).

Context

The human footprint from activities predating Gwaii Haanas’ establishment have had effects on the landscape (SoPR Section 3.1.6.). These disturbances damage wildlife habitat, fragment the landscape and act as vectors for the introduction and spread of non-native species. Cleared forest has led to a high incidence of slope failure, and mining has left numerous small and one large long-lasting footprint. Other activities, occupying relatively small areas, include canneries/salteries, a whaling station, trails, and a lighthouse/weather station (SoPR Section 6.2).

The largest footprint is that from logging. In Gwaii Haanas 9,017 ha have been logged, representing 6.1% of the total land base (Figure 11). However, logging has not occurred since 1987 and forests are regenerating. Although the areas first logged, about 100 years ago, have recovered visually, it will take centuries to complete the succession to old-growth forest. Further, browsing by introduced deer (Section 2.1.3.) means that forest regeneration can only occur to some deer-influenced state. Lyell Island, which was the most heavily logged area (from the 1970s to 1987), has undergone >$2M in landscape restoration (planted trees, decommissioned roads/culverts). Since 2004, Lyell Island has also had a multi-year stream restoration plan funded by Parks Canada and executed in collaboration with the Haida Fisheries Program and the Hecate Strait Stream Keepers (a local NGO). Fish habitat was assessed and various restoration actions taken for salmon rearing in streams of logged watersheds (HFP/HSS 2006).

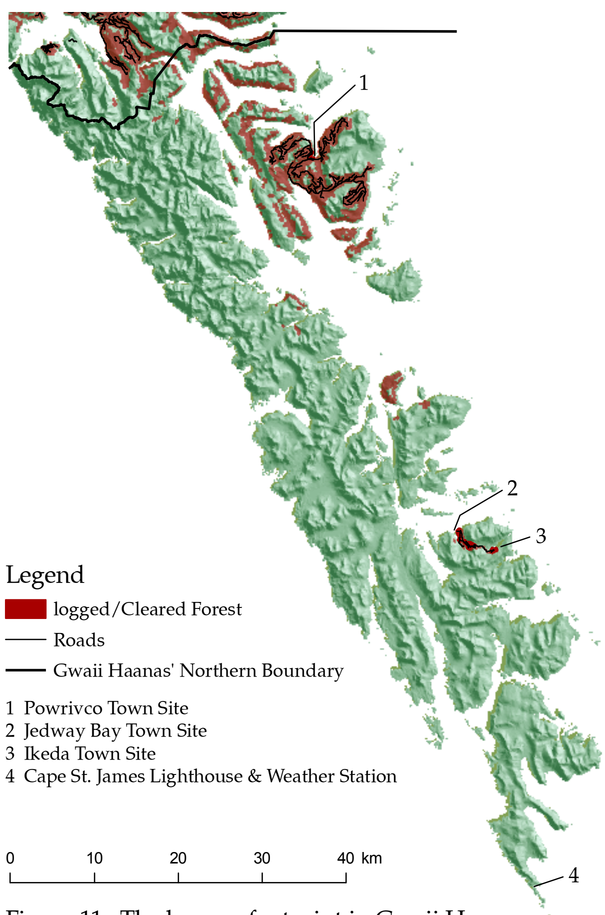


Figure 11. The human footprint in Gwaii Haanas from pre-establishment activities.

Two large producing mines (Jedway/Ikeda) and 10 smaller producing mines have operated in Gwaii Haanas. The smaller mines have left small but long-lasting changes to the landscape such as shafts, trenches and adits. The large mines have left large and permanent changes to the landscape including excavations, unstable tailing slopes, acid rock drainage, and heavy metal accumulations. Some sites have been decommissioned and restored. Further, Powrivco town site and the Cape St James lighthouse/weather station have been decommissioned.

EI metrics and thresholds for these past activities are not under development. Nor will monitoring occur. A possible exception is the old Ikeda/Jedway mine contaminated site (technically outside Gwaii Haanas in a Mineral Exclusion area).

2.1.8. Measure 8 – Human Footprint: Present

Monitoring Question

What is the length, width and condition of trails, and what is the footprint area for structures?

Context

The human footprint from activities after Gwaii Haanas' establishment includes campsites, trails and minor infrastructure such as Warden Operations stations, Watchmen camps, Swan Bay Rediscovery Camp, communications towers, Natural Resources Canada seismic stations, Environment Canada weather stations, survey monuments, Canadian Hydrographic Survey boundary markers and aids to navigation (Figure 12). Their individual and cumulative footprint is small, but they can be a vector for introduced species and they can visually affect visitors' wilderness experiences. The EI metrics and their thresholds have not yet been established – likely to be based on criteria set based on the allowable infrastructure in the management plan.

Results

Although the human footprint is increasing, it is very limited and allowed for in the management plan, such as the redevelopments at Huxley Island Warden Station and SGang Gwaay Watchman camp

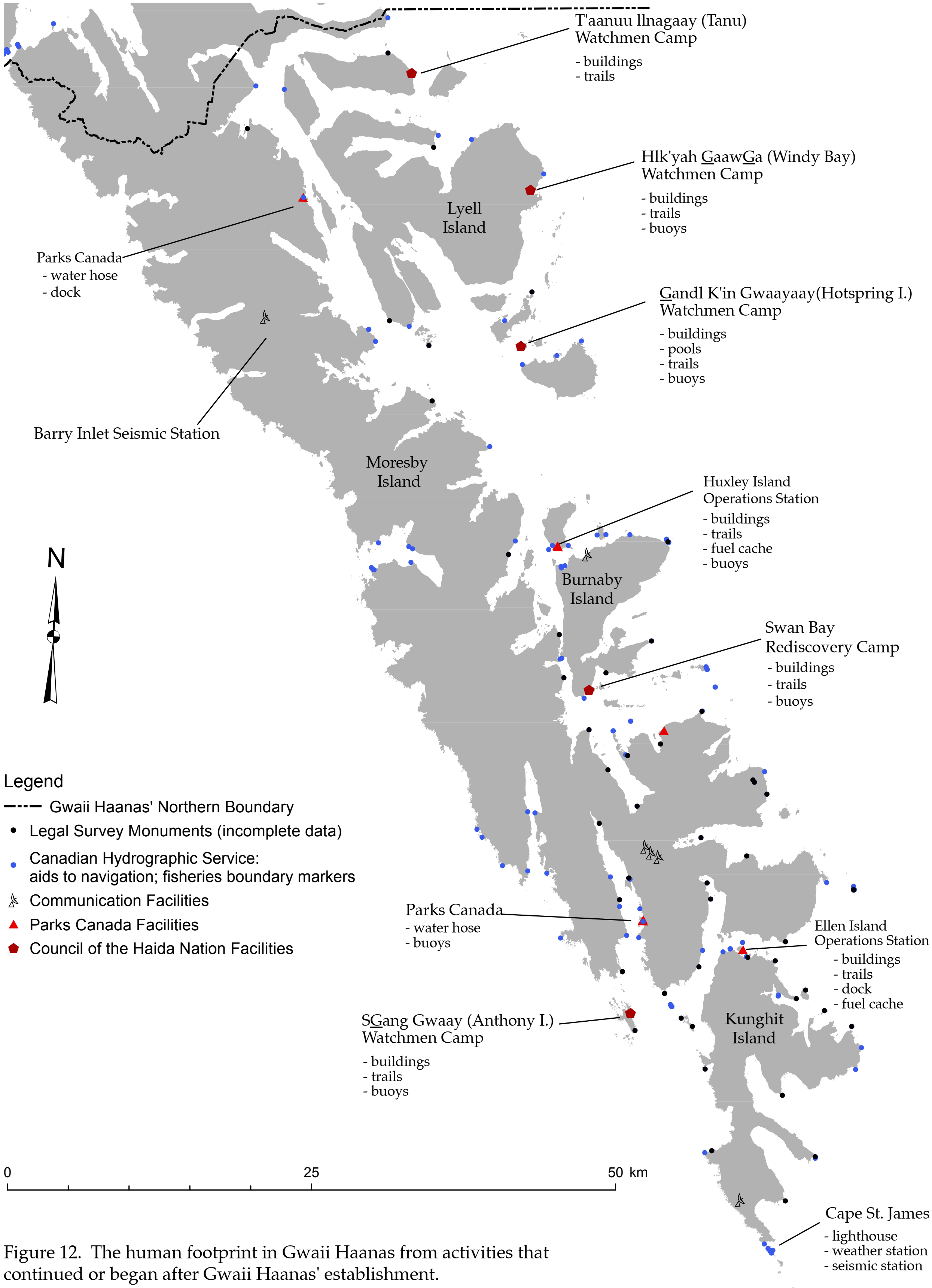


Figure 12. The human footprint in Gwaii Haanas from activities that continued or began after Gwaii Haanas' establishment.

2.2. NON-FORESTED

2.2.1. Measure 1 – Vascular Plants

Monitoring Question
Are there trends in the diversity and cover of the plant community in non-forested (alpine/ sub-alpine) ecosystems in Gwaii Haanas that we would not expect due to random fluctuations?

Context
As an isolated marine archipelago, Haida Gwaii is home to a unique insular biota, with many omissions, regional endemics and major disjunctions (Douglas et al. 1998). Several of the rare and unique species found on Haida Gwaii are restricted to, or most commonly found in, high-elevation alpine and sub-alpine areas (Ogilvie 1994).

The non-forested alpine and sub-alpine zone is very exposed, with patchy vegetation cover separated by outcroppings of bedrock or bare soil. Plant communities are a complex mix of heath/herb meadows, blanket bogs and dwarfed trees (Golumbia 2001).

Methods and Analysis
The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots, including species richness and vegetation cover at various strata. In the summer of 2006, we collected baseline data at plots established in a non-forested sub-alpine area at ~400 to 500 m above sea level. These plots are paired with exclosures designed to assess the impacts of introduced deer (see Section 2.2.3 - Deer) and will be assessed every five years. Trend information is not yet available for this measure and thresholds have not been established.

2.2.2. Measure 2 - Non-native Plants

Monitoring Question
Are cover and richness of the non-native plant community in non-forested (alpine/ sub-alpine) ecosystems increasing?

Context
Of the 741 vascular plant species recorded on Haida Gwaii, 203 are not native to the islands (Cheney et al. 2007) and new species continue to colonize. Most of these species are early successional plants, adapted to habitats that have been disturbed in some way (e.g. industrial sites or shorelines). Very few non-native plants have become established in undisturbed upland areas. Although only a small proportion of non-native species are considered invasive, rapidly expanding and threatening native flora and fauna through competition and habitat alteration, the presence of any non-native plant changes the floristic make-up of the area it colonizes.

Methods and Analysis
The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots, including species richness and vegetation cover for native and non-native species. In the summer of 2006, we collected baseline data at plots established in a non-forested sub-alpine area at ~400 to 500 m above sea level. These plots are paired with exclosures designed to assess the impacts of introduced deer (see Section 2.2.3 - Deer) and will be assessed again in 2008 and every subsequent five years. The 2006 data have not yet been analyzed. The EI metrics for non-native plants have not been finalized and thresholds have not been established.

2.2.3. Measure 3 – Introduced Deer

Monitoring Question
Is the presence of deer altering vegetation communities in non-forested (alpine/ sub-alpine) ecosystems in Gwaii Haanas and are the effects increasing?

Context
The effects of introduced Sitka black-tail deer some ecosystems has been dramatic and well documented (Gaston et al. 2007 a; Section 2.1.3). To date, research has focused primarily on forested areas, with very little attention paid to other regions such as the alpine/sub-alpine. Non-forested, high elevation areas of Gwaii Haanas are characterized by heavy precipitation, deep, wet and long lasting snow pack, and exposure to high winds and cool, humid air. The typical vegetation, composed of stunted trees and heaths, is interspersed with exposed rock. Conditions in these areas are likely a deterrent to year-round occupation by deer, given the increased energetic costs of travelling and thermoregulation, and the reduced availability of forage.

Methods and Analysis
In 2003, deer pellet group transects were run at 10 alpine sites in Gwaii Haanas (Parker and Burles 2003). Results indicated that despite the often-inhospitable conditions, deer effects in alpine areas may be nearly as severe as in forested areas. The alpine study found a median estimate of 233 pellet groups/ha, not much less than a was found in old-growth forest areas of Haida

Gwaii using similar methods (median estimate of 300 pellet groups/ha) (Engelstoft 2001).

EI would be assessed as good (green) for this measure when no effects of deer are recorded. Although a second threshold (going from fair to poor: yellow-red) and the EI metrics for this measure have not yet been determined, we assess the status to be poor (red), based on evidence that deer browsing pressure is similar in the non-forested alpine areas to that found in forested areas. There is an abundance of research demonstrating the profound and far-reaching effects of deer in forested ecosystems.

As part of the Gwaii Haanas Deer Management Strategy (Johnston 2006 a), four 15 m x 15 m exclosures were built in a non-forested sub-alpine area at ~400 to 500 m above sea level in 2007. In the summer of 2006, baseline vegetation data were collected at plots within each exclosure and paired non-exclosure site. These vegetation plots will be monitored in 2008 (two years after establishment) and every subsequent five years, to assess the impact of introduced deer on plant species composition and cover at high elevation, non-forested sites. No trend data are yet available.

The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots. We will be applying this methodology to the vegetation plots (exclosures and controls) in Gwaii Haanas to ensure consistency and sufficient statistical power to detect changes. As a bioregion, we are also assessing protocols to monitor the relative abundance of deer using pellet counts.

2.2.4. Measure 4 - Extent of Alpine Zone

Monitoring Question

What is the aerial extent of this non-forested, high-elevation zone compared to the past and is the zone being fragmented?

Context

The non-forested (alpine/sub-alpine) tundra of Gwaii Haanas contains numerous rare and threatened plants, including at least one endemic species. A historic baseline is in hand from aerial photography of 1933 to 1937 and 1954 to 1955. The non-forested landscape is under pressure from advancing treeline as even a small advance could fragment this zone that is restricted to long, thin polygons (SoPR Section 3.2.2.). As tree growth at higher altitudes is closely linked to climate, advancing treeline due to global warming is a real threat. Browsing by introduced deer is also a concern, but it is unknown what effect this will have on the treeline.

The alpine/sub-alpine landscape has been mapped according to various classification schemes, although to different scales and different standards that cannot be used to discern changes over time. The best current mapping of non-forested area is the British Columbia Ministry of Forests' "Provincial Biogeoclimatic Subzone/Variant Mapping" database at a scale of 1:250,000. According to this data set, 131.6 km2 (8.9%) of Gwaii Haanas is alpine/sub-alpine tundra. More detailed 1:10,000 mapping for Haida Gwaii is planned by the Province within the next few years using the Provincial Vegetation Resource Inventory (VRI) standard.

Methods and Results

Non-forested EI metric(s) and their thresholds will be finalized in 2008 and will likely include area, patch size distribution, patch connectivity and human disturbances. Thresholds would be based on the earliest possible aerial photography baseline from which any measurable decrease would be red. It should also be possible to apply the VRI standard to past data sets, the earliest being aerial photography from the 1930s. This scale should provide the detail necessary for monitoring this landscape.

Currently the status and trend of this measure are both unknown. After a solid baseline is developed (perhaps from the 1930s aerial photographs), the thresholds for EI metrics can be established. Given the time scale of tree growth, aerial photo series from perhaps 2025 and 2050 would be needed to enable measuring change in treeline. Also, Gwaii Haanas should collaborate with the British Columbia Ministry of Environment to ensure that the planned VRI mapping is completed for all of Haida Gwaii.

2.3. LAKE AND WETLAND

2.3.1. Measure 1 – Vascular Plants

Monitoring Question

Are there trends in the diversity and cover of the vascular plant community in wetland ecosystems of Gwaii Haanas that we would not expect due to random fluctuations?

Context

As an isolated marine archipelago, Haida Gwaii is home to a unique insular biota, with many omissions, regional endemics and major disjunctions (Douglas et al. 1998). With a wider diversity of plant species than is found in forested areas, Gwaii Haanas' wetlands are home to many of these unique species (AMB 1994).

Methods and Analysis
The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots, including species richness and vegetation cover at various strata. In the summer of 2007, we will be collecting baseline data at plots established in a wetland area in central Gwaii Haanas. These plots are paired with exclosures designed to assess the effects of introduced deer (see Section 2.3.3. - Deer) and will be assessed every five years. Trend information is not yet available for this measure and thresholds have not been established.

2.3.2. Measure 2 - Non-native Plants

Monitoring Question
Are cover and richness of the non-native plant community in wetland ecosystems increasing?

Context
Of the 741 vascular plant species recorded on Haida Gwaii, 203 are not native to the islands (Cheney et al. 2007) and new species continue to colonize. Most of these species are early successional plants, adapted to habitats that have been disturbed in some way (e.g. industrial sites or shorelines). Very few non-native plants have become established in undisturbed wetland areas. Although only a small proportion of non-native species are considered invasive, rapidly expanding and threatening native flora and fauna through competition and habitat alteration, the presence of any non-native plant changes the floristic make-up of the area it colonizes.

Methods and Analysis
The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots, including species richness and vegetation cover for native and non-native species. In the summer of 2007, we will be collecting baseline data at vegetation plots established in a wetland area in central Gwaii Haanas. These plots are paired with exclosures designed to assess the effects of introduced deer (see Section 2.3.3. - Deer) and will be assessed every five years.

The EI metrics for non-native plants have not been finalized and thresholds have not been established.

2.3.3. Measure 3 – Introduced Deer

Monitoring Question
Is the presence of deer altering vegetation communities in wetland ecosystems in Gwaii Haanas and are the effects increasing?

Context
The effects of Sitka black-tail deer browsing on Haida Gwaii ecosystems has been dramatic and well documented (e.g. Gaston et al. 2007 a; Section 2.1.3 - Deer). To date, research has focused primarily on forests, with very little attention paid to other areas such as wetlands. Anecdotal evidence indicates that deer also frequent and forage in these ecosystems. In Gwaii Haanas, wetland areas contain a wider diversity of plants than are found in forested areas, including many unique species (AMB 1994). These species include several shrubs and broad-leaved herbs, vegetation types particularly sensitive to deer browsing (Pojar et al. 1980; Daufresne and Martin 1997).

Methods and Analysis
As part of the Gwaii Haanas Deer Management Strategy (Johnston 2006 a), four 15 m x 15 m exclosures will be constructed in a wetland area in the 2007-08 fiscal year. Baseline vegetation data will be collected at plots within each exclosure and paired non-exclosure site in the 2007 field season. These vegetation plots will be monitored in 2009 (two years after establishment) and every subsequent five years to assess the impact of introduced deer on plant species composition and cover.

The Pacific Bioregion monitoring group is working on a standardized protocol for monitoring vegetation plots. We will be applying this methodology to the vegetation plots (exclosures and controls) in Gwaii Haanas to ensure consistency and sufficient statistical power to detect changes. As a Bioregion, we are also assessing protocols to monitor the relative abundance of deer using pellet counts. Thresholds have not yet been determined for this measure.

2.3.4. Measure 4 - Western Toad

Monitoring Question
Is there a trend in the number of active western toad breeding sites in Gwaii Haanas?

Context
The western toad (*Bufo boreas*) is the only amphibian species native to Haida Gwaii. In 2002 it was designated a *species of special concern*

by COSEWIC (the Committee on the Status of Endangered Wildlife in Canada) due to widespread and unexplained declines in the southern part of its range in British Columbia. Although it is currently yellow-listed in British Columbia, the western toad is ranked an S4 species, indicating that it is considered to be of conservation concern. Indeed, *B. boreas* is the only IUCN red-listed amphibian species occurring in Canada. While the causes of the species' decline remain largely unknown, they likely include habitat loss and alteration, disease, and introduced exotic predators and competitors (Wind and Dupuis 2002). Within Gwaii Haanas, western toads are protected from habitat loss, but remain vulnerable to the other identified threats. Chytrid fungus, a pathogen that is implicated in amphibian declines globally, has recently been reported in British Columbia. It has not yet reached Haida Gwaii. Two non-native amphibians have been introduced to Haida Gwaii, the Pacific treefrog (*Pseudacris regilla*) and the red-legged frog (*Rana aurora*). These frogs have not yet reached Gwaii Haanas, and the impact, if any, they have on toad populations is unknown. The raccoon (*Procyon lotor*), another non-native species, is an efficient toad predator, particularly at breeding congregations.

Because they are relatively resistant to water loss, western toads forage in a broad range of habitat types. On Haida Gwaii, toads are found throughout both Graham and Moresby Islands from the north to south and east to west. They have been observed in forested and open areas, from sea level to the subalpine (Burles et al. 2004; Johnston 2006 b). In the spring, however, adult toads congregate at a select few lentic sites (ponds, wetlands and shallow margins of lakes or streams) to reproduce. Surveys conducted in Gwaii Haanas have revealed only six toad breeding sites (Johnston 2006 b).

Methods and Analysis

Because of the large natural variation in population size and distribution, the most promising inventory and monitoring method for pond-breeding amphibians is to estimate site occupancy rates over a large geographic scale using presence/absence data, accounting for detection error (Proportion of Area Occupied, PAO, Mackenzie et al. 2002). This involves multiple visits to sites during the appropriate season when evidence of breeding activity (breeding congregations, eggs or tadpoles) is detectable. Unfortunately, documenting statistically significant shifts in PAO through time requires a minimum sample size much larger than the total number of breeding sites known in Gwaii Haanas (~ N = 50). Despite this limitation, we plan to continue monitoring the occupancy of western toad breeding sites in Gwaii Haanas because it is a species at risk threatened by infectious disease, non-native predators and competitors. Biologically meaningful trends in the number of active breeding sites can still be gained by periodically surveying the known and potential breeding sites accessible along the east coast of Gwaii Haanas (N = 9). To determine our survey frequency and thresholds, we must first determine the annual variability in the breeding activity at these sites. A three-year baseline of annual surveys will be conducted from 2006-2008. A long-term monitoring protocol will then be developed.

Visual encounter surveys were first conducted in the spring of 2006. Of 16 potential breeding sites surveyed, evidence of breeding (breeding congregations, eggs or tadpoles) was recorded at five sites (Figure 13). With no baseline data to compare this to, we are unable to determine the status or trend of this measure. We found evidence of raccoon predation at three of the five breeding sites. Although the level of predation observed in 2006 was quite low (1 to 12 toads killed per site), much higher predation (several hundred toads) was documented at one of these breeding sites in 1992 (Johnston 2006 b).

2.3.5. Measure 5 – Non-native Amphibians

Monitoring Question

Has the distribution of frogs expanded to or within Gwaii Haanas?

Context

The western toad is the only amphibian species native to Haida Gwaii. In the 1930s and again in the early 1960s, Pacific treefrogs were introduced and have now become widespread throughout Graham Island and the northern portion of Moresby Island. They have not yet reached Gwaii Haanas. In 2002, a second frog species, the red-legged frog, was identified at several locations near Port Clements and Juskatla on Graham Island. They have not been reported in the southern portion of the archipelago. The effects these frogs have on native ecosystems and toad populations are unknown, but they are predators and competitors of native species. For example, frogs and toads require similar breeding habitat, their tadpoles share the same vegetarian diet, as do numerous aquatic invertebrates, adult frogs likely compete with toads and other insectivores either directly or

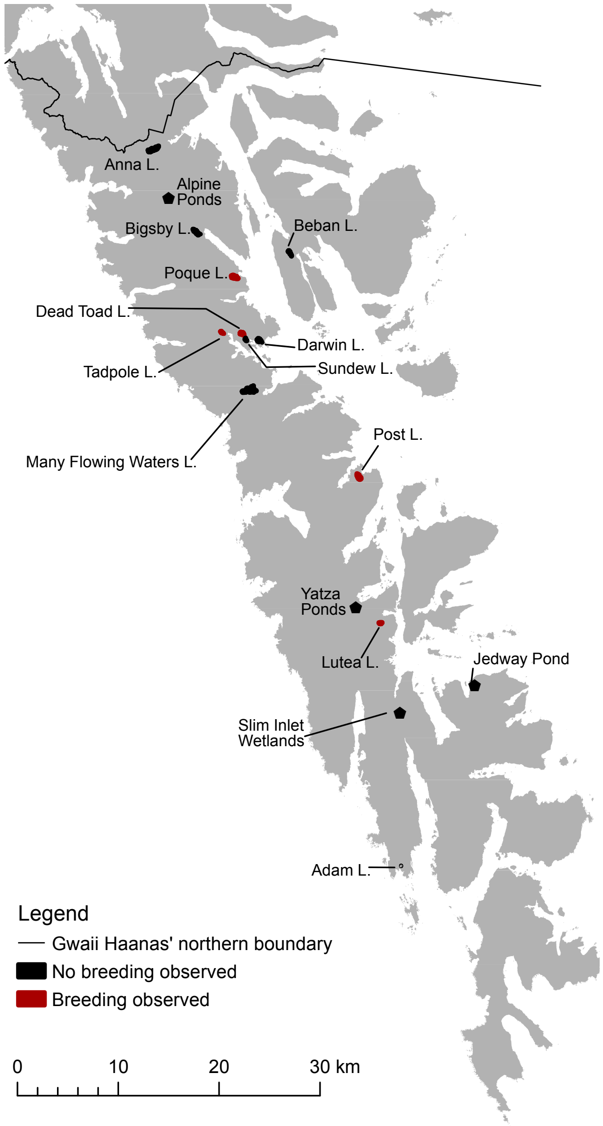


Figure 13. Location of Western Toad breeding sites within Gwaii Haanas. Sites surveyed in 2005-2006 with no evidence of toad breeding are also indicated.

indirectly, and adult treefrogs may be predators of toad eggs (Burles et al. 2004; Johnston 2006 b).

Methods and Analysis
Because frogs are most likely to colonize Gwaii Haanas from the north, our amphibian monitoring protocol will assess if frogs are present within Gwaii Haanas' northern watersheds (Anna Inlet and McEchran Cove) in addition to the potential western toad breeding sites throughout Gwaii Haanas (N = 9). Methods include visual encounter surveys and acoustic surveys.

Thresholds for this metric have been set according to the spread of frogs into and within Gwaii Haanas. The status is assesses as good (green) when no frogs are detected within Gwaii Haanas. If frogs become established within the boundaries, the status becomes fair (yellow). If frogs spread to the extent that they become established at the known toad breeding sites, the status becomes poor (red). As frogs have not yet become established within Gwaii Haanas, the current status of non-native amphibians is good (green). To date, red-legged frogs have only been found on Graham Island, but treefrogs are now within ~10 km of Gwaii Haanas' northern boundary on Moresby Island. With no control measures in place, it is likely only a matter of time before treefrogs spread into Gwaii Haanas. This measure is therefore given a declining trend.

2.3.6. Measure 6 – Extent of Lakes and Wetlands

Monitoring Question
What changes are there of the aerial extent of lakes and wetlands in Gwaii Haanas since the initial assessment?

Context
The extent of lakes and wetlands in Gwaii Haanas is shown in Figure 14. Gwaii Haanas has 118 lakes ranging from 1 to 228 ha in area plus hundreds of open-water areas <1 ha (Krishka 1997). Wetlands are found at all elevations from alpine to estuarine. Therefore, these wetlands range from freshwater wetlands (e.g., ponds, marshes, bogs, fens) to marine-influenced estuaries that transition from freshwater wetlands to tidally-influenced saltmarshes (Sloan 2006).

Methods and Results
EI metric(s) and their thresholds for lakes and wetlands are not yet developed. Metrics could include area, distribution and some assessment of wetland connectivity and human disturbance. Thresholds would be based on the earliest possible baseline from which any measurable decrease would be poor.

2.4. STREAM, RIVER AND ESTUARY

2.4.1. Measure 1 – Salmon Escapement

Monitoring Question
What are the trends in estimated annual numbers of spawning salmon species (coho, pink, chum) in "key indicator" streams draining Gwaii Haanas?

Context
Salmon (*Oncorhynchus* spp.) are enormously important to Haida Gwaii's future ecologically,

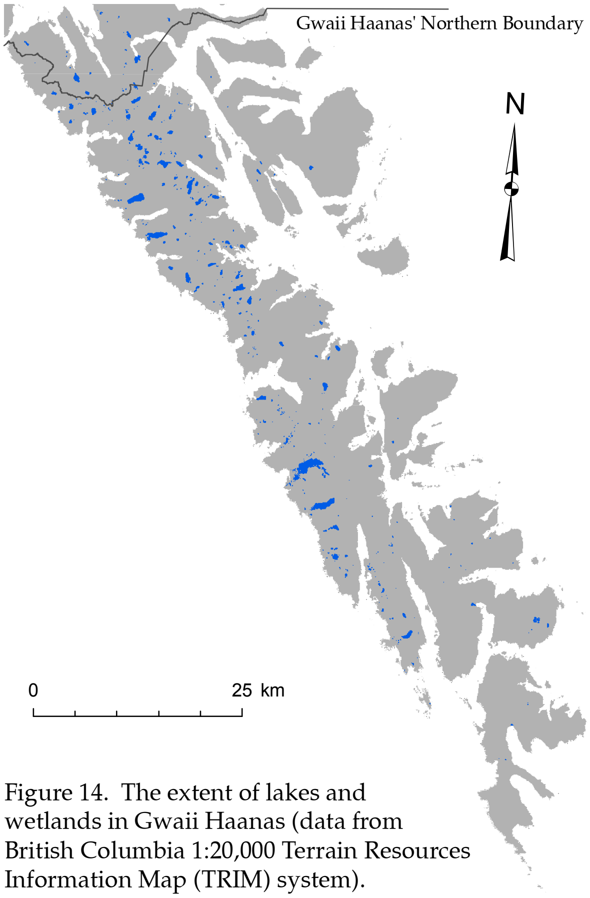


Figure 14. The extent of lakes and wetlands in Gwaii Haanas (data from British Columbia 1:20,000 Terrain Resources Information Map (TRIM) system).

culturally and economically. Among the salmonid species of Haida Gwaii (Table 16), the five main species fished are sockeye, coho, pink, chum and chinook. In addition, steelhead (anadromous rainbow trout) is a prominent sport fish in larger rivers. Most salmon are anadromous - adults leave the ocean and enter river systems to spawn (then die), and their young rear for different periods in those freshwater systems. The young of some species then grow further in nearshore waters such as estuaries before going to sea, while other species go directly to sea. All species spend their adult lives in the open North Pacific before returning to spawn in their natal watersheds.

A critical number for salmon management is "escapement," or the number of adult salmon that escape from their many predators (including humans) to spawn. Escapement numbers are the basis of salmon stock assessment and, therefore, key to management, as stated by Riddell (2004): "Given the limited quantitative information available for most species and areas in the central and northern regions, the historical escapement surveys and data are the core base for any future assessments." Along with Pacific herring spawn occurrence, salmon escapement

Table 16. Some resident freshwater and all anadromous salmonid species of Haida Gwaii (from Sloan 2006). Note that there are multiple life histories (full-time freshwater residence and anadromous) for some salmonids.

	Species
Resident	Dolly Varden char[1] (*Salvelinus malma*)
	Cutthroat trout[2] (*Oncorhynchus clarkii*)
Anadromous	Steelhead[3] (*O. mykiss*)
	Sockeye salmon[4] (*O. nerka*)
	Coho salmon (*O. kisutch*)
	Pink salmon (*O. gorbuscha*)
	Chum salmon (*O. keta*)
	Chinook salmon (*O. tschawytscha*)

1 Dolly Varden is the most common year-round resident lake and stream salmonid, but some populations are anadromous (Krishka 1997)
2 can occur also as anadromous Cutthroat Trout
3 can occur also as permanent lake- and stream-resident Rainbow Trout
4 can occur also as permanent lake-resident Kokanee

data are the oldest annual coastal time series (starting in the 1930s) for Haida Gwaii. Salmon species do more to link marine and terrestrial ecosystems than any other marine group.

There are five salmon fisheries, all with their own ecological needs and political constituencies around Haida Gwaii, as follows: (1) Haida food-social-ceremonial, (2) commercial at-sea "interception" by net (3) commercial at-sea "interception" by hook and line ("troll"), (4) commercial inshore "terminal" by net (seine or gill) and (5) recreational (sport) hook and line (Sloan 2006).

Commercial salmon fisheries and processing date from the early 20th century around Haida Gwaii (Sloan 2006). In recent years, salmon fisheries have declined coast-wide, including Haida Gwaii, with a greatly downsized commercial salmon fleet. Indeed, the coast-wide value of wild salmon landings has fallen ~80% over the past decade. The decline of regional salmon stocks and fishery focus local public concerns over well-being of marine ecosystems, prospects for sustainability and coastal community economies.

Local salmon net fisheries management decisions are documented annually by Fisheries and Oceans Canada's (DFO's) on-island Resource Management Coordinator in the Record of Management Strategies (RMS). These began in 1985, and, since 1994, the same individual has completed them – providing a historical series on local salmon and herring management decision-making whose continuity is unmatched along the north coast. North coast salmon troll fisheries around Haida Gwaii have been documented by DFO out of Prince Rupert since 1993. Troll landings are of intercepted chinook and coho from mostly non-Haida Gwaii stocks.

Concerning Gwaii Haanas only, commercial net fisheries supported by "surpluses" (numbers of returning salmon in excess of individual watersheds' escapement needs for stock maintenance) have been for chum and pink. These terminal net fisheries occur inshore and take incoming spawners (i.e., native salmon) in estuaries or inlets. Pinks have large runs only in even years, so pink takes are even-year only. Until 1981 (the last year before the selective terminal net fishing era), a seine net fishery occurred in larger inlets (such as Skincuttle Inlet or Juan Perez Sound) for spawners incoming to east coast of Moresby Island watersheds in August to September (V. Fradette, DFO, personal communication). These were relatively unselective, mixed-stock fisheries on northward-migrating fish.

The directed terminal salmon takes within Gwaii Haanas from 1985 to 2006 are listed in Table 17. Until the 1990s, there were appreciable chum and pink takes, particularly from the Darwin Sound area. Note that the last commercial take from the west coast of Gwaii Haanas was 1985. Chum and coho salmon have not recovered to fishable levels since the early 1990s and pink surpluses only reemerged in 2004. The small coho takes are always incidental by-catch in this net fishery. In recent years, there have been indications of recovering chum and pink runs into Gwaii Haanas (V. Fradette, DFO, Queen Charlotte, personal communication).

Just north of Gwaii Haanas' east coast there have been regular takes from the "enhanced terminal chum" surpluses entering Cumshewa Inlet, and to a lesser extent runs into Selwyn Inlet systems. The archipelago's largest hatchery (run by the Haida Fisheries Program since 1998) is in Pallant Creek that drains into Gillatt Arm at the head of Cumshewa Inlet. The hatchery produces mostly juvenile chum (over 21 M released annually into Pallant Creek and 4.2 M

Table 17. Gwaii Haanas area commercial salmon takes from terminal net (seine net and/or gillnet) fisheries, based on identified surpluses according to individual watersheds, from the first year (1985) of the Record of Management Strategies to 2006 (courtesy of V. Fradette, DFO, Queen Charlotte and from the individual annual Record of Management Strategies).

	Number of Salmon			
Year	Coho	Pink	Chum	Locations of Reported Catch
1985	228	0	238,722	Darwin Sound, Sedgwick Bay, Flamingo Inlet[1]
1986	154	193,972	14,975	Darwin Sound
1987	0	0	81,267	Atli Inlet, Richardson Inlet, upper Darwin Sound[2]
1988	825	61,493	140,052	Atli Inlet, Darwin Sound
1989	1,083	0	31,163	Darwin Sound
1990	334	49,320	103,981	Darwin Sound, Juan Perez Sound[3]
1991	755	0	30,159	Darwin Sound
1992	148	5	5,673	Darwin Sound
1993	0	0	0	
1994	0	0	29,500	Darwin Sound
1995	0	0	0	
1996	0	0	0	
1997	0	0	0	
1998	0	0	0	
1999	0	0	0	
2000	0	0	0	
2001	0	0	0	
2002	0	0	0	
2003	0	0	0	
2004	0	46,500	0	Salmon River estuary, Darwin Sound
2005	0	0	0	
2006	0	35,000	0	Salmon River estuary, Darwin Sound

1 Flamingo Inlet is in Pacific Fishery Management Area (PFMA) 2W - this is the last year that salmon (15.7% of the annual total catch) were taken commercially in PFMA 2W - for all years since, the reported catch comes from PFMA 2E only
2 PFMA sub-area 2-8 (Laskeek Bay south of the Tangil Peninsula and Logan and Richardson Inlets)
3 Juan Perez Sound accounted for 2.2% of the chum and 94.8% of the pink catch

into nearby Mathers Creek) and there are also small cultured runs of coho and chinook. Thus, the Cumshewa Inlet area has been a relatively regular commercial producer in contrast to Gwaii Haanas' stream systems. The closest recent commercial takes to Gwaii Haanas on the west coast come from Tasu Sound watersheds.

The transfer of marine nutrients from escaping salmon into riparian (near-water) forests by their predators, or directly from salmon carcasses, is a key characteristic of temperate coastal forests throughout the North Pacific, and the leading topic in the field of fish-forestry interactions and management. An important feature of salmon in the EI of coastal ecosystems is their homing behavior to their natal stream to spawn, hence providing a relatively stable, cyclical effect to the ecosystem and the foundation of their genetic diversity. Salmon and the riparian habitat (the "riparian fish forest") they affect are central to the land-sea connectivity and for Gwaii Haanas' prospective land-to-sea conservation continuum (Sloan 2006). Further, protecting salmon habitat, understanding ecosystem integrity and protecting genetic and geographic diversity of salmon are key elements of Canada's Wild Pacific Salmon Policy of 2005 led by DFO. Finally, monitoring salmon escapement within the proposed NMCA would be an important technical issue connecting Parks Canada, DFO, the Haida and commercial fishing.

The scale of marine nutrient transfer around Haida Gwaii is enormous as hundreds of thousands of salmon escape into hundreds of watersheds annually, as reviewed by Sloan (2006). Salmon escapements comprise one of the great local aquatic data sets. Since the 1930s, patrols funded by DFO have estimated numbers of spawning salmon annually from local watersheds, within which the dominant species are coho, pink and chum. Counts are made mostly by trained patrollers on charter (some active for many years), DFO staff and, more recently, Haida Fisheries Program patrollers. Counting methods are not standardized and reliability of counts varies with stream conditions at the time (weather, flow rate, extent of tanin staining) and patrollers' experience. Thus, there is marked innate variability in the escapement monitoring conditions encountered each year. This is offset by the reality that there are long-serving patrollers whose experience and expertise provides continuity.

Attempts by DFO to catalogue all local spawning began in the 1970s and by 1990, DFO had cooperated with the province to create massive catalogues ("Stream Information Summaries") for Haida Gwaii. This is one of four spawning data sources used to describe Haida Gwaii's spawning history, as described in Sloan (2006). Each has some escapement data not found in any of the others. Recent DFO budget cuts for stream patrolling in the mid 1980s, late 1990s and a large (about 75%) cut in 2004 have seen a change in emphasis from broad coverage to a focus on streams potentially yielding fishable surpluses. Because so many streams host salmon, DFO uses "index" streams of the reliable and characteristic producers as reference streams for particular coastal areas or species. Further, years of experience from walking streams had led managers to focus on a sub-set of these index streams called "key indicator" streams that are the priority streams to survey in the face of diminishing funding (V. Fradette, DFO, Queen Charlotte, personal communication).

To summarize, a narrower commercial focus has taken over from a broader coastal focus to decrease costs of escapement monitoring. Operationally, this means many fewer streams patrolled and a decrease in patrol intensity of those patrolled (i.e., key indicator streams). The net effect is poorer broad-scale coverage – we are losing that element of the quality of Haida Gwaii's historic escapement time-series - data that are fundamental to long-term salmon management coast-wide (Riddell 2004). Further, declines in escapement to the archipelago's streams generally are well recorded, but poorly understood. Speculation on causes include habitat damage, overfishing, poor at-sea survival related to changing ocean conditions, effects of altered riparian vegetation on spawning stream quality linked to intense browsing by introduced Sitka black-tailed deer, or climate change effects on watersheds.

<u>Results</u>

Gwaii Haanas' generally steep rocky terrain has small watersheds with brief water residence times and very high flows after heavy rains. There are over 1,400 streams from some 780 watersheds, half of which are only 50 to 100 ha in area (Krishka 1997). Escapements of chum, pink and coho are naturally low from these small watersheds, and have declined in recent decades. Sockeye do spawn in perhaps up to four streams in Gwaii Haanas (Sloan 2006), although in such low numbers (the most consistent presence being in the Salmon River) that these are not considered appreciable sockeye spawning populations (V. Fradette, DFO – Queen Charlotte, personal communication).

Annual escapement monitoring began in the Gwaii Haanas area in the 1930s, although data from the nine "key indicator" streams dates from 1947. Overall, counts are known from 73 streams, with a minority yielding counts on a consistent annual basis. The escapement goals for index and key indicator streams of Gwaii Haanas are provided in Table 18. These goals represent a management number, not a biologically-based number. Names, locations and species of all streams with spawning returns are shown in Figure 15. Most monitoring occurs along the east coast and pink and chum are the most important species. Little coho monitoring is now done within Gwaii Haanas, with two key indicator streams only (excluding the current Lyell Island stream restoration project). The spawning returns for the nine key indicator streams associated with Gwaii Haanas from 1947 to 2005 are shown in Figures 16. Note the low numbers of chum and coho since the 1970s.

In summary, the key indicator streams are the most important target for long-term spawning monitoring within Gwaii Haanas. This is the historical database that warrants maintenance into the future as a key land-sea linkage process within Gwaii Haanas. Further, the proposed NMCAR will likely become involved within overall regional salmon stock assessment and fisheries management. The opportunities for the proposed NMCAR will be to promote land-sea conservation connectivity through reference to salmon and to be a salmon escapement reference location for long-term stock monitoring.

2.4.2. Measure 2 - Benthic Invertebrates

Monitoring Question
Are the benthic invertebrate communities found in Gwaii Haanas streams representative of healthy ecosystems?

Context
Benthic invertebrates are large insects, crustaceans, worms, molluscs and related aquatic animals that live along the bottom of freshwater bodies. Many of their life history characteristics make them particularly good indicators of aquatic ecosystem health as they are relatively sedentary and, therefore, representative of their area. They come from a diverse range of trophic positions and responsiveness to changes in water and sediment quality. They are ubiquitous, easy to collect and identify, and their life cycles range

Table 18. "*Index*" and "*key indicator*" streams (in bold) with their estimated "*escapement goals*" used by Fisheries and Oceans Canada for assessment of coho, pink and chum salmon escapement in Gwaii Haanas (courtesy of V. Fradette, DFO, Queen Charlotte and from the annual Record of Management Strategies). "*Key indicator*" streams, established according to decades of local knowledge gathering, are the highest priority to census for the species in **bold** (V. Fradette, DFO). "*Escapement goal*" is a management number representing the desired number of spawners, not a verifiable biologically-based number reflecting, for example, a stream's spawning carrying capacity. The escapement estimates apply to spawners using "*index*" streams that are selected based upon being wild (not humanly-enhanced) and also having at least 10 years of escapement estimates within 1950 to 2002.

	Escapement Goal of Number of Salmon		
Stream Name	Coho	Pink	Chum
PFMA[1] 2 East			
Moore Creek	NG[2]	NG	3,000
Powrivco Creek	NG	NG	5,000
Sandy Creek	NG	NG	4,500
Anna Inlet Creek	NG	3,000	1,500
Crescent Creek[3]	**1,000**	**20,000**	**6,500**
Echo Harbour Creek	NG	**10,000**	NG
Kostan Creek	NG	NG	1,500
Salmon River	750	**25,000**	**25,000**
Alder Island Creek	NG	10,000	5,000
Arrow Creek	250	NG	2,000
Gate Creek	NG	**20,000**	NG
Hutton Head Creek	NG	15,000	5,000
Hutton L/H Creek	NG	NG	3,000
Island Bay L/H Creeks	NG	NG	2,500
Island Bay R/H Creeks	NG	NG	2,000
Marshall Creeks (3)	NG	7,000	3,000
Matheson Inlet L/H Creek	NG	**30,000**	6,000
Matheson Inlet R/H Creek	NG	5,000	3,000
Sedgwick Creek	250	NG	7,000
Skaat Harbour Creek	350	NG	7,000
Windy Bay Creek	500	**70,000**	NG
Bag Harbour Creek	**1,000**	**1,500**	12,000
George Bay Creek	500	1,000	**12,000**
Harriet Harbour Creek	NG	NG	6,000
Huston Inlet Creek	NG	NG	3,000
Jedway Creek	NG	NG	1,500
Slim Inlet Creek	NG	NG	1,500
Surprise Creeks (2)	300	50	8,000
Tangle Creek	NG	NG	4,000
PFMA[1] 2 West			
Flamingo Inlet Creek	200	750	7,000
Goski Bay Creek	500	NG	3,000
Louscoone Inlet Creek	750	5,000	**7,000**

1 Pacific Fishery Management Areas (PFMAs)
2 NG = no escapement goal (number)
3 a minor portion of the Crescent Inlet Creek watershed is inside Gwaii Haanas

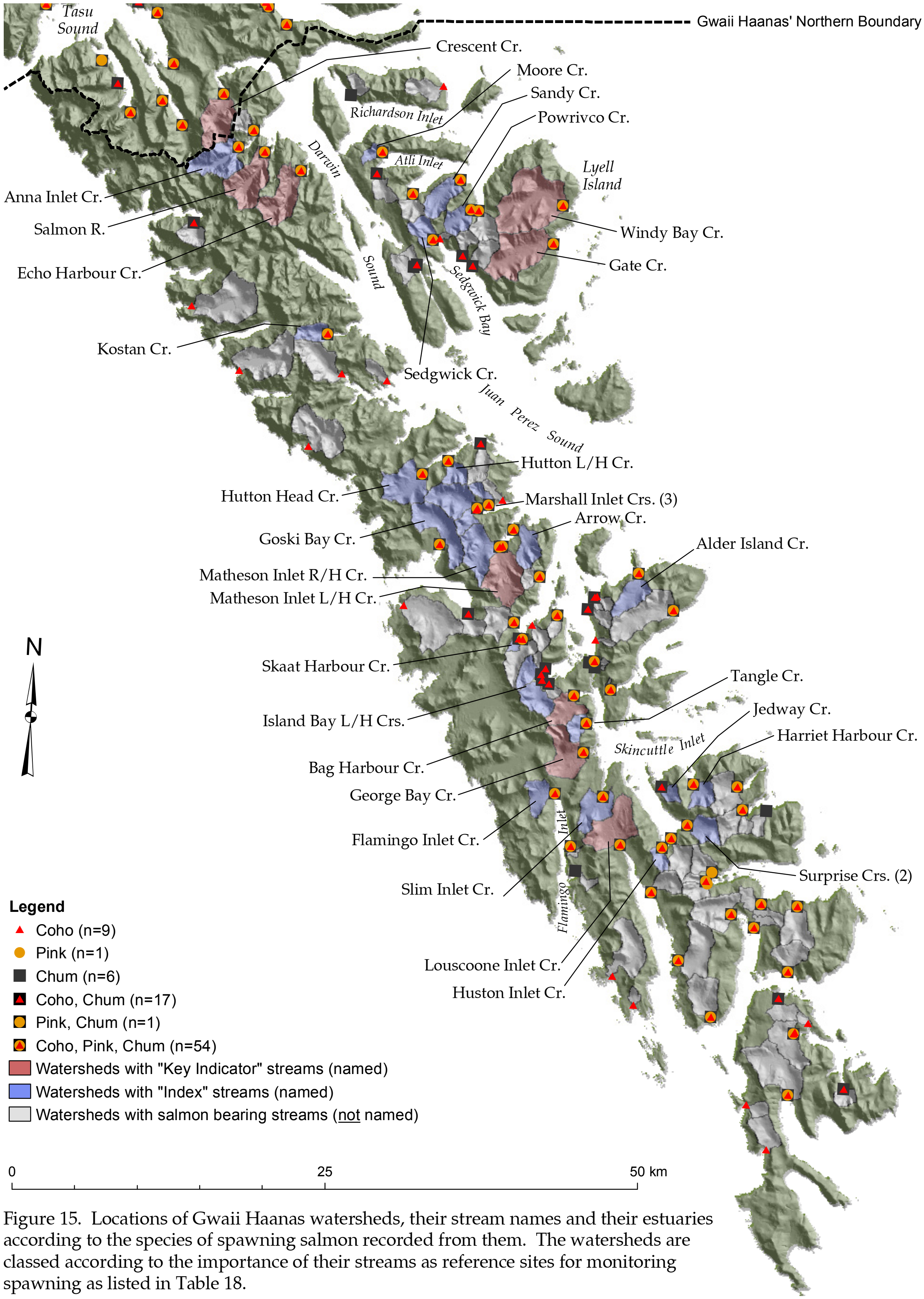


Figure 15. Locations of Gwaii Haanas watersheds, their stream names and their estuaries according to the species of spawning salmon recorded from them. The watersheds are classed according to the importance of their streams as reference sites for monitoring spawning as listed in Table 18.

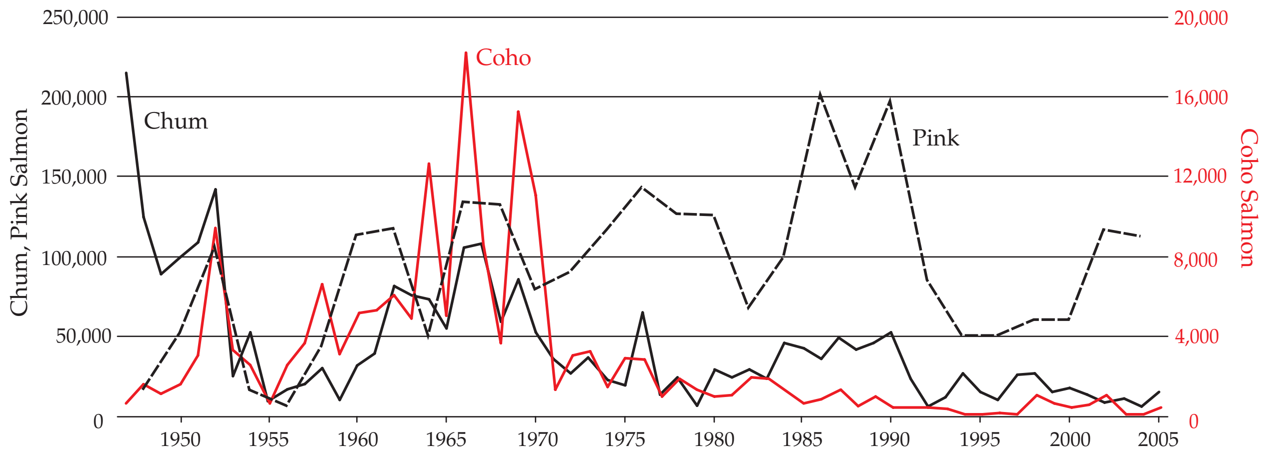


Figure 16. Aggregate numbers of spawning salmon from the nine "key indicator" streams associated with Gwaii Haanas from 1947 to 2005 (data courtesy of DFO, Prince Rupert). Data for coho and chum salmon are annual, but data for pink are even-year only. Note the much smaller scale for numbers of coho.

in length from months to years. They are also important components of the ecosystem, being the primary food source of many fish and essential to breaking down organic matter and nutrient cycling (Rosenberg and Resh 1993; Mackie 2001).

Over the past ten years, Environment Canada's National Water Research Institute (NWRI) has established the Canadian Aquatic Biomonitoring Network (CABIN), a program that uses benthic invertebrate communities to assess the ecological condition of streams and rivers [http://cabin.cciw.ca/intro.asp]. There is a partnership between Environment Canada and Parks Canada in support of the CABIN program. Gwaii Haanas is one of a growing number of national parks nation-wide to adopt this protocol.

Methods and Analysis

In the fall of 2006, we conducted pilot surveys at 16 streams along the east coast of Gwaii Haanas. Streams were selected to capture the diversity of geology, fish presence and catchment types represented throughout Gwaii Haanas. Our sampling methodology follows the CABIN protocol. We collect a standard suite of biological, water quality and habitat information at each study stream, and used the kick-net method to sample benthic invertebrates. Our invertebrate samples were sent to a NABS (North American Benthological Society) certified taxonomist for identification and enumeration. Results of this analysis are not yet available. The EI metric(s) and thresholds for benthic invertebrates have not yet been determined.

2.4.3. Measure 3 - Water Quality

Monitoring Question

Is the water quality at Gwaii Haanas' sample streams deteriorating?

Context

Water quality describes the physical, chemical, and biological characteristics of water and aquatic ecosystems. The utility for monitoring EI is the influence of quality on protection of aquatic life. The chemical and physical characteristics of water influence aquatic biota and the ecosystems in which they reside. They give a snapshot of conditions at the moment of sampling. Biological measures, such as benthic invertebrate community composition, integrate these physical and chemical conditions over time (see previous Section 2.4.2.) (CCME 2006 a).

Thresholds or guidelines have been developed for individual water quality variables for different water uses (e.g. protection of aquatic life or protection of agricultural water uses). When these guidelines are exceeded, there is the potential for adverse effects. The Canadian Council of Ministers of the Environment (CCME) Water Quality Guidelines Task Group sets guidelines at the national level. At the provincial level, the Ministry of Environment sets water quality guidelines. For the most part, these guidelines are in agreement, with one organization occasionally setting slightly more sensitive thresholds.

A water quality index is a tool that enables a large amount of complex water quality data to be translated into a simple overall rating. It is based on a formula that incorporates a number of different water quality variables and can be used to track changes over time. Because the parameters selected for inclusion in the index can be varied, it provides a flexible method for assessing water quality that can be applied across large geographic areas. In 2001, the CCME Water Quality Guidelines Task Group published their guidelines for the calculation of a national water quality index (CCME 2001 a,b). The index combines three different aspects of water quality: the scope (the percentage of water quality variables with observations exceeding guidelines), the frequency (the percentage of total observations exceeding guidelines), and the amplitude (the amount by which observations exceed the guidelines). The CCME water quality index is used to categorize the condition of a water body according to the ranking system outlined in Table 19.

Methods and Analysis

We are collecting water samples at each of the survey streams visited as part of the CABIN program (see previous Section 2.4.2 - Benthic Invertebrates). We plan to use a spatial water quality index as the EI metric for our water quality measure. Because of the remote nature of Gwaii Haanas, we are not able to visit each sample stream with the minimum frequency required to calculate the CCME water quality index (WQI). We are, therefore, currently working with Environment Canada's Water Quality Branch to develop a spatial water quality index that can be applied in Gwaii Haanas. For this index, the frequency aspect of the index (the percentage of total observations exceeding guidelines) may be calculated using samples taken from different sites rather than samples taken repeatedly from the same site. Thresholds will likely be set using the CCME water quality index categories outlined in Table 19. For example, the boundary between the categories of 'excellent' and 'good' would form the green-yellow threshold (at WQI = 95) and the boundary between the categories of 'good' and 'fair' would form the yellow-red threshold (at WQI = 80).

Table 19. Standardized water quality categories based on the Canadian Council of Ministers of the Environment (CCME) water quality index (WQI) (modified from CCME 2001 b).

Category	CCME WQI values	Description
Excellent	95-100	Water quality is protected with a virtual absence of threat or impairment; conditions very close to natural or pristine levels
Good	80-94	Water quality is protected with only a minor degree of threat or impairment; conditions rarely depart from natural or desirable levels
Fair	65-79	Water quality is usually protected but occasionally threatened or impaired; conditions sometimes depart from natural or desirable levels
Marginal	45-64	Water quality is frequently threatened or impaired; conditions often depart from natural or desirable levels
Poor	0-44	Water quality is almost always threatened or impaired; conditions usually depart from natural or desirable levels

In the fall of 2006, we conducted a pilot survey visiting 16 streams along the east coast of Gwaii Haanas. Water samples were sent to Environment Canada's Pacific Environmental Science Centre (PESC) and analyzed for 20 physical and chemical parameters as part of a partnership between Environment Canada and Parks Canada that was created in support of the CABIN program. Of the 20 variables analyzed from our samples, only four (pH, alkalinity, calcium and nitrogen) have Canadian or BC guidelines for the protection of aquatic life (CCME 2006 b; Nagpal et al. 2006). We are working with Environment Canada's Water Quality Branch to select additional priority variable for analysis in future surveys. These will likely include metals and additional physical variables such as temperature and dissolved oxygen.

In 2006, all streams fell within the guidelines for nitrogen (nitrates and nitrites), but five streams (31%) fell below the minimum alkalinity guideline, and seven streams (44%) fell below the minimum guidelines for both pH and calcium. Natural fresh waters have a pH ranging from 4 to 10. Coastal streams, however, commonly have pH values of 5.5 to 6.5, putting them outside the CCME guidelines of 6.5 to 9. All streams in Gwaii Haanas fell well outside the level where pH has lethal effects on aquatic life (below pH 4.5 and above pH 9.5). The alkalinity of a water body determines the water's ability to neutralize acids. Waters with low alkalinity have little capacity to buffer acidic inputs and are susceptible to acid deposition. Low alkalinity, however, is not uncommon in coastal areas, where alkalinity typically ranges from 0 to 10 mg/L, below the British Columbia minimum guidelines of 20mg/L. Because of the buffering capacity of calcium, streams below the minimum guidelines for calcium (8 mg/L in British Columbia) are also sensitive to acid inputs (BC MoE 2007).

All discrepancies form the guidelines that were recorded in Gwaii Haanas streams in 2006 appear to be natural, caused by the geological makeup of the individual watersheds. Water quality is therefore assessed to be in good (green) condition. The trend is unknown.

Our pilot work revealed that several streams within Gwaii Haanas are particularly susceptible to acid deposition because they have a naturally low pH and poor buffering capacity caused by low alkalinity and calcium. These streams require strict protection from acid inputs.

2.4.4. Measure 4 – Riparian Land Cover

Monitoring Question
What changes are there of the aerial extent of riparian land cover in Gwaii Haanas since the initial assessment?

Context
The riparian is the terrestrial area of transition between land and water. Riparian lands for our purposes here are adjacent to fresh waters where the vegetation and microclimate conditions are influenced by perennial or intermittent water, associated high water tables and wet soils. Riparian forests along the northeast Pacific coast are the most biodiverse on the continent in terms of both structure and species (Sloan 2006). On Haida Gwaii, riparian forests associated with significant salmon populations (Section 2.4.1.) have produced high accumulations of biomass.

Methods and Results
EI metric(s) and their thresholds for riparian land cover are not yet developed. Metrics could include area, distribution and some assessment of human disturbance. Thresholds would be based on the earliest possible baseline from which any measurable decrease would be red.

2.5. SHORELINE

Thinking separately of land and sea is not possible in the long-term for the well-being of Gwaii Haanas. Gwaii Haanas' landscape is profoundly maritime, that is, the sea influences its terrestrial ecosystems all the way to the alpine. Indeed, no point on land in Gwaii Haanas is >5 km from the shoreline. Specifically, future terrestrial management should not be contemplated without acknowledging the critical role of harmonizing coastal and marine values (Sloan 2006).

The legal coastal boundary of Gwaii Haanas is the "ordinary high water mark" stipulated in the federal-provincial *South Moresby Agreement* of 1988. This boundary could be considered the "shoreline" within the "coastal zone" – itself defined as the vegetated fringe landward of the shoreline and going seaward across the intertidal into the shallow subtidal to ~20m depth (Sloan 2006). Gwaii Haanas' coast represents the most complex segment of sea-to-land transition, the greatest levels of current and historical human uses (including visitor experience), and the location of productive, biologically diverse ecosystems.

2.5.1. Measure 1 - Black Oystercatcher

Monitoring Question
Is the number of Black Oystercatcher breeding pairs per km of shoreline in the Laskeek Bay study area changing by more than we would expect due to random fluctuations?

Context
Black Oystercatchers (BLOY - *Haematopus bachmani*) are large-bodied shorebirds completely dependent upon marine shorelines throughout their life cycle. BLOY occur along the North American Pacific coast from the Aleutian Islands to Baja California. Approximately one third of the global population breeds in British Columbia, of which ~37% are found on Haida Gwaii (Harfenist et al. 2002). Throughout their range, breeding sites are unevenly distributed and occur in a variety of shoreline habitats from mixed sand and gravel beaches to exposed rocky headlands. Pairs noisily defend nesting territories where they build camouflaged nests. They forage exclusively on intertidal invertebrates (e.g., chitons, limpets, mussels) and are most commonly found near sheltered areas of high tidal variation that support abundant invertebrate communities. Their populations are regulated by the limited availability of high quality nesting and foraging habitat. Because they are dependent upon a narrow range of ecological conditions throughout their annual cycle, BLOY are highly vulnerable to natural and human disturbances, including habitat alteration, predation of eggs and young by natural and exotic predators, human disturbance-induced nest abandonment and predation, direct mortality and indirect food reduction caused by oil spills, and food resource changes caused by global climate change. Accordingly, BLOY is a keystone species along the north Pacific shoreline and is thought to be a particularly sensitive indicator of the overall health of the rocky intertidal community (Tessler et al. 2006).

In 2006, the BLOY Working Group, an international group created to research and monitor BLOY populations throughout its range, drafted a conservation action strategy for the species. The strategy includes implementing a coordinated BLOY monitoring strategy that integrates monitoring at breeding sites across the range, including the Laskeek Bay region at the northeast edge of Gwaii Haanas and the other two national parks in the Pacific Bioregion (Pacific Rim and Gulf Islands). The standardized monitoring protocol involves boat-based shoreline surveys, focusing on the numbers of breeding pairs at a given site (Tessler et al. 2006).

Laskeek Bay straddles the northern boundary of Gwaii Haanas, forming part of the proposed Gwaii Haanas NMCAR as well as the study area of a local NGO, the Laskeek Bay Conservation Society (LBCS). Because both Gwaii Haanas and the LBCS share an interest in the protection of biodiversity and ecosystem function in this area, we collaborate on monitoring and public education efforts.

Methods and Analysis

The LBCS has been using boat based shoreline surveys to monitor BLOY in their northern Laskeek Bay study site since 1992. Surveys cover ~40.5 km of shoreline, encompassing an average of 34 oystercatcher breeding territories (range = 25 to 41, see Table 20). Starting in 2004, Gwaii Haanas has partnered with the LBCS to expand their surveys further south, covering the southern portion of the Laskeek Bay and northern Juan Perez Sound. The expanded survey covers ~136.8 km of shoreline, encompassing about 60 additional territories (Figure 17). To monitor the number of breeding pairs (or occupied territories) in the study area, a minimum of two shoreline

Table 20. The number of Black Oystercatcher breeding pairs (bp) and the number per km of shoreline recorded in the Laskeek Bay study area (40.5 km of shoreline), 1993 to 2006.

Year	Number of breeding pairs (bp)	bp per km
1993	31	0.765
1994	30	0.741
1995	32	0.79
1996	37	0.914
1997	41	1.012
1998	33	0.815
1999	37	0.914
2000	38	0.938
2001	34	0.84
2002	37	0.914
2003	30	0.741
2004	25	0.617
2005	37	0.914
2006	33	0.815
Mean	33.9	0.838
Standard Deviation	4.2	0.104

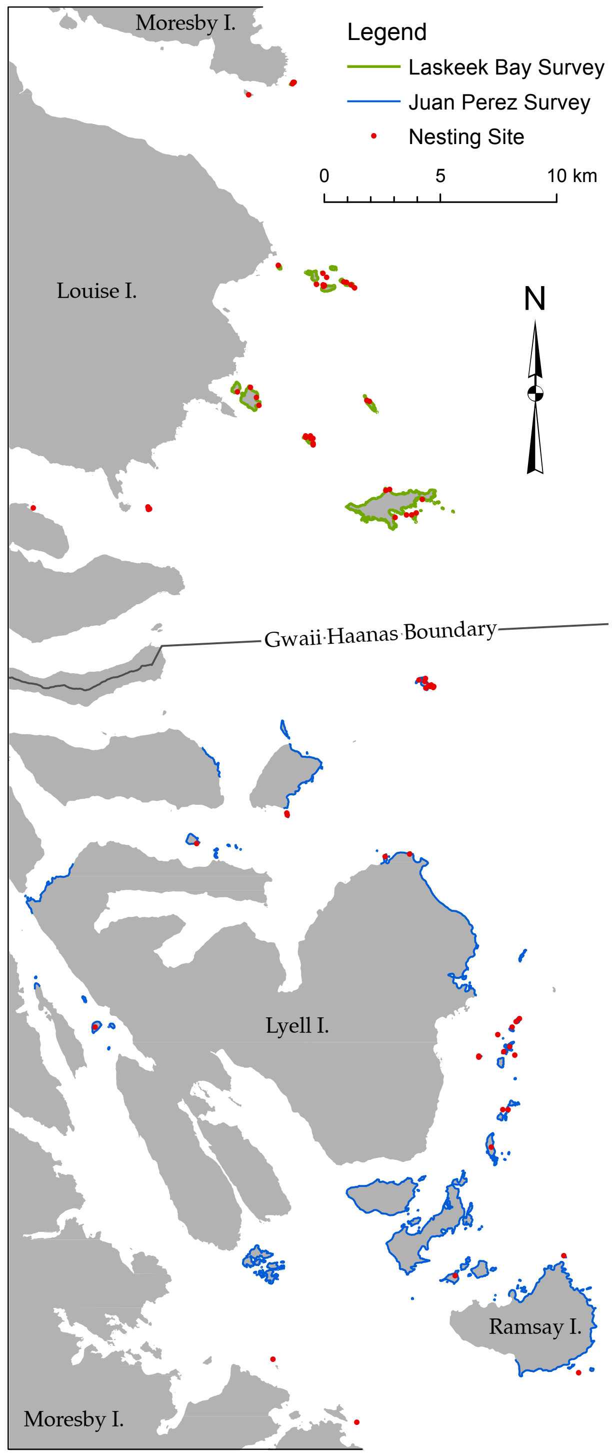


Figure 17. Map of Laskeek Bay and northern Juan Perez Sound area showing the survey routes and locations of known black oystercatcher nests.

surveys is conducted during a field season. An initial survey is conducted in late May, with at least one subsequent survey conducted by early July. A nest site is deemed occupied if a breeding pair (two adult birds) is found at the site during at least one of the surveys.

Results of the LBCS surveys suggest that the number of breeding pairs in the Laskeek Bay study area is stable. There has been no significant trend in the number of breeding pairs per km of shoreline in northern Laskeek Bay between 1993 and 2006 (GLM: N = 14 years, $R2 = 0.003$, $P = 0.85$). We report our results as the number of breeding pairs per km of shoreline (bp/km) to facilitate comparison with other parts of the species' range. Data from 1992 were not included in the analysis because of increased search efficiency following the first year of surveys. Since the number of breeding pairs (bp) has remained relatively stable since 1993, we have tentatively set thresholds at 1 (going from good to fair: green-yellow) and 2 (going from fair to poor: yellow-red) standard deviations below the historic mean. With a mean of 0.84 bp/km in the northern Laskeek Bay between 1993 and 2006, the tentative thresholds are 0.73 (green-yellow) and 0.63 (yellow-red) bp/km (see Table 20).

With 0.82 bp/km recorded in 2006, BLOY are assessed to be in good condition (green). Although the number of bp/km dipped below the green-yellow threshold in 2004, the number rebounded in subsequent years, suggesting that this was likely an anomaly due to specific conditions of that year.

A power analysis using the historic LBCS data (1993-2006) indicates that we can detect a minimum annual change of 2.4% in the number of bp/km (12% change over 5 years) if we survey the northern Laskeek Bay study area every year ($a = 0.05$, $b = 0.20$). If we reduce our sampling effort to every other year, we can still detect an annual change of 3.5% (17.5 % over 5 years). Expanding our sampling area into northern Gwaii Haanas should increase our power to detect change.

The LBCS has also been recording nest contents (chicks and eggs) during their BLOY surveys to estimate productivity. We chose not to use an estimate of productivity (e.g., number of eggs and chicks) as a metric for BLOY because the values as are so variable between years that our ability to detect a trend is very low (Johnston and Lee 2006). In addition, collecting productivity data is more intrusive than conducting occupancy surveys.

2.5.2. Measure 2 - Colony-nesting Seabirds

Monitoring Question

What are the annual trends in the population status of Cassin's Auklet, Ancient Murrelet and Rhinoceros Auklet breeding on reference colonies within Gwaii Haanas?

Context

The seabirds of Haida Gwaii spend most of the year feeding in offshore waters and only spend appreciable time on land to breed. Some 1.5 million seabirds breed colonially on >200 islands, islets and rocks around Haida Gwaii and disperse offshore during the non-breeding season. Thirteen species of seabirds breed on Haida Gwaii and the numbers of breeding pairs per species range from almost 300,000 (Cassin's Auklet - *Ptychoramphus aleuticus*) to less than 10 (Mew Gull – *Larus canus*) (Table 21). Some species are surface-nesters while others are burrow-nesters. Haida Gwaii hosts globally and nationally significant proportions of the breeding populations of five species. The breeding colony distributions of all species, except Marbled Murrelet (see Section 2.1.6.), are reviewed and mapped in Harfenist et al. (2002). As well, there are there are more than 40 other seabird species that feed locally (in season) but breed elsewhere.

Seabirds are exposed to threats in common with other marine birds, but some threats are of greater concern for seabirds. On land, introduced mammal predators damage accessible seabird nesting colonies as reviewed in Harfenist et al. (2002) and Gaston et al. (2007 a). Raccoons (*Procyon lotor* – Section 2.5.7.) and rats (*Rattus* spp.) have seriously affected more than 10 seabird colonies and are suspected of destroying others. Indeed, the Ancient Murrelet (*Synthilboramphus antiquus* - ANMU) was federally designated as a *species of special concern* based largely on the threat posed by introduced predators.

Results

The Canadian Wildlife Service (CWS) is committed to monitoring seabirds coast-wide as described in their Pacific region seabird conservation management program (Hipfner et al. 2002). Their management goal is to "restore seabird populations to desired pre-impact numbers by restoring and protecting habitat." Their first step is to identify population trends through monitoring. To that end, the CWS selected "key BC seabird colonies" in the 1980s among the 503 known breeding sites coast-wide.

Criteria for choosing "key" (reference) colonies include representation within the three

Table 21. The 13 species of seabirds that breed on Haida Gwaii with notes on their estimated breeding populations (data from Sloan 2006).

Species	Estimated breeding population[1] (numbers of pairs)	Notes
Fork-tailed Storm-petrel *(Oceanodroma furcata)*	53,000	~21% of British Columbia's storm-petrel population
Leach's Storm-petrel *(Oceanodroma leucorhoa)*	103,000	
Pelagic Cormorant *(Phalacrocorax pelagicus)*	300 ??	Likely an underestimate as very difficult to account for all nests
Mew Gull *(Larus canus)*	6 ??	Only 6 nests recorded since the early 1990s, very incomplete data
Glaucous-winged Gull *(Larus glaucescens)*	2,800	Likely an underestimate; coast-wide, the population has increased ~4-fold since the 1940s
Common Murre *(Uria aalge inornaata)*	200	Only the Kerouard Islands confirmed as breeding sites
Pigeon Guillemot *(Cepphus columba)*	2,500 ??	Very difficult to estimate breeding population; ~50% of the British Columbia population
Marbled Murrelet *(Brachyrhamphus marmoratus)*	2,900 ??	Very difficult to locate nests in old-growth trees (<20 nests confirmed to date); there may be 5,800 breeding in the region; radar surveys are underway (Harfenist *et al.* 2005)
Ancient Murrelet *(Brachyrhamphus brevirostris)*	256,000	~50% of the global population; the only breeding location in Canada
Cassin's Auklet *(Ptychoramphus aleuticus)*	297,000	~18% of the global population; the most abuntant breeding seabird around Haida Gwaii
Rhinoceros Auklet *(Cerorhinca monocerata)*	23,900	~4% of the global population
Horned Puffin *(Fratercula corniculata)*	16 ??	No confirmed breeding sites (3 probable, 2 suspected)
Tufted Puffin *(Fratercula cirrhata)*	560	14 confirmed or suspected breeding colonies

1 these are rough estimates only and those with (??) are particularly speculative, for example, the number for Pelagic Cormorant could be ou an order of magnitude as just one colony at Tow Hill has ~250 pairs (J. Broadhead, personal communication)

oceanographic domains off the British Columbia coast, colonies of mixed sizes and colonies with historical information on population sizes. Permanent plots along with mapped colony boundaries provide the optimal monitoring regime for ground-nesting seabirds (Hipfner et al. 2002). Permanent monitoring plots are used to count the number of active burrow nests to enable extrapolation of colony performance.

The CWS surveys around Haida Gwaii in the 1980s were part of a coast-wide inventory of seabird colonies, the goal of which was to establish baseline estimates of breeding seabird populations using standardized survey techniques to begin monitoring and enable between-area comparisons of those populations. The main focus for Haida Gwaii was three species of burrow-nesters. Therefore, surveys were generally conducted in April and May to coincide with the Cassin's Auklet (CAAU), Rhinoceros Auklet (*Cerorhinca monocerata* - RHAU) and ANMU breeding seasons.

The technique used to generate population size estimates of these large colonies of burrow-nesters was sampling quadrats along transects ("transect surveys"). A series of "permanent" monitoring plots were established on selected islands to enable detecting colony trends without having to resurvey the entire colony with the line transect/quadrat method. These monitoring plots were not intended to provide a population estimate, only trends. Subsequent to the baseline inventories of the 1980s, surveys of the monitoring plots were conducted from time to time, and occasional complete surveys of some of the island colonies also done.

The history of seabird monitoring by CWS in the Gwaii Haanas area is summarized in Table 22 and the monitoring locations are shown in Figure 18. Monitoring began in 1982 and monitoring plots, for the three burrow-nesting species, began at "key" sites in 1984 and 1985. There was a little CWS work prior to 1982. Population surveys and reproductive studies on CAAU and ANMU on Frederick Island occurred in 1980 and 1981, and on Langara Island in 1981.

Prior to the CWS involvement in seabird work around Haida Gwaii, the British Columbia Provincial Museum conducted explorations and surveys in 1971; and a coastal wide survey was conducted in 1977 – 1978, the results of which are published in map form (Campbell and Garrioch 1979). There were other visits to some of the islands by museum crews through the 1970s and sporadically in earlier years (Drent and Guiguet 1961; Summers 1974).

Within Gwaii Haanas, the CWS has monitored three species at five "key" colony sites since 1984 (Table 23). Since 2002, the CWS has tried to maintain a 5-year cycle of monitoring at each site to improve the utility of the data to reliably demonstrate population trends. As well, there are two "key" monitoring sites nearby – East Limestone and Reef Islands in Laskeek Bay (both for ANMU).

Burrow counts from the sites are provided in three CWS reports (Hipfner 2002; Lemon 2003, 2005). Analyses are complete for all islands except SGang Gwaay (in process).

Listed in Table 24 are the population estimates, suspected trends and desired population sizes in Gwaii Haanas' monitored colonies. A trend for RHAU at SGaang Gwaay is not available due to insufficient data. CAAU is decreasing at Rankine Island, but stable at both East Copper and Ramsay Islands. ANMU are either stable or increasing. All colonies are close to their "desired" population sizes. Trends at the nearby monitored colonies of East Limestone and Reef Islands are

Table 22. History of seabird monitoring by the Canadian Wildlife Service at all locations in the Gwaii Haanas area, 1982 to 2006 (data courtesy of M. Lemon, CWS, Delta, BC).

Year	Monitoring Location(s)	Notes on Monitoring
1982	Lyell Island	Seabird population transect surveys
1983	Skedans Islands south to Lost Islands	Seabird population transect surveys and total counts from small islands
1984	Islands in northern Juan Perez Sound and east side of Lyell Island; Ramsay Island; Rankine Island (west)	Seabird population transect surveys and total counts from small islands (Juan Perez) and ANMU[1] and CAAU[1] *monitoring plots established* (Ramsay and Rankine west)
1985	Islands in southern Juan Perez Sound south to Skincuttle Inlet; islands east of Houston Stewart Channel; SGang Gwaay; islands off west side of Kunghit Island including western mouth of Houston Stewart Channel	Seabird population transect surveys and total counts from small islands
	SGaang Gwaay	RHAU[1] *monitoring plots established*
	George Island; East Copper Island	ANMU (George) and CAAU (East Copper) *monitoring plots established*
1986	Kerouard Islands including St. James Island; Kunghit Island and nearby islets	Seabird population transect surveys and total counts from small islands
	Broad survey of offshore islets around Haida Gwaii	Counts of surface-nesting seabirds
1989	East Limestone Island	Seabird population transect surveys
1991	George Island; East Copper Island	ANMU (George) and CAAU (East Copper) monitoring plots
1992	Dodge Pt., Lyell Island; Ramsay Island	Seabird population transect surveys (Lyell) and ANMU and CAAU monitoring plots (Ramsay)
1993	Kunghit Island and nearby islets	Seabird population transect surveys
1995	East Limestone Island	Seabird population transect surveys
1996	George Island	Seabird population transect surveys and ANMU monitoring plots
2000	Rankine Island (west)	Seabird population transect surveys and ANMU and CAAU monitoring plots
2002	Ramsay Island	ANMU and CAAU monitoring plots
2003	East Copper Island; George Island	Seabird population transect surveys (Copper) and ANMU (George) and CAAU (East Copper) monitoring plots
2004	Kerouard, St. James and Kunghit Islands	Seabird population transect surveys
2005	Rankine Island (west); islets from south end of Kunghit Island (not St. James Island) to Juan Perez Sound	ANMU and CAAU monitoring plots (Rankine west) and BLOY and GLGU surveys (south end)
2006	SGang Gwaay	RHAU monitoring plots

1 RHAU = Rhinoceros Auklet / CAAU = Cassin's Auklet / ANMU = Ancient Murrelet / BLOY = Black Oystercatcher / GLGU = Glaucous-winged Gull

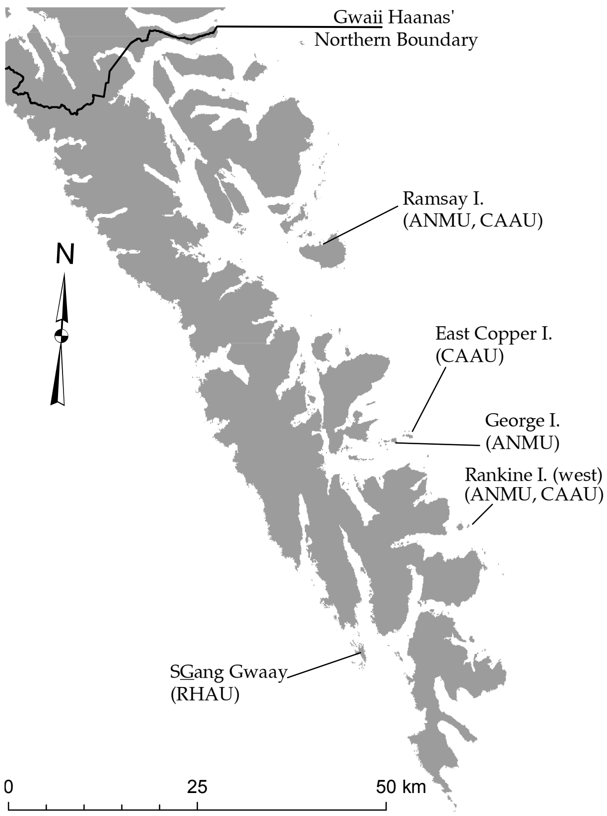


Figure 18. Monitoring locations for breeding seabird colonies of three species of burrow-nesting seabirds established by the Canadian Wildlife Service around Gwaii Haanas. ANMU = Ancient Murrelet, CAAU = Cassin's Auklet, RHAU = Rhinoceros Auklet.

declining and increasing respectively for ANMU (Hipfner et al. 2002). Overall, the status for these species is "good" and the trend "stable."

2.5.3. Measure 3 - Peale's Peregrine Falcon

Monitoring Question

What is the trend, monitored at 5-year intervals, in the number of occupied Peale's Peregrine Falcon territories with eyries (nests) around Haida Gwaii?

Context

Peale's Peregrine Falcon (*Falco peregrinus pealei*), the sub-species of Peregrine Falcon on Haida Gwaii, is a predator mostly of colony-nesting seabirds and this species' local life history is reviewed by Harfenist et al. (2002). This sub-species' range is from coastal areas of the Aleutian Islands, Alaska to southern Vancouver Island. The Haida Gwaii population may represent 60% to 70% of the total British Columbia population. These falcons tend to use long-term territories around their nests (eyries) on ledges of rocky cliffs nearby seabird breeding colonies. Peale's Peregrine Falcon were designated a *species of special* concern by COSEWIC in 1999 because they are limited by the availability of its seabird prey whose colonies have been experienced by predation by introduced mammals, the prospect of oil spills and global warming. Taking nestlings by falconers was unregulated until the 1960s and controlled under a permit system until 1972. Local opposition was instrumental in preventing the re-opening of a legal take in 1987. The British Columbia Ministry of Environment has led the Haida Gwaii coastal surveys since their inception in 1971. Since 1990, Parks Canada has collaborated on these archipelago-wide surveys that are a cooperative interagency (federal-provincial) monitoring initiative.

Results

Since 1971, a boat-based survey has been done around Haida Gwaii approximately every five years around late May when nesting is active - survey protocol is detailed by Schultze (2005). Using charts marked with historically recorded falcon territories, surveyors approach sea cliffs of potential territories by small boat and discharge firearms. Both historical sites and "all other cliffs

Table 23. The years during which the Canadian Wildlife Service has or will monitor three seabird species at *"key BC seabird colonies"* within Gwaii Haanas, 1984 to 2007 (data courtesy of M. Lemon, CWS and Hipfner *et al.* 2002).

Location	Species[1]	1984	1985	1991	1992	1996	2000	2002	2003	2005	2006	2007
SGang Gwaay	RHAU		X								X	
Rankine Island (west)	ANMU	X					X			X		
	CAAU	X					X			X		
East Copper Island	CAAU		X	X					X			
George Island	ANMU		X	X		X			X			
Ramsay Island	CAAU	X			X			X				X
	ANMU	X			X			X				X

1 RHAU = Rhinoceros Auklet / CAAU = Cassin's Auklet / ANMU = Ancient Murrelet

Table 24. The locations, population estimates, suspected trends and desired population sizes on *"key BC seabird colonies"* within Gwaii Haanas area made by the Canadian Wildlife Service (from Hipfner *et al.* 2002).

Location	Species[1]	Population Size[2]	No. of Permanent Plots	Population Trend[3] (no. of plots)	Desired Population Size[2]
SGang Gwaay	RHAU	10,500	8	ND	10,500
Rankine Island	ANMU	26,180	8	S (8)	26,000
	CAAU	35,000	8	D (8)	35,000
East Copper Island	CAAU	11,000	6	S (6)	11,000
George Island	ANMU	11,500	8	I (8)	11,500
Ramsay Island	CAAU	13,000	9	S (5)	13,000
	ANMU	18,000	12	I (11)	18,000

1 RHAU = Rhinoceros Auklet / CAAU = Cassin's Auklet / ANMU = Ancient Murrelet
2 estimates are of numbers of breeding pairs of birds
3 CWS trend conventions are: D = declining / S = stable / I = increasing
ND = no data as the second colony assessment did not occur until 2006 and these data are not yet analyzed

appearing even remotely likely to be used as an eyrie" were checked. If shots arouse a pair of birds (or a single) showing agitation and/or vocalization and a strong tendency to remain in the area, an "occupied" territory (with an assumed eyrie) is recorded. If a single bird leaves the area without hesitation (i.e., showing no territorial fidelity), the location is deemed "un-occupied" and no eyrie is assumed to be present.

The approximate distribution of all falcon territories recorded according to coastal segment of Haida Gwaii from 1971 to 2002 is shown in Figure 19. About 47% of these were recorded from the Gwaii Haanas area. The exact locations of eyries are in Gwaii Haanas' database, but these are kept secret for the security of eggs and young and cannot be shared without written permission from the British Columbia Ministry of Environment.

The estimated numbers of occupied falcon territories (containing eyries) from 1971 to 2005 are listed in Table 25. In 2005, the numbers of occupied territories and those "estimated" as occupied were the highest ever recorded. As well, there is a possibility of additional falcon territories outside the 2005 survey area (i.e., from Cumshewa Head north and west to Virago Sound). Numbers of single, non-nesting birds from 1990 onwards are shown in Table 26 . Productivity estimates of number of chicks per occupied territory are based on Langara Island data that is the most densely populated Peale's Peregrine Falcon area in Canada. The productivity, extrapolated from Langara Island data, for all of Haida Gwaii was 2.0 in 2005, compared to 1.0 for 2000 and 1.33 for 1995 (Schultz 2005). The

Table 25. The numbers of occupied territories (with eyries [nests]) of Peale's Peregrine Falcon around Haida Gwaii, 1971 to 2005 (from Schultze 2005).

Year	Number of Occupied Territories	Total Estimated[1] Occupied Territories	Number of Locations[2] Inspected
1971	62	ND	62
1975	66	ND	109
1980	79	ND	108
1986	56	ND	147
1990	71	78	136
1995	69	75	162
2000	67	76	166
2005	83	91	222

ND = no data
1 ncludes extrapolated values to unsurveyed portions of the total study area as well as the applied correction factor for expected survey error of 1.1 (Schultze 2005)
2 this is a very general indicator of the extent of monitoring - only locations with historical activity and an assigned number are included here; the number of locations is not strictly comparable between survey years as new locations are added in each survey; finally, since 1990, protocol requires that hundreds of additional potential sites (without assigned numbers) are checked

Table 26. Numbers of single, non-territorial Peale's Peregrine Falcons around Haida Gwaii, 1990 to 2005 (from Schultze 2005).

Year	Number of Single, Non-territorial[1] Falcons
1990	17
1995	16
2000	2
2005	8

1 non-territorial birds are assumed to be non-nesting

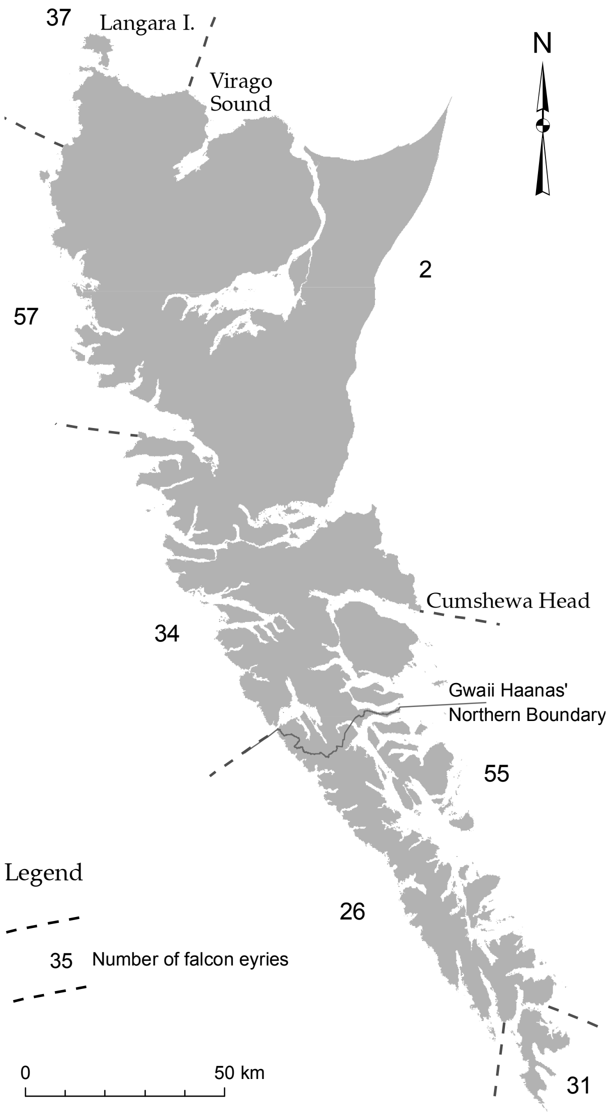


Figure 19. The number of all known Peale's peregrine falcon eyries according to coastline segment of Haida Gwaii from 1971 to 2002 (from Harfenist et al. 2002; Schultz 2005). In any given year, not all eyries are occupied and some may have been unoccupied for years. Specific eyrie locations are protected data.

status is "good" and the trend in the number of occupied territories since 1971 is "stable."

2.5.4. Measure 4 - Steller Sea Lion

Monitoring Question
What are the numbers and trends of pup and adult Steller sea lions attending the rookery at Cape St. James?

Context
Steller sea lions (*Eumetopias jubatus*) are Earth's largest eared seals. Bulls in prime pre-breeding condition can exceed 1,100 kg and cows ~350 kg. Steller sea lions are the most researched marine mammal in the North Pacific, and their life history around Haida Gwaii is fully reviewed in Heise et al. (2003). This non-migratory species of (mainly fish) predator ranges from southern California to Alaska, across the Aleutians and Bering Sea and southward to the Sea of Japan. In the western Gulf of Alaska, their population declined by more than 80% from the late 1970s to the late 1990s while the eastern (British Columbia-Southeast Alaska) population increased during that same period (Guénette et al. 2006) - and is continuing to do so (P. Olesiuk, DFO, personal communication). Nonetheless, the Canadian population was listed as a *Species of Special Concern* in 2003 because of population instability throughout the species' total range.

Key to population monitoring is that Steller sea lions aggregate at the following three types of shore sites where they can be counted: (1) breeding rookeries (locally active June - July) where they give birth, nurse pups, and mate; (2) year-round haulouts usually occupied continuously in consistent numbers throughout the year; and (3) winter haulouts used less regularly and primarily during the non-breeding season. Rookeries generally have peripheral haulout sites occupied mainly by non-breeding males and juveniles. In most cases, animals continue to use rookeries as haulouts year-round, albeit in much reduced numbers.

The shore sites used around Haida Gwaii are shown in Figure 20, including the region's only rookery as Cape St. James in Gwaii Haanas. Year-round haulouts have consistent numbers through seasons, including significant numbers during the breeding season. Summertime counts have been recorded from the rookery since 1913 (Table 27). Two winter counts yielded much lower numbers. Summer numbers had declined from the first assessment to ~1,000 by 1916. The population had recovered to ~2,500 (+1,500 pups) by 1956. After that, a series of annual culls (1959 to 1966) reduced the count sharply by 1961. After the 1970s, numbers have steadily increased.

Before legal protection (established 1970), Steller sea lions were subject to Fisheries and Oceans Canada (DFO)-managed culls for fishery predator control from 1913 and 1968, including around Haida Gwaii (Heise et al. 2003). Haida Gwaii largely escaped the pre 1950s culling era, but from 1959 to 1966, culls at Cape St. James totaled ~3,530 (including 278

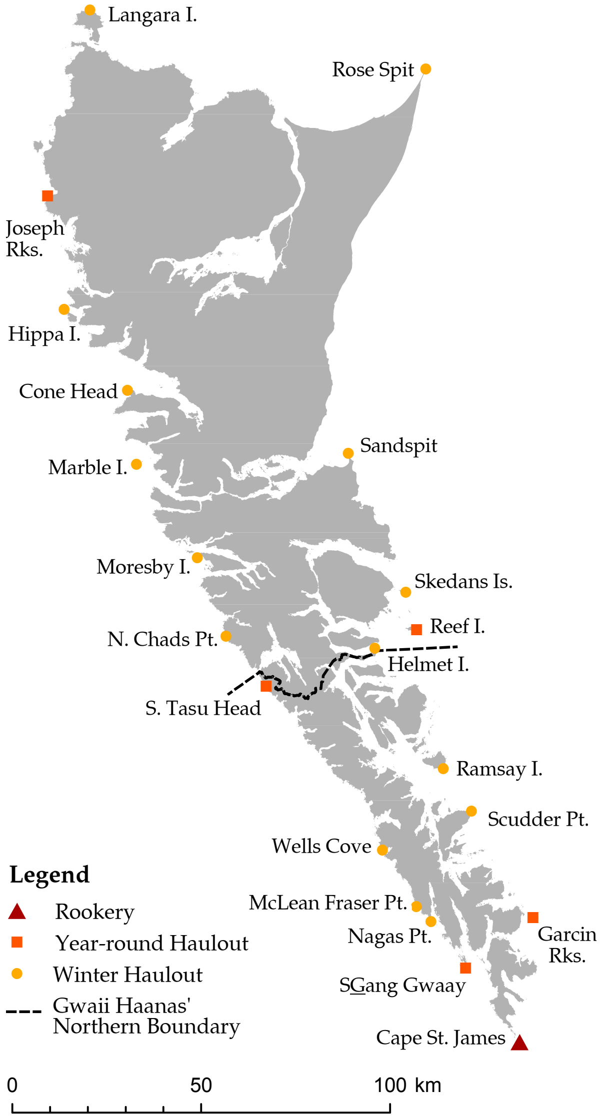


Figure 20. Steller sea lion rookery and haulouts (year-round and winter) around Haida Gwaii (from Heise et al. 2003). Counts are not made at winter haulouts.

pups). Besides the rookery, another 820 were killed at various haulouts around Haida Gwaii.

Results
With legal protection, the systematic aerial surveys by DFO began in 1971 and are now on a 4-year cycle. Surveys are flown during the last few days of June or first few days of July, by which time most pups have been born, but are not yet old enough to disperse. This enables estimates (from photographs) of total pup production, and total abundance by applying multipliers derived from life tables based on the expected ratio of pups to "non-pups" (those older than six months). Monitoring within the Gwaii Haanas area

Table 27. Summer counts of non-pup (over 6 months old) and pup Steller sea lions recorded from the Cape St. James rookery, 1913 to 2002 (from Heise *et al.* 2003). The years of aggressive culling were 1959 to 1966 with a peak of 1,438 sea lions in 1960. There are two winter counts (of non-pups): 258 in 1971 and 310 in 1976.

Year	Non-pups (pups)
1913	2,500
1916	1,000
1938	2,800
1956	2,500 (1,500)
1961	797 (644)
1971	631 (350[1])
1973	549 (282[1])
1977	782 (315[1])
1982	698 (420[1])
1987	1,021 (381[1])
1992	867 (503[1])
1994	797 (346[1])
1998	763 (503[1])
2002	982 (660[1])

1 adjusted pup counts using a correction factor invoked by multiplying the raw pup count by1.04 to account for the difference between pups not visible using oblique photography versus more pups visible using verticle overhead photography of the rookery.

would be another focus of federal interagency cooperation between Parks Canada and DFO.

The total number of non-pups around Haida Gwaii has increased at a mean rate of 2.9% annually (compared to 3.2% coast-wide) from 1971 to 2002 (2006 survey data are not yet available). Numbers were quite stable up to 1982, but subsequently increased more rapidly at an annual rate of 3.8%. In 2002, about 4,500 (including 660 pups) were estimated from around Haida Gwaii. This yields a non-pup:pup ratio of 5.8:1 compared to the 3.5:1 expected from life tables. The former ratio suggests that local waters support the local breeding population plus a surplus of foraging non-breeders associated with other rookeries, most likely Forrester Island (75 km north of Haida Gwaii). Accordingly, Haida Gwaii regional waters now represent the epicenter of the species' eastern Pacific distribution with its the positive population trend (P. Olesiuk, DFO, personal communication). Together, the rookeries Forrester Island, Cape St. James and the Scott Islands (170 km south of Haida Gwaii) support >60% of the northeast Pacific, and 35% of the total North American populations.

Listed in Table 28 are all the census results from sites in Gwaii Haanas from 1971 to 2002 (2006 data not yet available from DFO). The correction

Figure 28. Counts of non-pup (over 6 months old) and pup Steller sea lions according to site around Gwaii Haanas made in late June-early July, 1971 to 2002 (from Heise et al. 2003 and courtesy of P. Olesiuk, DFO, personal communication).

Site Name[1]	Site Type[2]	1971	1973	1977	1982	1987	1992	1994	1998	2002
Cape St. James	R	631	549	782	698	1,021	867	797	763	982
		350[3]	282	315	420	381	503	346	503	660
South Tasu Head	Y	76	NS	278	117	263	80	196	285	151
SG̲ang Gwaay	Y	NS	NS	NS	NS	44	279	617	359	313
Garcin Rocks	Y	NS	NS	NS	NS	NS	NS	NS	NS	329
	Non-pups	707	549	1,060	815	1,328	1,226	1,610	1,407	1,775
	Pups[3]	350	282	315	420	381	503	346	503	660

1 these sites are illustrated in Figure 20
2 R = rookery / Y = year-round haulout
3 adjusted pup counts: adjusted using a correction factor invoked by multiplying the raw pup count by 1.04 to account for the difference between pups not visible using oblique photography versus more pups visible using over-head photography of the rookery
4 NS = not surveyed

factor for pup counts when using various photographic techniques is given in Olesiuk et al. (2003). Although pup counts are relatively complete, 20 to 25% of non-pups are at sea feeding and likely missed during surveys (Olesiuk 2003). Pup production at Cape St. James increased from an average of 319 in the 1970s to 660 in 2002. The total count at Cape St. James in 2002 was roughly 66% of the estimated number present at the relatively undisturbed rookery in 1913.

Using just the rookery results only as the measure, the status is “good” and the trend is “improving.” There has been a significant increase in the number of sea lions at the rookery since surveys began in 1971. This is true for the total count (pups + non-pups), non-pups (animals older than six months) and pups.

2.5.5. Measure 5 - Keen’s Long-eared Bat

Monitoring Question

Is Keen’s long-eared bat continuing to use the maternity roosts at Gandl K’in Gwaayaay (Hotspring Island) to rear young?

Context

Keen’s long-eared bat (*Myotis keenii*) was first discovered on Haida Gwaii (Burles et al. 2004). It has since been found in only a few localities along the British Columbia coast, southeast Alaska and northwestern Washington State, and is believed to have one of the most restricted distributions of any bat in North America. Because of this restricted distribution and apparent rarity, it was identified nationally as a *Species of Special Concern* by COSEWIC in 1989. In November 2003, an updated status report on this species was reviewed by COSEWIC, with the conclusion that it should be reclassified to *Data Deficient* until its taxonomic relationship with the very similar western long-eared bat (*Myotis evotis*) are clarified. The recommendation of the report is that molecular genetic studies be conducted.

Keen’s long-eared bat is known from only 25 locations in Canada and 13 in the United States. The only known maternity colony for this species is located at Gandll K’in Gwaayaay (Hotspring Island) in Gwaii Haanas. First confirmed as a maternity roost in 1991, the colony of both Keen’s and Little brown (*Myotis lucifugus alascensis*) bats has been monitored since 1998. The colony is unusual in that the roosts are geothermally-heated crevices that provide ideal conditions for rearing young. An intensive study of the colony was conducted during 1998 and 1999 to determine the number of each species using the roosts, as well as their breeding biology and foraging habits (Burles 2001).

Results

A standardized protocol for conducting emergence counts was developed during the initial study, and has since been used to continue to monitor the number of bats using the roosts (Table 29). Count results have been variable over the years, due mainly to variation in emergence because of weather conditions (Burles 2001), but show that overall numbers have remained relatively stable since 1998. The number of Keen’s long-eared bats appears to have decreased during 2002 to 2003, however it is uncertain whether this decline was real. The calculation of the relative numbers of each species depends on the observer identifying emerging bats to species by their flight behaviour and echolocation call characteristics, which is not always possible for every bat seen. In 2006, a time expansion detector system was used to record emerging bats, and subsequently identify them to species. This new technology showed great promise to enhance the

Table 29. Summary of emergence counts and relative numbers of Keen's long-eared bat (*Myotis keenii*) and little brown bat (*Myotis lucifugus alascensis*) present at Gandll K'in Gwaayaay (Hotspring Island), Gwaii Haanas.

Year	Average Number Counted	Highest Number Counted	Keen's long-eared bat (%)	Little bown bat (%)	Number of Counts
1998	51	110	49	51	10
1999	70	111	38	62	9
2000	126	126	32	68	1
2001	29	29	ND	ND	1
2002	76	96	18	82	2
2003	37	59	19	81	3
2004	53	83	22	78	5
2005	75	82	ND	ND	2
2006	95	122	33	67	3

ND = no data

accuracy and objectivity of classifying emerging bats to species and will be tested further in future years. Once this new equipment has been further tested, it may be possible to more clearly state the monitoring question, the species' status and establish thresholds for change.

2.5.6. Measure 6 - Coastal Erosion

Monitoring Question
What do the shoreline geoindicators reveal concerning climate change-driven erosion?

Context
The coastal zone is perhaps Earth's most dynamic and resilient biophysical system, within which the effects of climate change will be enormous (Harley et al. 2006). The coastal zone of Haida Gwaii is essentially a linked system of interdependent terrestrial and marine ecosystems and landforms (Sloan 2006). This includes offshore, nearshore and backshore environments. Given the complexity, sensitivity and responsiveness of the coastal system, effective management requires monitoring and research. This is particularly so in light of rapidly changing pressures on coastal parks such as climate change, tourism, and increasing development.

Beaches and coastlines naturally strive to attain a form in balance with wind, wave and tidal regimes. As each of these processes have their own cycles and extremes, beaches constantly change their form and function over time. In the winter, many beaches appear steeper and coarser as they attain balance with strong winter winds and waves. In contrast, beaches during summer may appear flatter and finer as lesser summer waves move sediment onshore that was removed during the winter. Thus, what is seen on a beach at a point in time may be in balance with some processes (e.g., wave swash, tides) but out of phase with others (e.g., winter storm waves). For this reason, beaches are viewed as inherently "unstable." As such, erosion and sedimentation are also natural responses of beaches as they adjust to seasonal or interannual changes in weather and ocean conditions. In the face of increasing climate variability (e.g., storminess) and change (e.g., sea-level rise), it is uncertain how beaches will respond.

Here, the objectives, approach and initial results from a geoindicator-based monitoring program to assess coastal erosion and potential climate change effects within Gwaii Haanas are reported. The first phase of a 5-year (2006 to 2011) monitoring project of key geoindicators linked to coastal erosion and climate change effects within Gwaii Haanas is completed (Walker 2006). A reconnaissance of selected sites was conducted in June 2005 to assess their geomorphology and erosion potential. From this, initial recommendations for protecting and/or preserving culturally and ecologically important sites form the groundwork for the monitoring network for sandy beach-dune systems in Gwaii Haanas. Background material on coastal geomorphology, related geoindicators, and climate change is provided for context.

The Gwaii Haanas program is part of a larger, 3-park initiative: "Climate variability and change impacts monitoring of beach-dune systems on the Pacific Coast" that will involve similar activities in Pacific Rim and Gulf Islands national parks. This initiative addresses several key "geoindicators" identified by Parks Canada (Welch 2002) for climate change effects monitoring including: shorelines, beach-dune systems, sea-level change, coastal sediment transport, and wave erosion. The 3-park network covers a wide spatial scale of the Pacific coast over an appropriate time to begin identifying regionally specific responses over varying beach types, tide ranges, and sea-level rise signals. In turn, this will enhance our understanding of the vulnerability of coastal sites with key cultural and ecological value and will aid in developing strategies to reduce future impacts.

Many of the sandy beach-dune systems in Gwaii Haanas host distinct ecosystems and provide specific, limited habitat. Selected beach-dune systems and culturally significant sites are documented. These systems are highly responsive to climate variability events (e.g., El Niño seasons) and climate change trends (e.g., sea-level rise). For instance, during the El Niño of 1997-98, mean sea level in Hecate Strait was 0.4 m higher due to warmer oceans and enhanced erosion from 1-3 to 12 m/yr occurred on NE Graham Island (Barrie and Conway 2002). SE storms have also increased since the mid 1970s (Abeysirigunawardena and Walker in review), perhaps due to a regime shift in 1976 in a longer cycle (20 to 30 year) pattern known as the Pacific Interdecadal Oscillation (PDO) (Hare and Mantua 2000). The long-term eustatic trend of sea-level rise in Hecate Strait is +1.6 mm/yr and extreme annual water levels are rising at more than twice this rate (+3.4 mm/yr) (Abeysirigunawardena and Walker in review). Climatic variability and change are influencing beach structure via changes in formative wind, wave and water level regimes. Oceanographic and geomorphic responses vary regionally however, with beach aspect, wave exposure, sediment supply and tide range.

Little research exists linking climatic variability and change to longer-term, regional coastal landscape changes in British Columbia. In particular, continuous climate data, water levels and beach profile monitoring data are sparse. To address this, the purpose of this research is to collect and analyze baseline data for a suite of key coastal geoindicators including: dunes, shorelines, sea-level, coastal erosion/sediment transport, and marine nearshore environments (Welch 2002).

Specific Gwaii Haanas project objectives include:

- to establish geoindicator monitoring sites at sandy beach-dune systems and key cultural sites to detect interannual changes;
- to interpret and classify the geomorphic controls and morphodynamics that govern beach-dune form at each site (e.g., the study by Cumming (2007) at Woodruff and Gilbert Bays);
- to identify regional climatic variability and longer-term climatic change-induced signals in wind, wave and total water level regimes;
- to assess interannual responses of monitoring sites (e.g., erosion and/or rebuilding) to possible causal mechanisms (e.g., high water level erosion events, Aeolian (windblown) sand transport potential) to set the context for longer-term regimes in erosive and/or rebuilding events; and
- to provide recommendations to Gwaii Haanas for long-term planning and management of coastal erosion and geomorphic change.

Climate is defined by long-term patterns of daily to seasonal weather properties (e.g., temperature, precipitation, wind speed and direction, solar energy). For instance, Environment Canada's "Climate Normals" are determined by long-term averages for a 30-year period - the most recent being 1971to 2000. These are useful for comparing daily observations to "normal" conditions for an area as departures from the average and also allow for identification of longer-term climate change (CC) trends.

In British Columbia, sea-levels have risen by 4 to 16 cm over the past century (Abeysirigunawardena and Walker in review; BC-MOE 2002), with the higher rate of rise (1.6 mm/yr) occurring in Hecate Strait. While this occurs, the height of annual extreme (storm generated) water levels is also happening at 3.4 mm/yr at Prince Rupert and 1.6 mm/yr at Vancouver (Abeysirigunawardena and Walker in review; BC-MOE 2002). These extreme events pose more immediate coastal damage and hazards and changes in their frequency and magnitude have been related to known climate variability phenomena.

Climatic variability (CV) is a part of the natural climate system and is defined as shorter-term (i.e., seasonal to interannual) fluctuations above long-term averages in key climate variables (e.g., air temperature, precipitation type and amount, ocean surface temperature, sea level). In the northeast Pacific, large variations in climate and ocean levels are common and can be extreme. Such variations are often driven by known ocean-atmosphere phenomena such as the El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO). Recently, changes in the frequency and magnitude of extreme events (e.g., storm surges, droughts, floods, fires) are occurring and have been linked to more extreme ENSO and PDO phases.

Evidence exists linking increasing CV events to climate change. CV is experienced on various spatial scales (i.e., locally to regionally) and on temporal scales typically longer than that of individual weather events. These impacts pose serious challenges for managing EI, especially in coastal parks where linked knowledge of terrestrial and marine systems is lacking. Recent examination of long-term meteorological and oceanic data on the north coast shows changes in both shorter-term (i.e., inter-annual) CV patterns (e.g., seasonal temperatures, SE storm winds, storm surges) as well as longer-term (i.e., inter-decadal) CC trends (e.g., sea level rise, ocean surface temperatures). Current trends in Haida Gwaii show increasing storminess and a rise in extreme water levels that is twice the rate of ongoing sea-level rise (3.4 mm/yr Vs 1.6 mm/yr) (Abeysirigunawardena and Walker in review).

Coastal geomorphic research along northeast Graham Island suggests a recent increase in coastal erosion and/or sedimentation in response to these changes, particularly during strong El Niño years (e.g., 1982-83, 1997-98) when conditions favour elevated sea-levels and increased storminess due to warmer ocean currents and changes in regional weather patterns (Walker and Barrie 2006). Although much of the shoreline of Gwaii Haanas is rocky, steep, ecologically diverse and, thus, potentially resilient to climate variability and change impacts, many of the culturally significant sites and embayed beaches are experiencing erosion. The extent and rate of this, as well as ensuing ecological effects, remain uncertain and require monitoring.

Both CV events and coastal erosion are naturally occurring phenomena. However, a growing scientific consensus exists that climate changes, including changing frequency and magnitude of extreme climate variability events (e.g., droughts, floods, forest fires, coastal storm surges), may be increasing as our ocean-atmospheric system adjusts to greenhouse gas-induced global warming (IPCC 2001). In addition, well-known natural CV patterns (e.g., El Niño) that already drive regional weather and oceanic circulation, may also be exacerbated (Timmermann et al. 1999). What remains uncertain is whether interdependent coastal (marine and terrestrial) ecosystems can maintain ecological integrity and resilience in the face of such relatively rapid climate and landscape changes. This is of particular concern for Parks Canada's protected coastal ecosystems and should be considered proactively for current and future management.

Erosion is a net loss of material (usually sediment) from a beach system. A beach is not considered erosional unless there is a net sediment loss over some time period (years). In response, the shoreline may retreat landward. For instance, some areas of East Beach in Naikoon Park, NE Graham Island are retreating at 1-3 m/yr over the past few decades and more rapidly (10s m/yr) in response to extreme events (Walker and Barrie 2006). Erosional features common on beaches include scarps or bluffs – typically steep-faced, exposed sedimentary slopes that can range from tens of cm to 100 m or more (e.g., Cape Ball on East Beach). River outflows may also cause incised (notched) channels that may have developed over tidal cycles, seasons or thousands of years as the elevation of the river system relative to the ocean changes in response to sedimentation and/or sea-level changes.

If a beach receives more sediment than it loses, it is accreting. Sediment may accumulate in new or existing dunes and/or beach ridges. Over time, sediment accretion may cause the shoreline to advance seaward via progradation. For instance, North Beach has prograded several hundred metres seaward over the past 3,000 years by means of foredune and beach ridge accretion. Both erosion and progradation can occur very gradually. Erosion is often occurs as episodic events linked to winter storms or more extreme events such as the Christmas Eve 2003 storm surge. When subjected to waves and storm surges, most beaches will erode and their sediment will move seaward. Often this sediment is stored in nearshore bar systems or it moves along the beach in strong nearshore currents. If stored in bars, this sediment may return to the beach during less energetic wave conditions (e.g., summer). If moved in nearshore currents, the sediment can travel great distances and may never return to the beach from which it was removed.

Beach landforms reveal important information on the changes experienced by the shoreline in response to wind and wave action. Accretional features common on sedimentary beaches include beach berms on the upper shoreface formed by wave swash. During the falling stages of the spring – neap tidal cycle, berm platforms (or tidal terraces), often cuspate in form, can form at 2 to 3 heights on the beach. This occurs as the locus of wave action that forms the berms and cusps drops with the falling tide level toward the neap. Many macro-tidal beaches also display a flat, often sandy, low tide terrace only observed at the lower low tides.

Wind-transported (aeolian) sands are an important and often overlooked component of the coastal sediment budget. Aeolian sands moved from the beach are deposited in shore-parallel dunes or foredunes. Foredunes are commonly vegetated with dune grasses and provide distinct habitat for many rodents, shorebirds, prey birds and insects. Older, established foredunes can reach 10 to 15 m in height and are commonly vegetated with various tree species including Sitka spruce, Shore pine, Western hemlock and Red alder. Foredunes act as important stores of sediment that buffer or protect shorelines against wave attack, storm surge flooding and gradual sea-level rise. Recent research suggests that dune migration (landward movement) is an important process that must be maintained to allow accreting shorelines to continue to buffer ongoing sea-level rise impacts (Davidson-Arnott 2005).

Driftwood jams are very common. These jams and other flotsam act as an important accretion anchor for coastal sediments in the backshore buffering the shoreline against wave attack and providing new sites for vegetation colonization (Walker and Barrie 2006). By generating extra turbulence in airflow over the beach, driftwood jams cause wind-transported sand to be deposited. Sand accretion in driftwood jams occurs very rapidly and, provided a subsequent high water level event does not remove the debris, can lead to the development of new, incipient dunes in the backshore. Incipient dunes can also develop on the beach in the absence of driftwood via beach vegetation colonization, though this is not as common in Haida Gwaii given the macrotidal, energetic wave regime.

Geoindicators

Geoindicators are measures of geological processes occurring at or near the Earth's surface that can be monitored for understanding changes in magnitude, frequency, trend, extremes, or rates of natural processes over periods of less than 100 yrs. A list of 27 geoindicators was identified by International Union of Geological Sciences in 1996 and was revised for Parks Canada's interests by Welch (2002) to include other key coastal elements such as: coastal sediment transport, extreme events, marine nearshore environments, proxy record, and built environments. Table 30 lists key geoindicators relevant for monitoring coastal change and includes descriptions of key geomorphic measures and links to climate variability and change via responses in related geomorphic

Table 30. Key geoindicators for monitoring coastal change including relevant geomorphic measures, linkages to climate variability/change via formative process regimes, and overall relevance and practicality for ecological integrity monitoring (modified from Walker 2006).

Geoindicator	Geomorphic measures(s)	Process linkage to Climate Variability/Climate Change	Relevance and Practicality
Dunes	Size, type, position, migration rates	Δ[1] wind regime with ENSO[1]	High, High
Wind erosion	Aeolian (wind-driven) drift potential sand traps, erosion pins	Δ wind regime with ENSO	Medium, High
Shorelines	Position, slope, features, sediments, beach width	Δ TWL and/or wave regimes, SLR[1] linked to ENSO	High, Medium
Relative sea-level	TWL[1] = tide +/- surge; land rise/subsidence	Δ TWL with storms, ENSO, PDO[1] eustatic SLR	Low, Low[2]
Soil/sediment erosion	Erosion/deposition pins, topographic profiles, sand traps	Δ TWL, wave and/or wind regimes, SLR linked to ENSO	High, Low
Nearshore marine environment	Profiles, swath bathymetry, sediments, features	Δ TWL and/or wave regimes, SLR linked to ENSO	Medium, Low
Extreme events	TWL, winds, waves, erosion relative to geomorphic thresholds or encroachment on built environments/infrastructure	Δ TWL, wave and/or wind regimes linked to ENSO, PDO	Medium, Medium[2]
Built environments	Footprint areas, distance from shorelines/dunes/flood zones	Δ TWL and/or wave regimes, SLR	High, High
Proxy (paleontological) record	Soil/sediment pits, tree rings, ^{14}C materials, geoarch, optical dating	Δ climate, TWL, wind, wave regimes linked to ENSO, PDO	Low, Medium

1 Δ = change in; TWL = total water level; ENSO = El Niño-Southern Oscillation; PDO = Pacific Decadal Oscillation; SLR = sea level rise
2 for Gwaii Haanas, these adjusted from Low to Medium or from Medium to High in relevance (Walker 2006)

processes (e.g., wind, wave and water level regimes). The relevance to EI and practicality of each (compared to Welsh 2002) for monitoring and managing ecosystem integrity is listed.

Methods
The methods for this research and proposed objectives, from Walker (2006), are as follows:

- Repeat terrestrial and bathymetric surveys using a laser total station from georeferenced benchmarks and a boat-mounted depth sounder, respectively, to gather data on nearshore-backshore topography. Profile morphology and changes can be used to classify and ascribe responses of site morphodynamics to natural regimes, extreme events and longer-term (seasonal to interannual) trends. From this, and airphoto/GIS work, erosion and/or progradation rates can also be determined.
- Tree-ring (dendrochronological) sampling using a non-destructive small gauge increment borer and/or chainsaw (to disk fallen snags) to obtain information on tree growth responses to on-site climatic variations and potential geomorphic changes (e.g., dune stabilization and/or migration, aeolian abrasion, coastal flooding). Not yet concluded.
- Sediment and datable materials sampling (e.g., charcoal for ºC, sediments for optical dating) from approved soil pit locations. This allows reconstruction of pre-historical landscape and sea-level changes and will be conducted in collaboration with the Geological Survey of Canada. Not yet concluded.
- Airphoto interpretation and GIS analyses to identify geomorphic features and changes such as coastal erosion and/or progradation responses and rates.
- Analyses of regional meteorological and oceanographic data, obtained from Environment Canada met stations and buoys and the Canadian Hydrographic Survey tidal stations respectively, using spreadsheets and statistical analysis software. From this, recurrence interval curves for extreme events (e.g., 10-yr or 100-yr water levels) will be produced. These are useful for management purposes (e.g., coastal setbacks for flooding or erosion hazards). In addition, trends and variations in climatic and oceanic variables will be examined in relation to various climatic variability indices (e.g., ENSO, NOI, PDO).

Results
Sites assessed within Gwaii Haanas (Figure 21) were selected either for their cultural significance (village sites) experiencing coastal erosion or sandy beach sites. As well, these beach-backshore systems are responsive to both extreme water levels and storm wind events possibly linked to CV and CC effects, and these systems provide a biophysical setting that is ecologically distinct within Gwaii Haanas. Brief site descriptions with monitoring activities are listed in Table 31 – much greater detail is provided in Walker (2006). To interpret modern geomorphology and potential responses to climate variability and change, cross-shore monitoring transects were established, as illustrated in Walker (2006).

Gilbert Bay is an exceptional example of a prograding, embayed beach-dune system with well developed, actively accreting and prograding foredune ridges. This is evidenced by a sequence of stabilized dune ridges in the forest and by an actively accreting, incipient driftwood jam dune on the beach. There is also a lack of eroded dune scarps such as those found on most beach-dune

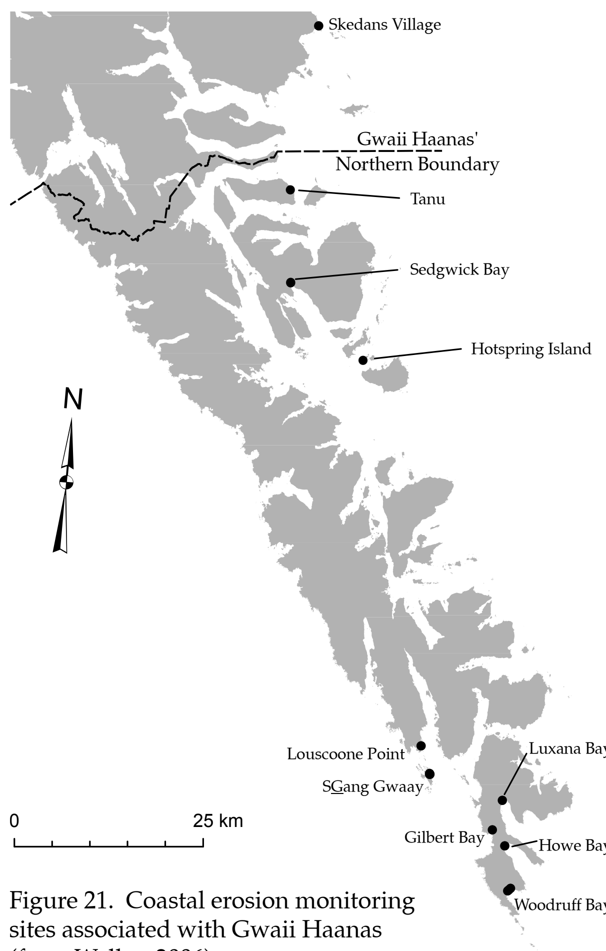


Figure 21. Coastal erosion monitoring sites associated with Gwaii Haanas (from Walker 2006).

Table 31. Descriptions of sites and activities including coastal monitoring profiles in Gwaii Haanas (from Walker 2006).

Site (*Type*)	Description of Site, Benchmarking and Profile
Gilbert Bay *(Sandy beach-dune site)*	Prograding beach-dune site with several stabilized dune ridges in forest and driftwood jam on beach; 2 benchmarks established; a profile surveyed several hundred m landward of the second benchmark atop a large, stabilized (parabolic?) dune ridge
SG̲aang Gwaay (*Cultural site*)	An embayed village site; 8 monitoring profiles extend seaward from benchmarks and/or totem poles; evidence for recent backshore erosion from high water levels and tourist traffic
Louscoone Point (*Cultural site*)	Eroding midden berm, at a campsite; profile extends seaward from forest behind campsite
Woodruff Bay *(Sandy beach-dune site)*	Beach-dune site with eroded driftwood jam terrace fronting larger, stabilized prograded (?) dune ridges; 1-3 profiles from S-N on site, extend over stabilized dunes onto beach; one is particularly good for optical dating of dune events/stabilization with several dune ridges progressing inland ~1km before dropping into river
Howe Bay (*Gravel-sand beach*)	Gravel/sand, cuspate beach with cobble low tide terrace and extensive backshore driftwood jam; profile from forest onto beach, near small creek outflow, ~mid-beach, trees flagged near benchmark
Luxana Bay (*Gravel-sand beach*)	Gravel/sand beach with extensive backshore driftwood jam and storm-rafted debris ~200m into forest; recent beach ridge progradation
Skedans (*Cultural site*)	Village site on exposed gravel isthmus (aligned 110°); two profiles extending to N and S beaches; on the north, embayed, sandy/gravel with bedrock outcrops and cobbled intertidal driftwood jam; on the south embayed, steep gravel face, 15 m driftwood jam
Sedgwick Bay (*Gravel-sand beach*)	Gravel beach backed by ~15m driftwood jam fronting a backshore swale (wetland); flat coastal plain extends ~100m landward of swale to relict scarp in forest
Tanu (*Cultural site*)	Eroding, low beach bluff; some gravel swash deposits and erosion potential near cemetery and Watchmen cabin
Hotspring Island (*Cultural site*)	Eroding west-facing beach with storm-rafted debris near change house and beach pool

sites on the east coast (e.g., Woodruff Bay). The drop in elevation (over 20m to the beach) and relative height of the stabilized and modern dunes at Gilbert Bay (~5 to 15 m) suggests either changes in onshore sediment supply and/or wind regime or, hypothetically, relative sea-level regression (drop) at this site due to tectonic forces.

A beach scarp and driftwood at the SGaang Gwaay village site are indicative of tidal encroachment and localized storm erosion. The frequency and extent of this cannot be assessed without more detailed airphoto analyses, examination of regional meteorological and oceanographic data, historical and traditional accounts, and continued onsite monitoring. Several totems are within metres of the eroding scarp.

The beach scarp and exposed midden at Louscoone Inlet are indicative of tidal encroachment, localized storm erosion, and most likely, tourist traffic over the berm to the campsites (Figure 22). This figure also serves as an example of profiling a site. Continued profile monitoring is required to estimate erosion rates, as the scarp is not visible on airphotos. One shore-normal monitoring profile was established in the central portion of the eroding berm that extends ~30m landward through the campsites. This midden site is actively eroding and will continue to do so naturally as it is highly exposed to higher water levels. Full preservation of the site is not likely, but protection via closed access to backshore bluff face (e.g., shell or rock line 2 to 3 m from the scarp) is recommended to reduce rates of erosion.

Woodruff Bay hosts a significant pre-historic dune field that, based on distinct scarping of the established foredune and near complete removal of a smaller, modern driftwood jam complex seaward of the foredune, has seen significant recent erosion. This is common to beaches with E-SE aspect as they face into the exposed SE storm fetch of Queen Charlotte Sound and there is evidence for regional sea-level rise and increasing extreme water levels (Abeysirigunawardena & Walker, in review). Dune geomorphology at Woodruff Bay suggests that, pre-historically, this dune complex was much larger, more active, and possibly, prograding seaward. Confirmation of

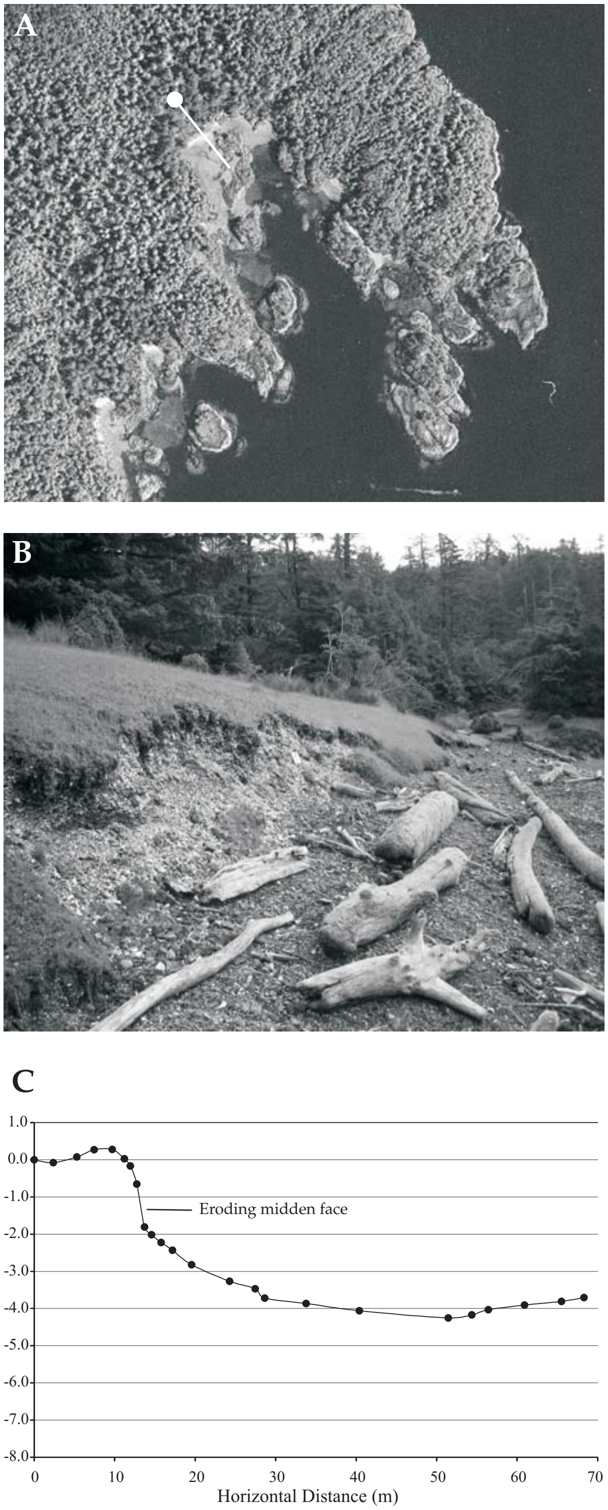


Figure 22. (a) Airphoto of Louscoone Inlet showing monitoring transect and BM locations, (b) eroding backshore scarp exposing a midden site on the beach (c) topographic profile of 3 June 2006 (from Walker 2006).

rates, timing, and possible events of retreat and/or progradation must be assessed from further airphoto analyses, optical dating of dune ridges, dendrochronological studies, and examination of total water level and storm surge regimes.

Beach ridge features at Howe Bay were likely formed by wave swash rather than winds. The sets of ridges suggest limited late Holocene progradation about a falling datum (i.e., sea-level). Further research would be required to confirm a sea level regression signal at this site (e.g., more extensive mapping, sediment coring, optical dating). The modern beach and backshore geomorphology suggests localized erosion and/or retreat in response to fairly recent high water level events. A scarp at the backshore is laden with, and overlain by drift logs and debris at the grass line. There is little sediment accumulation in this drift and it has not been colonized by trees, suggesting recent deposition, possibly within the last 5 years. The beach has much less fluvial inputs compared to Luxana and Woodruff, and it is very exposed to SE waves and surge. More detailed examination of historical airphotos is required to estimate current erosion rates, but it is likely the beach will erode and retreat in response to observed changes in storminess and sea-level rise, depending on rates of onshore littoral sediment supply (which appear limited).

There is little evidence for extensive beach progradation or aeolian activity at Luxana Bay. Recent (historical) accretion is evidenced by the driftwood jam being colonized by hemlock, however a fairly recent (decadal scale) scarp behind this drift jam suggests that this section of beach could be eroded by a high magnitude storm surge. Wind rafted debris deep into the backshore suggests that recent storms may not have coincided with high water levels within the bay. Long bay length and orientation being slightly askew from SE, may complicate the incoming wave field compared to other more exposed bays such as Woodruff and Howe.

Overall, Sedgwick Bay beach shows recent erosion and storm surge impacts, particularly along the southern half of the beach and NE side of the bay near the protest camp. This is not surprising given the open SE exposure into Juan Perez Sound. Near the river at the SW end, a large in situ stump with springboard notches is found on the beach, suggesting at least 7m of erosion since this tree was logged (i.e., the stump is located seaward of the modern beach scarp, and would have been in the forest when logged). Several undercut and windthrow snags also exist in the vicinity. Historical artifacts (hatchet head, plate fragments, metal tin) were found atop the storm berm and at the base of the blowdowns. This too suggests extensive erosion and a possible historical settlement alongside the river. Some localised accretion is evident, however, along the northern half of the beach as an extensive and stabilizing backshore driftwood jam. It is uncertain whether this portion of the beach is prograding, without airphoto analyses and sedimentary analyses, however the topographic profile does show some stepwise transgression from the benchmark into the intertidal zone. The presence of large swash bars and storm berms suggests that the beach may be both tidally- and storm wave-dominated in terms of its sedimentary dynamics.

Skedans Village (K'uuna llnagaay) is an isthmus exposed to seasonal storm winds and waves from both N and SE. This site is particularly vulnerable to coastal erosion and increasingly so with observed sea-level rise and increasing storminess. Continued erosion, extreme storms, and sea-level rise could see complete flooding and/or erosion of the isthmus within 200 to 300 years, assuming a conservative rate of 3.4 mm/yr for future extreme water levels. This estimate may be generous as it does not consider increases in rates of both average sea-level rise and storm-generated extreme events (i.e., it only considers the historical trend to modern day). Both of these rates are anticipated to accelerate with future climate changes. Eventually, the site could be completely eroded or submerged. Continued monitoring and further airphoto analyses is required to verify and refine current rates of erosion.

Given the complicated geomorphology of the Tanu Village (T'aanuu llnagaay) beach (obstructed by bedrock outcops and reefs), a monitoring profile was not established at Tanu as a representative site could not be identified. Shell markers used to direct tourist traffic along backshore are an effective means of reducing rates of accelerated beach bluff erosion. This should also be used at other sites of concern such as Louscoone, SGang Gwaay, and Skedans. There are some locations at Tanu, however, that direct traffic across the scarp due to recent erosion of the former pathway. These locations should be re-routed. Grave sites are very vulnerable to wave/surge exposure and may be impacted in the near future. The Haida Watchmen cabin is also very close to the erosive bluff and drift log line. Cabin relocation is recommended for future site development and/or upgrades.

The section of Hotspring Island (Gandll K'in Gwaayaay) beach between the hotsprings and Watchmen's cabin show no erosive features (e.g., scarps), however, storm surge deposits and drift logs are present within metres of structures and pools. Beach structures are highly vulnerable to continued storm damage, even though this side of the island does not face openly toward SE storms. The stairs to the Watchmen's cabin fall within metres of the active beach and the structure itself may be at risk to future high storm surge damage, though unlike the other structures, the threat is not as immediate. Erosion monitoring profiles were not established at this site, given it's complex geomorphology.

To summarize, key beach and cultural sites were visited in 2005 and 2006 for initial geomorphic and erosion assessments. At some sites, cross-shore topographic monitoring profiles were installed. Though some of these profiles were measured in 2005, instrumentation used was rudimentary and the profiles are not accurate. In 2006, a precise laser total survey station was used to resurvey these sites, and semi-permanent benchmarks were installed and geopositioned. This network of sites forms the basis for an ongoing monitoring effort of coastal erosion and potential climate change impacts resulting from sea-level rise and increasing storm surge damage. Status is highly site-specific, but "fair" overall. Inexorable sea level rise means that the trend is "deteriorating." The Cultural resource consequence of the eroding village sites is negative. Based on these site assessments, the following key recommendations are made for monitoring and protecting beach and cultural sites within Gwaii Haanas (Table 32).

2.5.7. Measure 7 - Raccoons on Seabird Islands

Monitoring Question
Have raccoons reached any of the seabird colony islands designated as being vulnerable?

Context
Raccoons (*Procyon lotor*) were introduced to Haida Gwaii in the early 1940s to provide another source of fur for local trappers (Gaston et al. 2007 a). Shortly after their introduction, demand for their fur plummeted and trapping ceased. The raccoons quickly spread throughout Graham, Moresby and to some smaller offshore islands.

Raccoons on Haida Gwaii are capable of swimming up to 600 m, making many offshore islands vulnerable to colonization (Harfenist

Table 32. Key management recommendations for monitoring and protecting sandy beach and cultural sites within Gwaii Haanas (from Walker 2006).

Recommendation Issue	Notes on Recommendations
Shell-mark paths and signage	Direct tourist traffic away from eroding beach and bluff scarp at key cultural sites (e.g., at Tanu, see also Section 6.3 in the State of the Park Report).
Update orientation information	Inform visitors of the effects of climate change and coastal erosion and how they can minimize their visitor effects
Tanu village graves	Tanu, being at risk from wave attack and erosion, it may be prudent for families to consider re-locating the graves
Skedans village	The village is highly vulnerable to coastal erosion and storm surge attack from both N and SE seasonal storms and sea level rise; aside from careful control of tourist traffic (e.g., shell pathways to guide tourist access), little can be done to protect this site in the long-term - it is possible that the site may be flooded and/or significantly eroded within the next century
Haida Watchman cabins	Some cabins are within metres of eroding beach scarps and storm drift log lines; cabin relocation with a greater backset from the shoreline is recommended for the future
Fueling stations	These are sited near/on active beaches and erosive scarps at Parks facilities and Watchmen's sites (e.g., Hotsprings, Huxley); to avoid fuel spills and contamination, relocate to stable, sheltered backshore locations preferably bedrock or cement pads with spill catchments and sufficient clearance from coastal erosion
Continue monitoring	For storm surge and sea-level rise effects, repeat <u>survey of coastal erosion profiles</u> and installation of basic <u>weather stations</u> for seasonal/annual monitoring

et al. 2002). By the late 1980s, raccoons were recorded from remote islands, and were likely responsible for the decline and perhaps abandonment of some seabird colonies (Harfenist et al. 2002). Raccoons were feeding to a large extent on intertidal invertebrates, as well as fish, terrestrial invertebrates and mammals. Raccoons that made it out to a seabird nesting island, however, shifted their diet to almost exclusively seabirds, with devastating results to colonies (Hartman et al. 1997).

Because of the vulnerability of offshore seabird colony islands to raccoons, a working group of British Columbia Ministries of Environment and of Parks, Parks Canada and the Canadian Wildlife Service was established in 1993. One of the first group tasks was to develop a strategy for monitoring and controlling raccoons on seabird colony islands. Islands separated from the main islands by <600 m were considered vulnerable.

Within Gwaii Haanas, 15 islands (Figure 23) were identified for annual monitoring for the presence of raccoons. Most support active seabird colonies but two (Shuttle and Faraday) are "stepping stone" islands to major seabird colonies.

<u>Results</u>

Monitoring consists of walking pre-determined shoreline segments on offshore islands to assess evidence of raccoons (e.g., latrines, dug burrows, headless seabirds). Should evidence of a raccoon(s) be found, actions such as further shoreline surveys, use of motion/infrared triggered cameras at bait stations, setting of live traps and night spotlight circuits of shorelines are taken. Immediate actions are either to trap or shoot raccoons during shoreline walks.

A summary of monitoring results is provided in Figure 23. To date, no raccoons have been found on any of the islands being monitored. The status of this measure is "good" (green) and the trend is "stable." There are currently no raccoons on any of the islands being monitored, as has been the case since monitoring began in 1993. For the status to change to "poor" (red) would be the discovery of one raccoon on one seabird colony island. The unconfirmed presence of a raccoon, or the presence of a raccoon on one of the stepping-stone islands would change the status to fair.

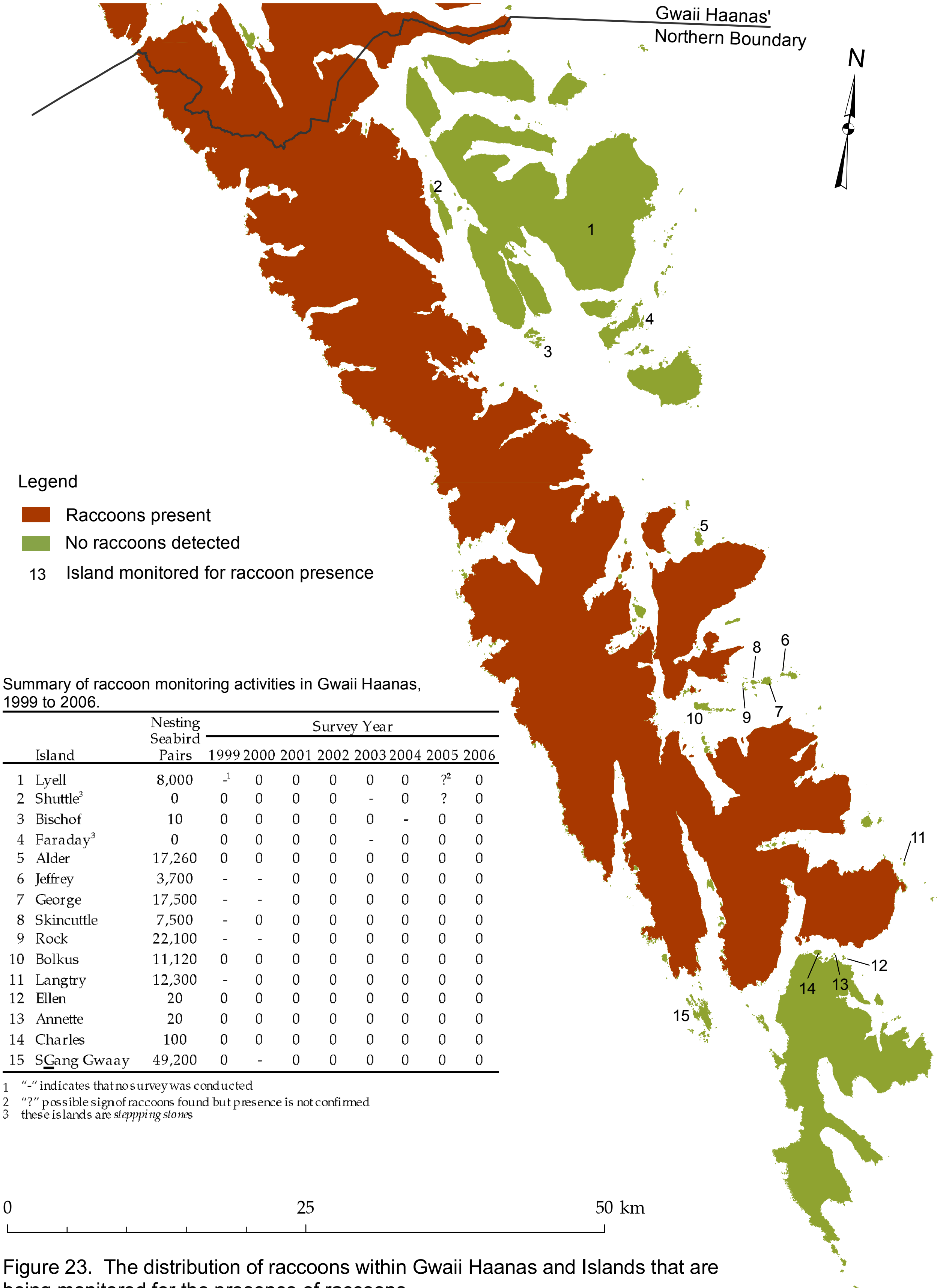
<u>2.5.8. Measure 8 - Invasive Plants</u>

<u>Monitoring Question</u>

Is the occurrence of invasive alien plants increasing at campsites in Gwaii Haanas?

<u>Context</u>

Of the 741 vascular plant species recorded on Haida Gwaii, 203 are not native to the islands (Cheney et al. 2007) and new species continue to colonize. Invasive alien plants are those non-native species that become established <u>and</u> expand rapidly across a new habitat. They



Summary of raccoon monitoring activities in Gwaii Haanas, 1999 to 2006.

	Island	Nesting Seabird Pairs	Survey Year							
			1999	2000	2001	2002	2003	2004	2005	2006
1	Lyell	8,000	-[1]	0	0	0	0	0	?[2]	0
2	Shuttle[3]	0	0	0	0	0	-	0	?	0
3	Bischof	10	0	0	0	0	0	-	0	0
4	Faraday[3]	0	0	0	0	0	-	0	0	0
5	Alder	17,260	0	0	0	0	0	0	0	0
6	Jeffrey	3,700	-	-	0	0	0	0	0	0
7	George	17,500	-	-	0	0	0	0	0	0
8	Skincuttle	7,500	-	0	0	0	0	0	0	0
9	Rock	22,100	-	-	0	0	0	0	0	0
10	Bolkus	11,120	0	0	0	0	0	0	0	0
11	Langtry	12,300	-	0	0	0	0	0	0	0
12	Ellen	20	0	0	0	0	0	0	0	0
13	Annette	20	0	0	0	0	0	0	0	0
14	Charles	100	0	0	0	0	0	0	0	0
15	SGang Gwaay	49,200	0	-	0	0	0	0	0	0

1 "-" indicates that no survey was conducted
2 "?" possible sign of raccoons found but presence is not confirmed
3 these islands are *stepppi ng stones*

Figure 23. The distribution of raccoons within Gwaii Haanas and Islands that are being monitored for the presence of raccoons.

threaten the EI of native ecosystems because they can compete with native species, often altering habitats and ecosystem functions. Most of these species are early successional plants, adapted to disturbed habitats. Consequently, many of these species are concentrated along the shoreline, where the constant disturbance caused by wind and wave action promotes their colonization and expansion (Myers and Bazely 2003).

Camping areas are scattered along the shoreline throughout Gwaii Haanas. Visitor activity at these sites adds to the natural disturbance caused by wind and waves, making these particularly likely areas for alien invasive plants to become established. Monitoring for invasive plants at these sites, therefore, serves as a shoreline bell-weather.

<u>Methods and Analysis</u>
The Gwaii Haanas campsite monitoring program assesses visitor effects at 65 commonly used camping areas throughout Gwaii Haanas (see Section 2.5.9). During these surveys, we record the occurrence (presence / absence) of select invasive alien plants at the camping areas. The frequency with which campsites are monitored varies depending upon the intensity of site use. Surveys are conducted annually at a few heavily used sites and at a minimum of once every four years. For our analyses, we assume the occurrence (presence/ absence) of a plant species at a campsite remains constant until it is resurveyed.

To date, the only invasive plants for which consistent data are available from campsite surveys are thistles (*Cirsium* spp.). Presence / absence of thistles has been recorded at 65 campsites surveyed from 1998 to 2006. During this period, there has been no significant trend in the proportion of campsites in Gwaii Haanas with thistles (GLM using arcsine transformed proportions: N = 9 years, R2 = 0.006, P = 0.84) (Figure 24). Power analysis indicates that we can detect an annual change of 1.47 % in the

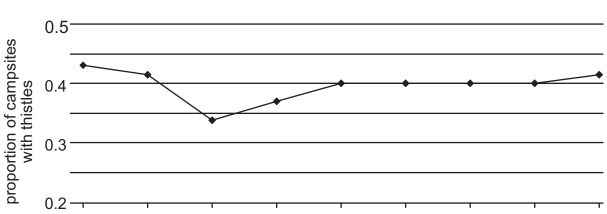


Figure 24. The proportion of campsites in Gwaii Haanas where thistles were detected, 1998 to 2006.

proportion of campsites with thistles, or a 7.35 % change over 5 years (a = 0.05, b = 0.20).

Thresholds have not yet been established for the occurrence of invasive plants at campsites. We are currently in the process of developing an invasive plant management strategy for Gwaii Haanas. We will use a non-native species risk assessment and targeted inventories to help us select additional priority invasive plants to be monitored and to set thresholds.

Numerous invasive plant species have been observed in coastal areas throughout Gwaii Haanas, including at camping areas. The status of invasive shoreline plants is “fair,” based on the occurrence of numerous invasive species throughout Gwaii Haanas. There is no evidence of them competitively excluding native species. To date, the only invasive plants for which consistent data are available for trend analysis are thistles (*Cirsium* spp.). Since 1998, the occurrence of thistles at campsites in Gwaii Haanas has remained “stable.”

2.5.9. Measure 9 - Visitor Effects at Campsites

<u>Monitoring Question</u>
Are human effects at campsites in Gwaii Haanas increasing?

<u>Context</u>
Gwaii Haanas has a “leave no trace” visitor camping policy. In fact, the vision outlined in the management plan includes each visitor sharing “the sensation of being the first person to set foot here” (AMB 2003 a). Visitor quotas have been put in place and there is very low tolerance for visitor impacts, with clearly defined thresholds that trigger management actions. To disperse crowds and minimize ecological damage, random camping is strongly encouraged and there are no designated campsites. However, the rugged coastline leaves only ~80 sites with accessible beaches that are conducive to camping. In addition, both independent and commercially-guided visitors often position their campsites for easy access to Watchmen sites. Consequently, a handful of popular campsites receive the majority of use. A campsite monitoring program has been established to ensure that management action can be taken before any of the camping areas deteriorate due to overuse. A small number of sites have been permanently closed to access and/or camping because of ecological, cultural or spiritual sensitivity (e.g., seabird breeding colonies, archaeological sites, burial places).

In 1996, Gwaii Haanas contracted researchers specializing in the effects of human use in protected areas to conduct an inventory of backcountry campsites in Gwaii Haanas and design a monitoring program (Marion and Farrell 1996). From 1996 to 1998, 77 campsites were identified in Gwaii Haanas and very intensive baseline data were collected to assess human use impacts associated with camping activities. Based on the results of the baseline study, monitoring protocols were refined and a long-term strategy was implemented in 2001. To increase survey efficiency, the number of variables recorded has since been reduced to include only those necessary to make informed management decisions and those required for long-term monitoring of visitor effects and other measures of ecological integrity (e.g., presence of invasive plants). Over the years, a few new campsites have been added to the survey and several sites that are in very close proximity have been combined. A total of 65 distinct campsites were used in our analyses (Figure 25).

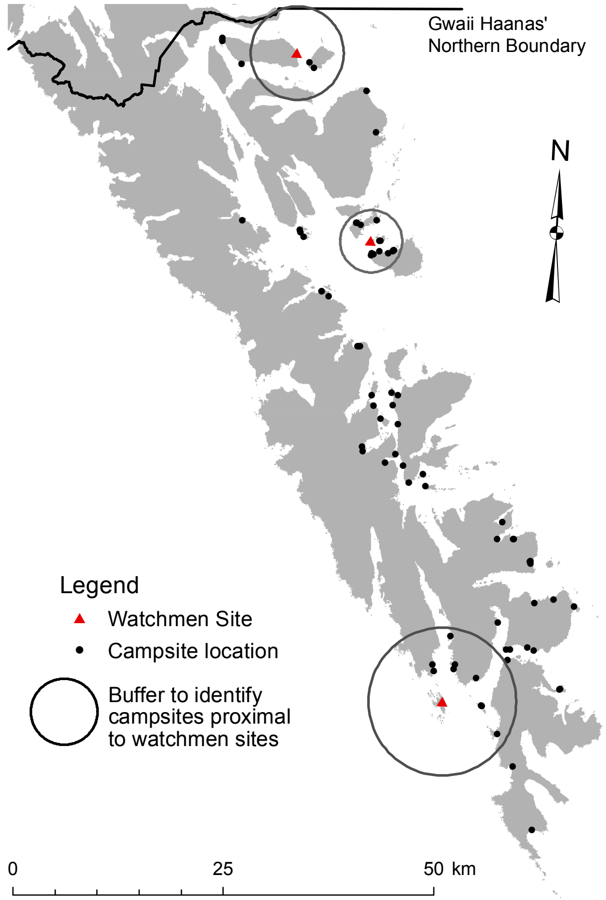


Figure 25. Campsites surveyed as part of the Gwaii Haanas campsite monitoring program. Sites in close proximity to Watchmen sites have different management thresholds.

Methods and Analysis

Annually In August, we survey some campsites as part of the Gwaii Haanas campsite monitoring program. The frequency with which campsites are monitored varies depending upon the intensity of use at the site. Surveys are conducted annually at a few heavily used sites and at a minimum of once every four years. The metrics used to assess human effects at campsites are: campsite condition class, length of developed trails, and extent of shoreline disturbance. Campsite condition class (CC) is based on the percent loss in vegetation cover, with each condition class representing a range (Table 33). Because many campsites are made up of a number of discrete tenting areas, a campsite's overall condition class is set at the maximum condition class reported for any use area (tent site) within the campsite. For our analyses, we make the assumption that metric values (condition class, trail length, length of shoreline disturbance) remain constant until they are resurveyed. Trends are assessed from 1998 (completion of the initial baseline assessment) until present.

The distribution of campsites within the different condition classes has been changing over time (Figure 26). There has been no significant trend (multinomial chi square using the midpoint as leverage: $X^2 = 21.18$, df = 24, P = 0.63). For this analysis, we pooled data in the upper and lower condition classes (0+1 and 4+5) because there is a bias introduced by having cells with low numbers (e.g. condition classes with few campsites in them on any given year). The lack of detectable trend may in part be due to the relatively low power of this test (effect size = 0.59, $\alpha = 0.05$, $\beta = 0.20$). Figure 26 reveals that over time, the number of sites in the lowest condition classes (0+1) has been steadily increasing, while the number of sites with moderate impact (CC = 3) has been decreasing. The number of sites in the highest condition classes (4+5) has remained relatively low and stable.

Table 33. Campsite condition class rating system for Gwaii Haanas.

Condition Class	Impact	% vegetation loss
0	non-existent	0
1	detected	<10
2	low	10-25
3	moderate	26-75
4	high	>75
5	extreme	>75 + exposed roots / erosion

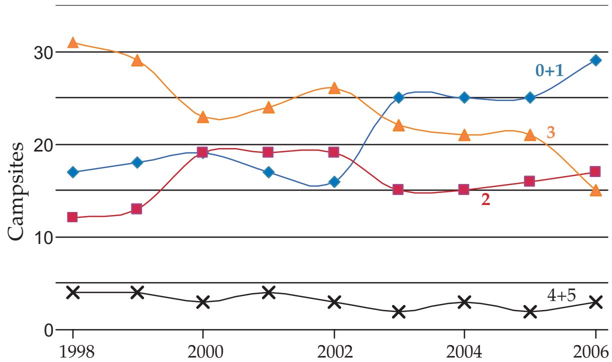


Figure 26. The number of campsites in each condition class between 1998 and 2006.

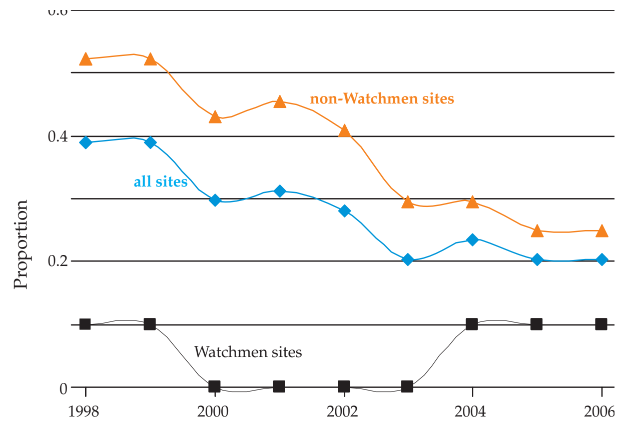


Figure 27. The proportion of campsites exceeding site-specific thresholds for condition class.

Site-specific thresholds, based on campsite condition class, have been set to trigger management action for campsites on an individual basis. Because Watchmen sites will remain popular destinations and the Gwaii Haanas vision supports continued visitor access to them, the AMB has accepted a higher level of visitor effects at campsites near Watchmen sites. For campsites in close proximity to Watchmen sites (N = 20), the site-specific threshold has been set at condition class 3. For the remainder of the campsites within Gwaii Haanas (N = 45), the site-specific threshold has been set at condition class 2. Results are reviewed annually by the AMB and site-specific management actions are considered at campsites exceeding these thresholds. The most common management approach is to "red flag" the sites as having effects requiring mediation. Tour operators are asked to voluntarily reduce use at these sites to allow recovery. To date, this approach has been effective. If a voluntary reduction does not result in recovery, then other management options (e.g., temporary or permanent closures, rehabilitation or hardening) may be invoked.

In 2006, the proportion of sites exceeding the site-specific threshold was 0.2 or 20%. The proportion of campsites exceeding thresholds has been decreasing since the surveys began (GLM using arcsine transformed proportions: N = 9 years, $R^2 = 0.88$, P = 0.0002) (Figure 27). This trend is most pronounced for sites that are away from Watchmen sites (R2 = 0.93, P<0.0001). For sites close to Watchmen sites, most sites have remained below the threshold, with only a couple of sites jumping back and forth across the threshold annually. Power analysis indicates that we can detect an annual change of 2.1% in the proportion of sites exceeding the management threshold, or10.6% over five years ($\alpha = 0.05$, $\beta = 0.20$).

Park-wide thresholds have also been set in the Backcountry Management Plan (AMB 2003 b). Excluding the camping areas in close proximity to Watchmen sites, no more than 20% of the campsites should have a condition class of 3 or greater. This forms the upper (yellow-red) threshold, with the lower (green-yellow) threshold tentatively set at 10%. In 2006, 24.4% of the camping areas (11 of 45) had a condition class of 3 or greater. Based on the Backcountry Management Plan threshold, the current status of visitor effects at campsites in Gwaii Haanas is "poor." This metric has been improving (a significant decline in the proportion of sites with CC of 3 or greater) since the surveys began (GLM using arcsine transformed proportions: N = 9 years, $R^2 = 0.93$, P<0.0001) (Figure 28). The total amount of shoreline disturbance has remained stable. Overall, the trend of this measure is "improving."

During each campsite survey, the length of human-developed trails and shoreline disturbance is measured. Each metric was summed across all campsites to calculate a total length of trail development and shoreline disturbance found each year at all the surveyed

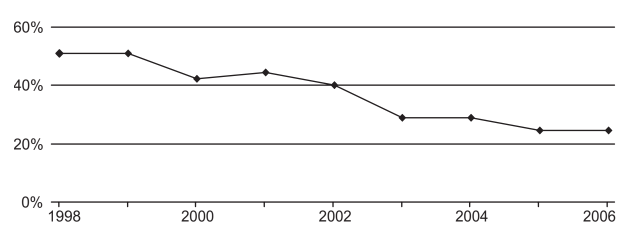


Figure 28. The percent of campsites (excluding those in close proximity to Watchmen sites) with a condition class of 3 or greater.

campsites throughout Gwaii Haanas. Since 1998, there has been a significant decline in the total length of trail development at campsites (GLM: N = 9 years, $R^2 = 0.65$, $P = 0.0083$) (Figure 29). The trend remains significant if the unusually high 1998 data is excluded from the analysis (GLM: N = 8 years, $R^2 = 0.83$, $P = 0.0015$). There has been no significant trend in the total amount of shoreline disturbance at campsite since 1998 (GLM: N = 9 years, $R^2 = 0.004$, $P = 0.87$) (Figure 30). Our power to detect trends using these variables is not very high. Power analysis indicates that we can only detect an annual change of 15.8% in total length of trails at campsites, or a 79% change over five years ($\alpha = 0.05$, $\beta = 0.20$; or an annual change of 7.3% if 1998 data are excluded from the analysis). For length of shoreline disturbance, power analysis indicates that we can only detect an annual change of 13.6%, or a 68% change over five years ($\alpha = 0.05$, $\beta = 0.20$). No thresholds have been established for these measures.

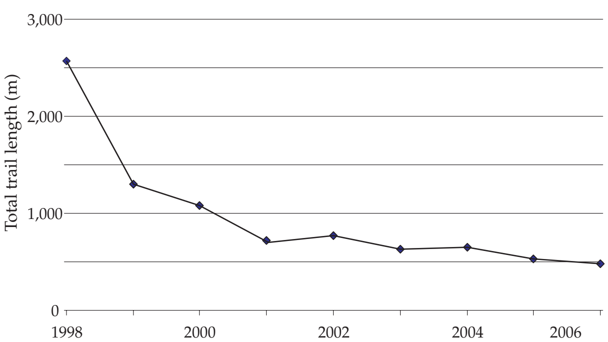


Figure 29. The total length of human-developed trails at campsites in Gwaii Haanas, 1998 to 2006.

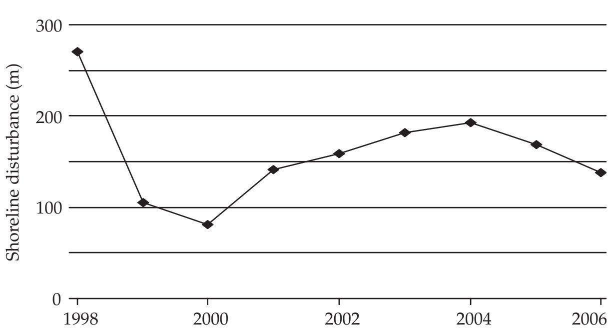


Figure 30. The total length of shoreline disturbance at campsites in Gwaii Haanas, 1998 to 2006.

2.6. MARINE (INTERTIDAL AND SUBTIDAL)

2.6.1. Measures 1 to 4 - Coastal Health Assessment Program (CHAP)

Monitoring Question
What is the status of marine EI using eelgrass ecosystems as a biosentinel?

Context
Seagrasses are perennial vascular plants, distantly related to land grasses such as those of sandy beach backshores and salt marshes. Unlike seaweeds (macroalgae), seagrasses have true roots, seeds, leaves and an internal tubular transport (vascular) system for nutrients and gasses. A key characteriatic of seagrasses is epiphyte (plant or animal / single- or multi-celled) growth on their leaves. Leaves provide a solid surface off the substrate and closer to light and water movement. Epiphytes are more numerous and diverse at the distal (older and eroding) leaf tips. Excessive epiphyte development covers leaves and impairs seagrass productivity.

Seagrass populations straddle the intertidal and subtidal in that they occur in the lower intertidal, down to a maximum depth of perhaps 5 to 10 m below the lowest tide line (Chart Datum). Their upper limit is determined by exposure to desiccation and wave energy in the intertidal, whereas their lower limit is imposed by the penetration of sufficient light in the subtidal to allow photosynthesis to exceed respiration.

The occurrence of seagrasses in nearshore waters positions them in the land-sea interface. This strategic location renders them a globally threatened marine ecosystem particularly vulnerable to effects of human activities such as habitat destruction, sedimentation or pollution (Orth et al. 2006). Changes in seagrass distribution, a reduction in their maximum depth limit, or their widespread loss signal important losses of ecosystem function. By monitoring these habitats, early detection of coastal environmental degradation can be made before irreparable loss occurs (Short et al. 2006). Further, seagrasses are useful in that they can respond rapidly to changing environmental conditions.

The major factors causing seagrass declines world-wide are excess nutrients (eutrophication) or sediments, both of which ultimately reduce the amount of light available to meadows. For example, anthropogenic (human-caused) eutrophication typically leads to large and persistent blooms of seaweeds that shade and eventually displace seagrass (Kentula and Dewit 2003). The establishment of relationships between light availability, water quality, and depth distributions of seagrass has provided a valuable tool for establishing habitat requirements for the species. The deepest water depth where seagrass grows is generally regarded as an indicator of coastal water quality because light availability at depth is reduced as eutrophication increases (Duffy 2006).

The competition between phytoplankton and epiphytes not only eliminates seagrasses but the resulting bare substrate supports a much lower diversity and abundance of fish (Deegan et al. 1997). Conversion of seagrass meadows into seaweed-dominated ecosystems is equivalent to habitat loss. Such replacement changes the structural complexity, food web dynamics, and chemical suitability. Anthropogenic nutrient enrichment causes a shift in primary producers and alters the fish and invertebrate communities and food webs (Deegan et al. 2002). Excessive seaweed growth interferes with seagrasses through competition for light or space.

Three seagrass species occur around Haida Gwaii (Sloan and Bartier 2000). "Eelgrass" (*Zostera marina*) roots in sheltered sediment (muddy to sandy) shores forming contiguous "meadows" that achieve their greatest width off estuarine deltas, in protected bays or in the heads of inlets. Eelgrass is found in the intertidal and shallow subtidal (+2 m to –5 m relative to Chart Datum). Eelgrass is tolerant of a wide range of salinities and temperatures, but generally flourishes in clear, low-nutrient (oligotrophic) and well-oxygenated waters. As well, there are two species of "surfgrass" (*Phyllospadix scouleri* and *P. torreyi*) that do not form large, contiguous meadows, but rather occur in patches or as bands along relatively more exposed and rocky shores that eelgrass. Only eelgrass meadows are monitored around Gwaii Haanas.

Eelgrass meadows provide a variety of ecological functions important for maintaining healthy coastal ecosystems. For example, eelgrass directly supports food chains through the secondary production of invertebrates associated with epiphytes. Meadows also indirectly support food chains through supplies of plant material to detrital pathways and adjacent ecosystems such as bare mudflats. Eelgrass provides a 3-dimensional rearing and foraging habitat for invertebrates, fishes and birds. Finally, eelgrasses reduce shoreline erosion from wave action, help stabilize sediments, and act as an integral component of the shallow water nutrient recycling process (Sloan 2006).

Eelgrass meadows are, therefore, protected as important fish habitat including "no net loss" policies for these ecosystems. Protection in Canada comes under the *Fisheries Act* (Section 34) in which the definition of "fish habitat" fits well with the ecosystem role of meadows. As well, meadows could also be "residences" (of listed species) warranting protection under the *Species at Risk Act*. Although eelgrass ecosystems are relatively small compared to other inshore ecosystem types, they are very important habitat for young fish such as salmon, many invertebrates and marine birds.

Over 21% of the Haida Gwaii coast is lined with eelgrass (Figure 31) that was first mapped in the early 1990s (Sloan 2006). Gwaii Haanas' wardens surveyed the distribution of eelgrass along all sheltered shorelines of Gwaii Haanas from June to September 1999 to 2002. The results of the warden survey are compared with the distribution of the first eelgrass survey (Figure 32). Both surveys confirm that meadows occur wherever there are sheltered sediment shores, with more sheltered habitat (and meadows) along the east coast compared to the relatively more exposed west coast. Further, the data table in Figure 32 shows reasonable congruence between the early data and the warden survey data. Two important points arising are: (1) as a field verification of this ecosystem type, the independent warden data overlap well with the original data in our GIS and (2) although the surveys were separated by about a decade, the meadows' appreciable overlap reflects the long-term persistence of eelgrass wherever suitable habitat (substrate and exposure) occurs.

<u>CHAP Methods</u>

The status of coastal ecosystem EI in national parks within Parks Canada's Pacific Bioregion is being evaluated using two indicators: "Intertidal" and "Subtidal" that Gwaii Haanas has rolled up into the single "Coastal" indicator. A key metric for our Coastal indicator is the Coastal Health Assessment Program (CHAP). The CHAP is being developed based on field sampling (initiated in 2004) in Gwaii Haanas, Pacific Rim (Barkley Sound and Clayoquot Sound) and the Gulf Islands national parks.

Eelgrass meadows are the ecosystem being used as the biological sentinel within the CHAP that consists of the following four measures:

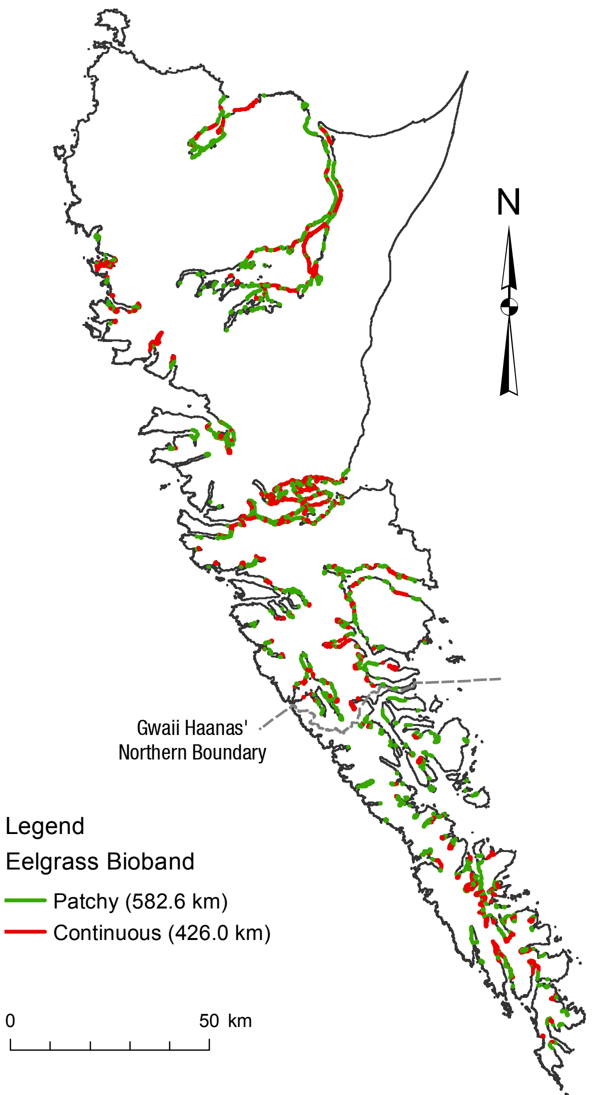


Figure 31. Distribution of the Eelgrass (*Zostera marina*) bioband around Haida Gwaii (from Sloan 2006).

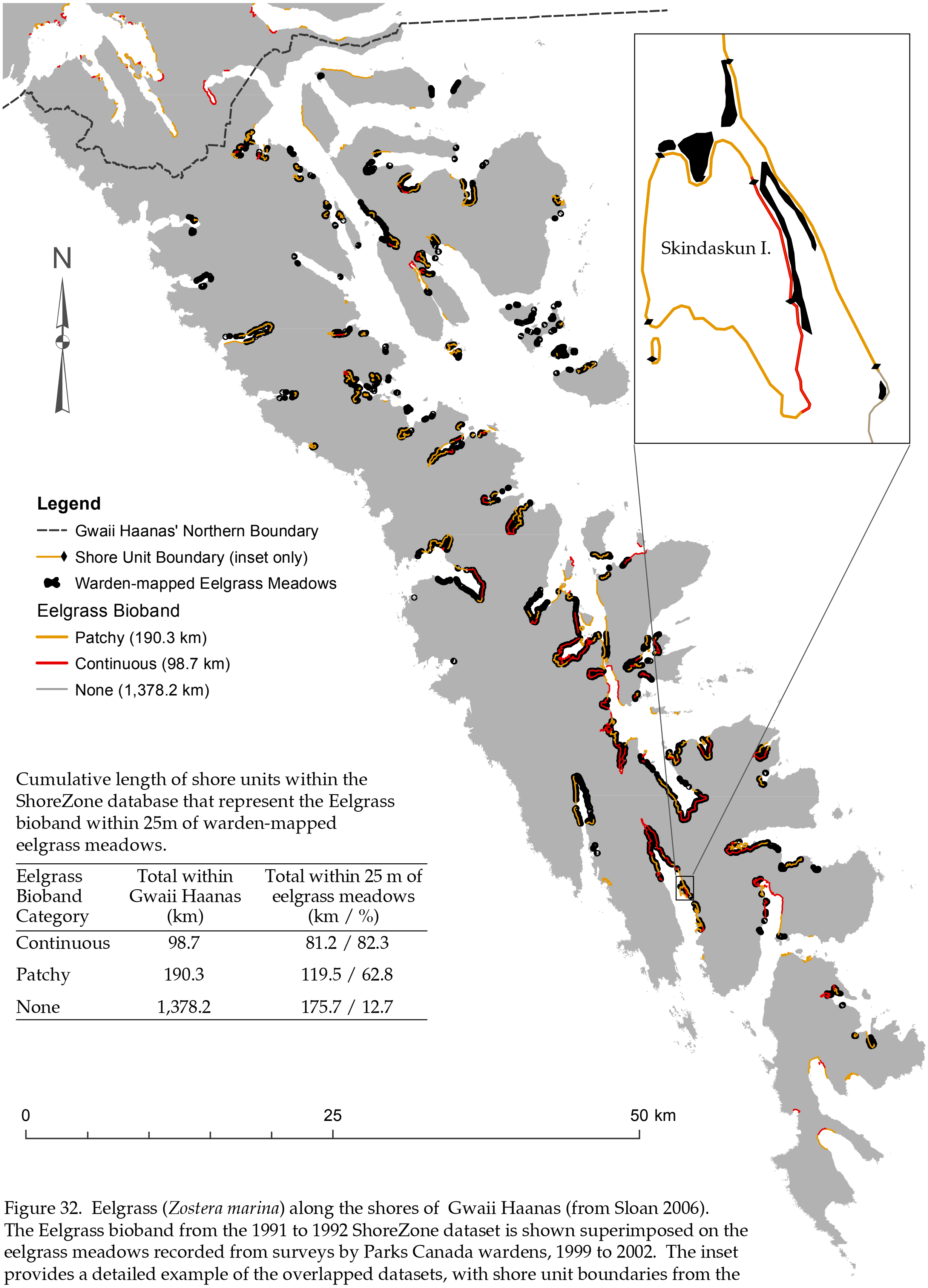
(1) Anthropogenic Disturbance Index; (2) Environmental Assessment; (3) Eelgrass Health Assessment; and (4) Fish Assemblage Assessment.

The two major ideas behind the CHAP are, firstly, that the CHAP approach considers multiple lines of converging evidence when evaluating coastal health and this evidence is from a bio-indicator, eelgrass ecosystems, because they are highly visible, easily sampled, and their properties respond relatively quickly to degradation. Secondly, the CHAP aims at assessing the health of several spatially separated eelgrass meadows, including their surrounding environmental properties and fish communities, within a narrow temporal window (i.e. a low tide cycle in mid July).

Thus, the focus is on comparing several meadows sampled within a region at the same time and comparing among years. It is not the intent of the monitoring program to follow the changes in one eelgrass meadow, because resources do not allow for sufficient sampling to meet parametric statistical approaches that require lofty assumptions about the data. Instead, we have followed the lead in the marine environmental assessment literature and use non-parametric multivariate analyses. This approach warrants further discussion about the concept of "power" - so prominent in Parks Canada's monitoring programs - that tends to focus on single species in a univariate way.

While the general concept of power is a reasonable consideration for non-parametric and multivariate cases, it is near impossible to carry over in any formal sense. Bob Clarke (founder of the PRIMER statistical package – Clarke and Gorley 2006) in a personal communication indicates that one has to specify the alternative hypothesis to the null hypothesis, and because this includes a vast number of possible ways in which the community can change, it is unrealistic to be able to specify this in the multivariate case. "Looking at a single species, if the assumption of normality is justified (which it never is) you may want to detect a 10% increase or decrease in the abundance of the species if you had information on the variability in abundance for that species over replicates. In the multispecies case, you not only have to assume joint normality (impossible) and be able to specify the variances of each species (near impossible) but you also have to say whether you want to detect an increase or decrease in all individual species by a certain amount (but which species will go up and which ones down?). It would be impossible to specify the alternative hypothesis that you would like to have good power to detect. In non-parametric multivariate approaches used in PRIMER, such as non metric multidimensional scaling or analysis of similarity, the experimental design should include "enough" replicates to generate sufficient permutations for comparing observed statistics (see below). In other words, although we cannot formally test for power, it makes good intuitive sense that more replicates increase the chances of ensuring conclusions from non parametric multivariate statistical approaches are meaningful.

Eelgrass meadows were sampled once annually, in mid July when juveniles and young-of-year fishes use meadows for rearing and foraging (Robinson et al. 2006). Triplicate beach seine sets were completed at each site using a 9.2 m long



Cumulative length of shore units within the ShoreZone database that represent the Eelgrass bioband within 25m of warden-mapped eelgrass meadows.

Eelgrass Bioband Category	Total within Gwaii Haanas (km)	Total within 25 m of eelgrass meadows (km / %)
Continuous	98.7	81.2 / 82.3
Patchy	190.3	119.5 / 62.8
None	1,378.2	175.7 / 12.7

Figure 32. Eelgrass (*Zostera marina*) along the shores of Gwaii Haanas (from Sloan 2006). The Eelgrass bioband from the 1991 to 1992 ShoreZone dataset is shown superimposed on the eelgrass meadows recorded from surveys by Parks Canada wardens, 1999 to 2002. The inset provides a detailed example of the overlapped datasets, with shore unit boundaries from the ShoreZone dataset, from around Skindaskun Island on the eastern shore of Louscoone Inlet.

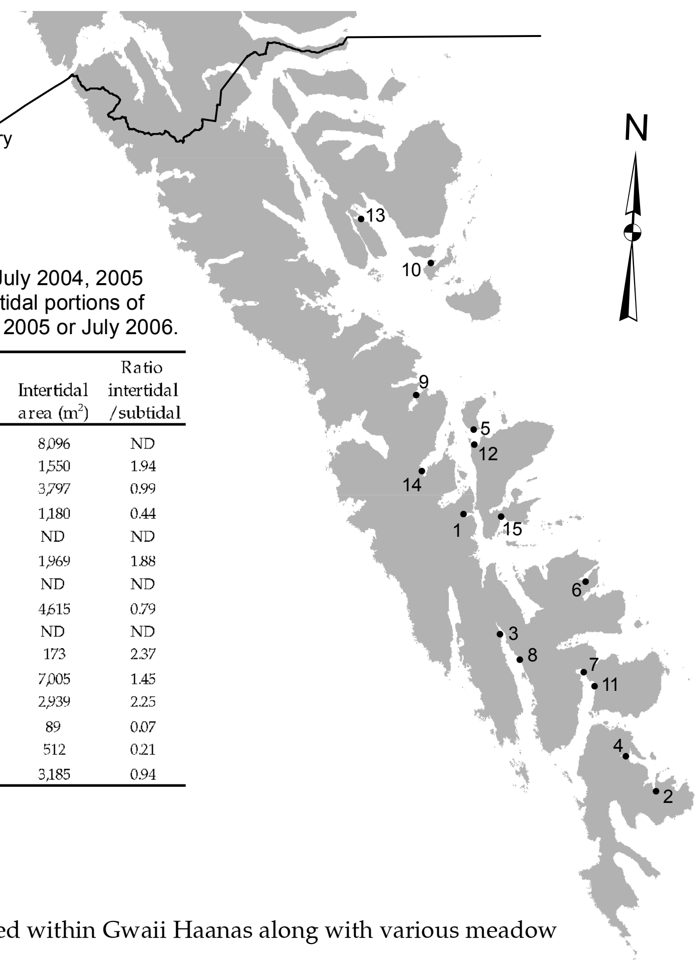
beach seine with 4 mm stretch mesh, having a 3.1 m drop in the centre and tapering to 1.1 m at the wings. Seining was conducted during a two-hour window before and after the early morning lowest low water (tidal height < 0.6 m). The beach seine was set in a round-haul manner, passing through meadows, from a small boat. After a seine, fish were removed and held in water-filled totes, and then the next seine was conducted about 5 to 10 m alongshore. After the third seine, all fish were identified to species, counted, and returned to the sea. The total area of each meadow sampled after 3 beach seines was ~150 m^2. The table in Figure 33 summarizes sampling in Gwaii Haanas' meadows during July 2004, 2005 and 2006. Year is an independent factor in the analyses because meadows are recolonized by new young-of-year fish annually.

Environmental information was collected for meadows sampled each year. Point measures of salinity, water temperature, nitrates and fluorescence (an indicator of chlorophyll) were collected after seining. In addition, above-ground eelgrass biomass samples were obtained from nine randomly placed 0.1 m^2 quadrats. Eelgrass samples were returned to the lab, scraped free of epiphytes and both dried for 24 hours at 60 ºC in a muffle furnace (see Robinson and Yakimishyn 2005 for all methods).

Results

The Anthropogenic Disturbance Index (ADI) is used to describe surrounding landscape or seascape disturbances that a single meadow may be subjected to. The ADI consists of five measures (Table 34), and each meadow is given a rank value for each measure based on local knowledge, field observations, nautical charts, topographic maps, and creel (sport fish) census data (when available) from DFO. Because these measures typically change little from year-to-year, the ADI for each meadow needs only to be updated every five years.

The ADI scores calculated for 16 meadows sampled in Gwaii Haanas averaged 6.5 out of a maximum possible score of 25 (Table 35). To appreciate how undisturbed Gwaii Haanas' meadows are, Figure 34 shows the median



Eelgrass meadows sampled in Gwaii Haanas during July 2004, 2005 and 2006. The surface areas of the subtidal and intertidal portions of each meadow were estimated once either during July 2005 or July 2006.

Site	Site Visits 2004	2005	2006	Meadow area (m^2)	Subtidal area (m^2)	Intertidal area (m^2)	Ratio intertidal /subtidal
1 Bag Harbour	X	X	X	8,096	ND[1]	8,096	ND
2 Balcom Inlet	X	X	X	2,347	797	1,550	1.94
3 Head of Louscoone Inlet	X	X	X	7,623	3,826	3,797	0.99
4 Heater Harbour	X	X	X	3,876	2,696	1,180	0.44
5 Huxley Camp			X	ND	ND	ND	ND
6 Ikeda Cove		X		3,014	1,045	1,969	1.88
7 Kendrick Point	X			ND	ND	ND	ND
8 Louscoone Inlet	X	X	X	10,425	5,810	4,615	0.79
9 Matheson Inlet			X	ND	ND	ND	ND
10 Murchison Island	X	X	X	246	73	173	2.37
11 Rose Inlet	X	X	X	11,827	4,822	7,005	1.45
12 Section Cove	X	X	X	4,243	1,304	2,939	2.25
13 Sedgwick Bay	X	X	X	1,379	1,291	89	0.07
14 Skaat Harbour		X		2,967	2,455	512	0.21
15 Swan Bay	X	X	X	6,560	3,376	3,185	0.94

1 ND = no data

Figure 33. Locations of eelgrass meadows sampled within Gwaii Haanas along with various meadow criteria, 2004 to 2006.

Table 34. Anthropogenic Disturbance Index (ADI) metrics, rank scores and descriptions used to calculate ADI scores for eelgrass meadows sampled in Gwaii Haanas.

Metric	Rank	Description
Land use	1	Forested, pristine area with no development
	3	Single lodging or campsite on shore
	5	More than one house or building on shoreline
Marine use	1	No structures or use
	3	Light anchorage or a single dock or a small boat ramp
	5	Marina, large dock or heavy anchorage
Boat traffic	1	Virtually no traffic, secluded
	3	Light boat use and traffic
	5	Heavy boat traffic, eelgrass bed adjacent to a navigation aid
Human accessibility	1	Isolated
	3	Easy access by boat
	5	Easy access by land/road
Regional fishery pressure	1	Low
	3	Moderate
	5	High

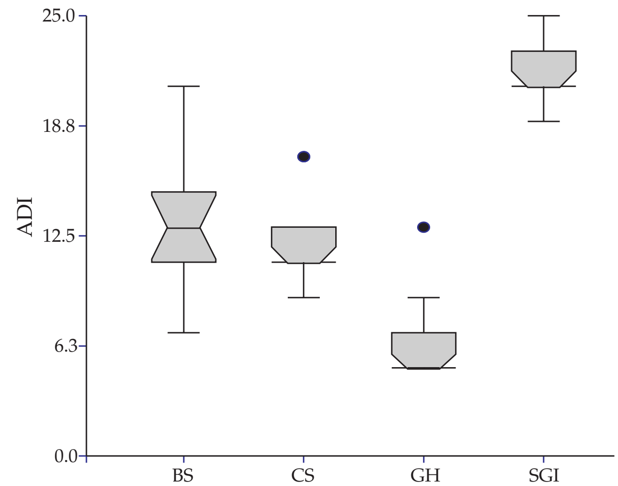


Figure 34. Box plots of Anthropogenic Disturbance Index (ADI) score distributions for four regions sampled in 2004 and 2005. The boxes represent 50% of all values (25th to 75th percentile) and the whiskers cover the range of points, while outliers are represented by dots. The four study regions are placed along a N-SE axis. The abbreviations are: from Pacific Rim (BS = Barkley Sound / CS = southern Clayoquot Sound); Gwaii Haanas (GH); Southern Gulf Islands (= SGI). Gwaii Haanas' meadows have significantly lower ADI scores than BS and CS, while SGI scores are significantly higher.

and 95% of ADI scores for the other Pacific national parks. Note that the southern Gulf Islands region have significantly higher total ADI scores than the other three regions and Gwaii Haanas sites have significantly lower ADI scores (Tukey-Kramer HSD test, $P < 0.05$). Gwaii Haanas meadows are under lower anthropogenic stress, due to their relative isolation from human settlements and coastal development.

Concerning Environmental Assessment, Thom et al. (2003) and Kentula and Dewitt (2003) indicate that interannual variations in eelgrass biomass show changes in system-carrying capacity. This is regulated by climate (e.g., broad regional factors such as precipitation), within-estuary factors (e.g., salinity gradient) and then within-site factors (e.g., environmental factors influenced by local human activities such as increased water column turbidity). Important environmental factors to eelgrass include precipitation (through increased runoff and associated turbidity and nutrient

Table 35. Anthropogenic Disturbance Index (ADI) scores for 16 eelgrass meadows sampled in Gwaii Haanas, July 2004, 2005, 2006.

Site (code)	Land use	Marine use	Boat traffic	Human access	Fishing pressure	Total ADI
Bag Harbour (BH)	1	1	1	3	1	**7**
Balcom Inlet (BI)	1	1	1	1	1	**5**
Head of Louscoone Inlet (HL)	1	1	1	1	1	**5**
Heater Harbour (HH)	1	3	1	1	1	**7**
Huxley Camp (HC)	3	3	3	3	1	**13**
Ikeda Cove (IK)	1	3	1	1	1	**7**
Kendrick Point (KP)	1	1	1	1	1	**5**
Louscoone Inlet (L)	1	1	1	1	1	**5**
Matheson Inlet (MI)	1	1	1	1	1	**5**
Murchison Island(MU)	1	1	1	1	1	**5**
Rose Inlet (RI)	1	3	3	1	1	**9**
Section Cove (SC)	1	1	1	3	1	**7**
Sedgwick Bay (SE)	1	3	3	1	1	**9**
Skaat Harbour (SK)	1	1	1	1	1	**5**
Swan Bay (SB)	1	1	1	1	1	**5**

loading), salinity and water temperature. Data are readily available to describe interannual variability in these three environmental factors at a regional scale. River discharge is often used as a surrogate for precipitation and was obtained from Environment Canada (http://scitech.pyr.ec.gc.ca/waterweb/formnav.asp) for the Yakoun River, Graham Island. Monthly sea surface temperature and salinity were obtained for Langara Island lighthouse from a DFO web site (http://www-sci.pac.dfo-mpo.gc.ca/osap/data/SearchTools/Searchlighthouse_e.htm).

Regional environmental data for the period 2004 through 2006 were compared to the long-term median and upper and lower quartile values expected for each month. If data from 2004 to 2006 were above or below the long-term quartiles, then an anomalous regional climate may be influencing observed local changes in environmental properties, eelgrass, and/or fish assemblage properties. For instance, high river discharge indicates excessive precipitation (i.e., above the upper quartile) and if it occurs during April to September it would be potentially detrimental to eelgrass growth because of a reduction in water clarity (increased turbidity due to runoff and phytoplankton blooms), an increase of land-derived nitrates (which may increase epiphyte or macroalgal blooms), and a reduction in salinity and increase in water temperature Increased temperatures and decreased salinities have been shown to correlated with decreased eelgrass biomass (Kentula and DeWitt 2003).

The top of Figure 35 shows the discharge for Yakoun River for 2004 and 2005 compared to the 41-year median discharge. Yakoun discharge (and hence precipitation) was at or below the lower quartile of the 41-year discharge record from April to August 2004, and mainly below the lower quartile or within the expected range for all months in 2005 except July. Daily river discharge for the Yakoun River in 2006 also is within the long-term precipitation regime (not shown).

The monthly sea surface temperature (SST) and sea surface salinity (SSS) data from Langara lighthouse indicate that the three sample years have been slightly (2 ºC) warmer and about one part per thousand (1 ppt) lower in salinity compared to the past 64 years (Figure 35). Note however, that the relatively high and stable sea surface salinity at Langara lighthouse indicates that oceanic conditions predominate. A lack of freshwater input during the summer growing months would be a positive environmental condition for eelgrass.

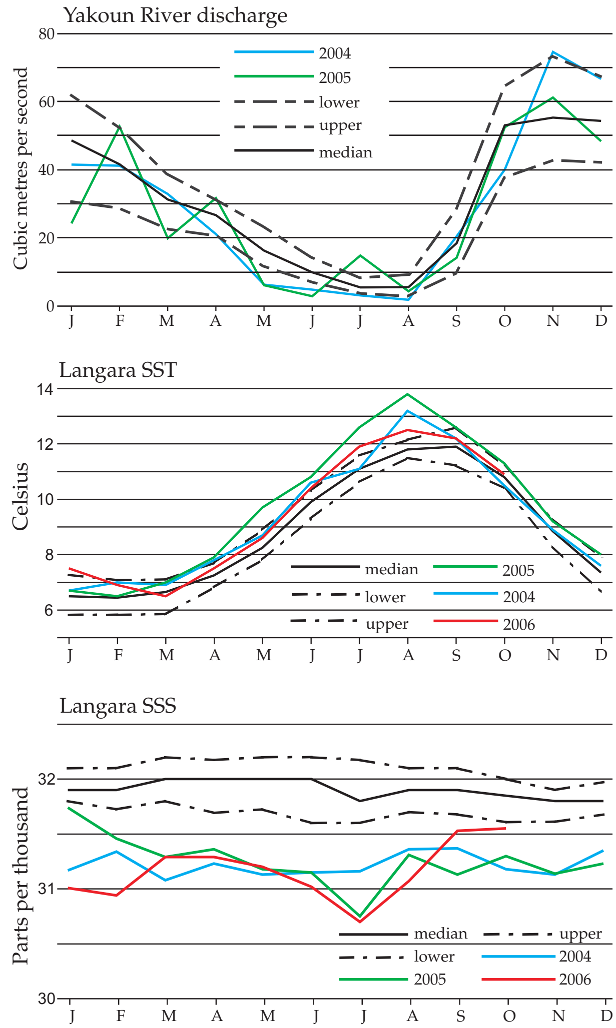


Figure 35. Comparison of monthly Yakoun River discharge and Langara lighthouse sea surface salinity (SSS) andsea surface temperature (SST) observed during 2004-2006 with the long term median and upper and lower quartile values.

Overall, environmental conditions in Gwaii Haanas are within the longer-term observations, and that there is relatively little interannual variation in local conditions. The latter is mainly due to low levels of local precipitation influencing nearshore marine areas - Gwaii Haanas is physically removed from the influence of large mainland rivers (e.g., Nass River).

The similarity in environmental conditions within meadows sampled among years was evaluated using box plots. Box plots were chosen because of their ability to visually represent the central tendancy of a range of data generated by replicate sampling. The box represents the interquartile range (IQR: 50% of all samples) and extends from the 25th to 75th percentile. The dark horizontal line within each box represents the median (50th percentile) value. If the median line is closer to the bottom of the box than the top, there is a tail toward larger values. The whiskers on the boxes represent the range of data, while extreme values are represented by dots above or below the whiskers (outliers). Notched boxes are used to make multiple comparisons among samples (Hintze 2001). Visually, if notches of two boxes do not overlap, we can assume that the medians are significantly different. The notches are calculated using: Median +/- 1.57*IQR/ $(n)^{0.5}$ - the 1.57 is selected for the 95% level of significance. To confirm visual differences between boxes, the differences among median values were tested using a Kruskal-Wallis one-way ANOVA on ranks (corrected for ties).

The box plots and Kruskal-Wallis tests indicate that salinities measured at eelgrass meadows were significantly higher in 2005 compared to 2004 and 2006 (Figure 36). However, values for all three environmental parameters measured indicate very small ranges of conditions, and hence relatively stable environments during mid-summer. Although the salinities are statistically higher, the range is small and oceanic in nature (i.e., good growing conditions for eelgrass).

The Eelgrass Health Assessment incorporates two measures reflecting the health of the intertidal portion of meadows, and two measures reflecting the subtidal portion of meadows. Health of the intertidal portion was assessed by measuring the epiphyte load and eelgrass biomass. The two intertidal metrics (eelgrass biomass and epiphyte load) were derived from nine 0.1 m^2 quadrat samples collected after fish sampling (Robinson and Yakimishyn 2005). Higher epiphyte load and or a low eelgrass biomass are indicative of poor overall health. Ultimately, the species of epiphyte should also be considered because a high biomass of certain epiphytes (e.g., *Smithora*)

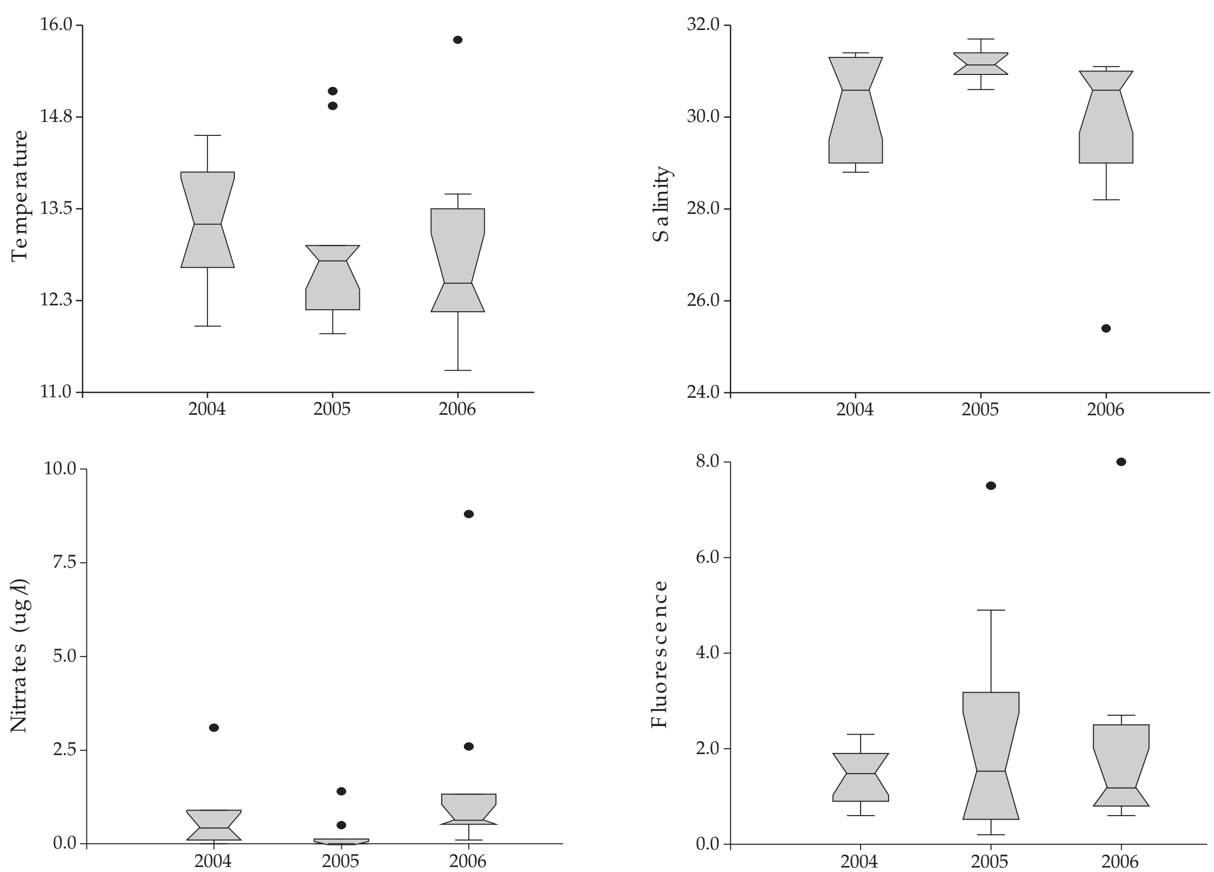


Figure 36. Box plots of sea surface temperature, salinity, nitrates, and fluorescence measured at several eelgrass beds (n in brackets) sampled during July of 2004 (11), 2005 (12) and 2006 (11). Note that one site in 2006 (Louscoone) exhibited extreme (outlier) values in all environmental properties. Also, note that the median salinity values observed in 2005 were significantly higher than 2004 or 2006, while nitrate levels were significantly lower in 2005 than 2004 or 2006. Both temperature and fluorescence values were not significantly different among years (Kruskal-Wallis One-Way ANOVA on ranks corrected for ties).

will likely be ecologically more beneficial to fishes than a high load of benthic diatoms.

Overall, Gwaii Haanas' meadows had significantly higher median eelgrass biomass in 2004 compared to 2005 and 2006 (Figure 37). Median epiphyte biomass and median percent epiphyte load on eelgrass was very low in 2004, but has been increasing at a rate of 7

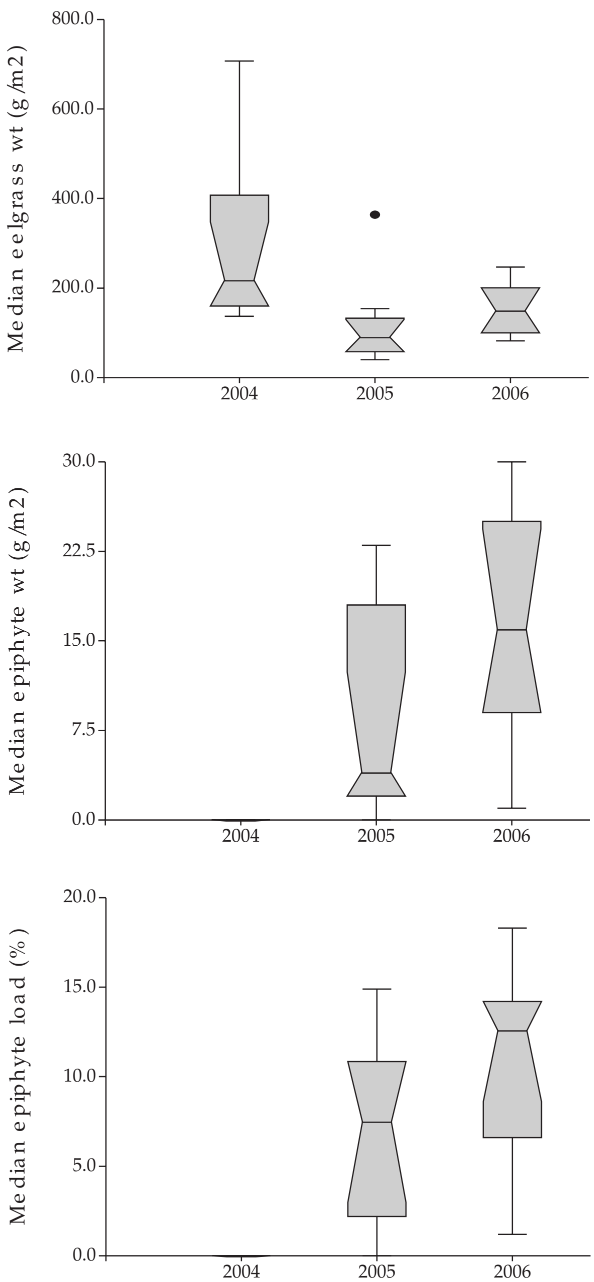


Figure 37. Median values and IQR of eelgrass biomass, epiphyte biomass, and percent epiphyte load (epiphyte biomass / eelgrass biomass X 100) for the same 8 eelgrass beds re-sampled in Gwaii Haanas during July 2004, 2005 and 2006.

to 8% annually. A likely explanation for this increase could be related to differences in cumulative monthly precipitation. Increased precipitation leads to higher loading of nitrogen from surrounding watersheds; nitrogen is considered limiting for algal populations in coastal waters. In 2004 at Sandspit airport, the cumulative precipitation from January to the end of June was 347 mm, from January to June 2005 it was 429 mm, and from January to June 2006 it was 510 mm. A greater cumulative precipitation through late winter, spring and early summer would result in higher nitrate concentrations available to epiphytes, and hence higher biomasses and subsequent epiphyte load percentages. In retrospect, the cumulative precipitation observed in 2004 was anomalously low, and hence epiphyte loads observed in 2005 and 2006 may be more representative of natural epiphyte conditions. Another consideration is that although epiphyte percent load has been increasing in each of the last two years, load percentages for Gwaii Haanas' meadows are at the lower end of values observed in the southern parks (Figure 38). Continued monitoring could establish the relationship between cumulative precipitation and epiphyte loads.

New in 2005 was a qualitative assessment of the health of subtidal portions of meadows. The two subtidal measures were derived from qualitative analyses of underwater video (wasting disease and subtidal epiphyte load). Robinson et al. (2006) describe the video method used. In general, wasting disease symptoms appear to be caused by the infection of a marine slime mould-like protest (*Labyrinthula zosterae*). *L. zosterae* can rapidly invade healthy blades, impairing photosynthesis. It is considered a primary pathogen causing the wasting disease infection. Some investigators have suggested that infection in eelgrass is linked to already-stressed eelgrass, and it is believed that healthy tissue can resist infection. Apparently, in the 1940s disease symptoms and eelgrass declines were reported from Washington and British Columbia. Criteria used for Eelgrass Health Assessment from underwater video recording are listed in Table 36.

Subtidal underwater video for 2006 is not yet analyzed. Analysis of 2005 video indicates that the average rank of subtidal epiphyte load was highest for Gulf Islands sites (4.8), and similar among the other regions (Gwaii Haanas, southern Clayoquot Sound, Barkley Sound: 2.7, 3.0, 2.6, respectively). The incidence of wasting disease was found to be highest in Barkley Sound (3.8), followed by Clayoquot Sound (1.8), Gwaii Haanas

(1.3) and Gulf Islands (1.0). The subtidal portion of Gwaii Haanas meadows in 2005 appear to be relatively healthy compared to the other regions.

There are several reasons doing Fish Assemblage Assessments, as follows:

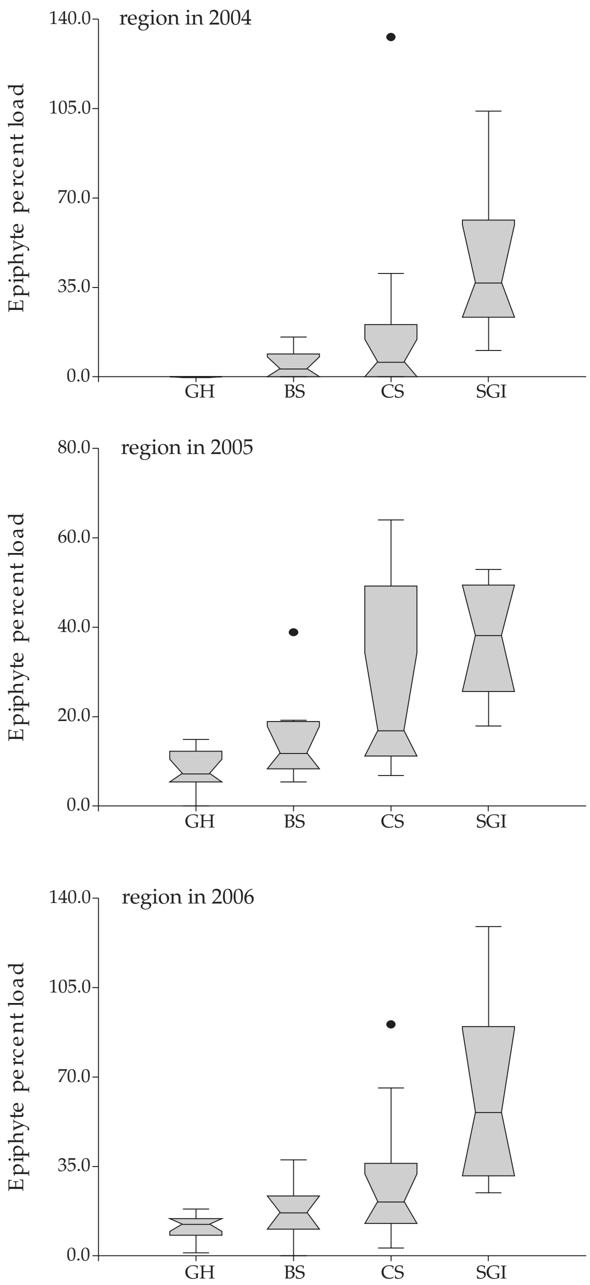


Figure 38. Comparison of the percent epiphyte load (epiphyte biomass divided by eelgrass biomass X 100) for eelgrass beds sampled during the summers of 2004, 2005, and 2006 in 4 regions of interest to Parks Canada. Abbreviations: GH: Gwaii Haanas, BS: Barkley Sound, CS: southern Clayoquot Sound, and SGI: Southern Gulf Islands.

Table 36. Criteria usedfor Eelgrass Health Assessment (wasting disease and ephiphyte load on eelgrass blades) from underwater video recording. Video enabled estimates of epiphyte load in subtidal portions of meadows not accessible at low tide.

Measure	Rank	Description
Subtidal epiphyte load	1	Less than 10% of blades with epiphytes
	3	Most blades have some epiphyte load, but no large mass
	5	Most blades have heavy epiphyte load: distinct mass of *Kornmannia*, diatoms or *Smithora*
Subtidal Wasting Disease	1	None
	3	From 5-10% to < 25% of blades affected
	5	More than 25% of blades affected

(1) young-of-the-year fishes are attracted to the 3-dimensional structure of eelgrass for protection from predators and for feeding opportunities, and thus it is relatively easy to quantitatively sample for fish in meadows compared to other habitat types;

(2) fish assemblage properties are known to change with changing meadow health (Deegan et al. 1997), e.g., as meadows deteriorate there is generally a reduction in the number and types of species, abundances, and a reduction of benthic and sensitive species; and

(3) changes in certain aspects of a fish assemblage found in eelgrass (e.g., number of juveniles of rockfishes, lingcod, greenlings) may also indicate changes in the health of fish populations in the region or changes in habitats other than eelgrass.

To assess for the EI of fish assemblages, three major aspects of fish community structure were evaluated over time: species similarity, dominance, and relatedness. This approach is more consistent with ecosystem level assessments compared to evaluating how single species may change over time. Non parametric multivariate approaches were used to assess for changes in fish assemblage structure as discussed below, and can be found in the PRIMER 6.0 software package (Clarke and Gorley 2006). Refer to the PRIMER web site for many published studies describing their methods.

Multivariate methods base comparisons of more than two samples on the extent to which the samples share particular species, at comparable levels of abundance. In this study, non metric multidimensional scaling (nMDS) is used because it makes few assumptions about the form of data or the interrelationships of the samples. The starting point for an nMDS is the generation of a similarity (or dissimilarity) matrix calculated between every pair of eelgrass sites. To assess changes in similarity in eelgrass fish assemblages in Gwaii Haanas we used the Bray-Curtis similarity coefficient. Abundances of each fish species (excluding juvenile seaperches) were square-root transformed to remove the influence of overly abundant species. The beauty of nMDS is that a plot of the configuration of samples is displayed in 2 or 3 dimensions, and that it satisfies all of the conditions imposed by the rank similarity matrix. The key to interpreting an nMDS plot is to understand that sites with the most similar species assemblage and abundance are closest together on the plot, while least similar sites are furthest apart. If the fish assemblages were different among years, one would expect a cluster of year 2004 sites, a cluster of year 2005 sites and a cluster of year 2006 sites. Furthermore, these clusters of samples would be widely separated on the plot. The adequacy of the sample representation in an MDS plot is evaluated using a stress value. A stress value < 0.05 gives an excellent representation of the relationships among samples. A stress of < 0.1 gives a good ordination with no real prospect of a misleading interpretation, while a stress of < 0.2 gives a potentially useful 2-d picture. Stress values > 0.3 indicate that points (samples) are close to being arbitrarily placed in the 2-d ordination space. Stress values between 0.2 to 0.3 should be treated with caution for a small number of points (< 50).

An analysis of similarity (ANOSIM) is a non-parametric permutation procedure that was also applied to the rank similarity matrix. ANOSIM allows for a test of the null hypothesis that there are no fish assemblage differences between eelgrass sites specified *a priori* by levels of a single factor or group (e.g., year). It is critical that group specification is made prior to seeing the data. The ANOSIM is not a valid test of differences between groups generated by cluster analysis or MDS (or other ordination methods). The ANOSIM test uses a generalized permutation and randomization approach for generating significance levels and a global R statistic (Clarke and Gorley 2006). The R statistic varies between 0 (no differences between groups) and 1 (complete discrimination between groups). The statistical significance of the R statistic indicates the percentage chance of obtaining an observed R-value compared to randomly generating r-values. For example, a p < 0.01 means a < 1 in 1,000 chance that the observed r came from 1,000 randomizations of r-values. The significance level (p) is very dependent upon the number of replicates in each group, and as with univariate statistics, biologically trivial differences can still be statistically significant when "power" is large. The r-values is not a function of the number of replicates but an absolute measure of differences between two groups in the high-dimensional space of the data, whereas p is always hijacked by the sample size (Clarke and Gorley 2006). Thus, the key is not to focus on the p values but on the Global R values; the higher the value of R the greater the separation of replicates from the groups. Global values > 0.5 are worth noting. In this baseline assessment of fish assemblage similarity we were interested in evaluating for interannual differences in fish assemblage structure. Hence, eelgrass sites were grouped a priori by year (2004 to 2006) and these groupings were used in an MDS and ANOSIM of fish species richness and abundance.

Figure 39 shows a 2-dimensional nMDS plot of meadows sampled in each of three years. The relatively high stress value (0.24) and mixing of samples of any one year with other years is indicative of no obvious clustering of fish diversity and abundance. This conclusion is consistent with the results of the analysis of similarity, indicating a very small global r (0.025) and a high p value (0.276; Table 37). Overall, we conclude that the fish assemblages are very similar from year-to-year in the Gwaii Haanas meadows.

How evenly individuals are distributed among the different species at an eelgrass site is called "evenness" (or its opposite, dominance). More diverse sites would have an equitable distribution of species contributing to total site abundance or biomass. In theory, as sites become disturbed they

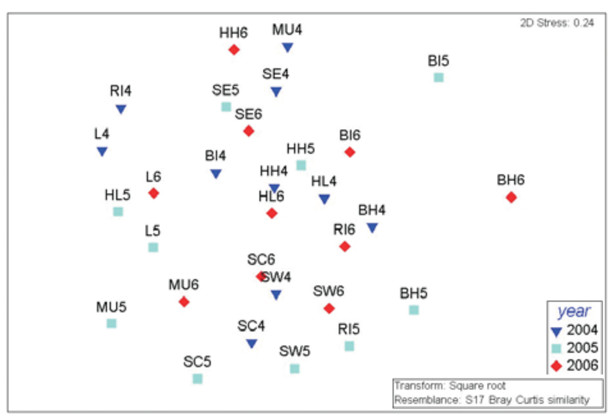


Figure 39. A plot of fish assemblage diversity and abundance sampled in eelgrass meadows in Gwaii Haanas during July 2004 to 2006. Fish assemblages are very similar over time (i.e., there is no clustering of sites by year).

Table 37. Results of an analysis of similarities for fish assemblage richness and abundance from Gwaii Haanas eelgrass meadows, July 2004, 2005, 2006.

Global Test
Sample statistic (Global R): 0.025
Significance level of sample statistic: 27.6%
Number of permutations: 999 (Random sample from a large number)
Number of permuted statistics greater than or equal to Global R: 275

	Pairwise Tests				
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
2004, 2005	0.025	30.2	92378	999	301
2004, 2006	0.025	34.5	92378	999	344
2005, 2006	0.032	27.1	92378	999	270

may become dominated by fewer (more tolerant) fish species. Hence, a change in dominance over time would be indicative of unwanted change to the fish assemblage. In this assessment, ranked species dominance curves are based on the rankings of species in decreasing order of their importance in terms of abundance or biomass (Clarke and Gorley 2006). Cumulative ranked abundance curves plotted against species rank are called k-dominance curves. To test for differences in k-dominance curves, single cumulative curves need to be compared across replicates, both within a year and between years. The distance apart of every pair of cumulative curves is computed using the Manhattan distance. The triangular matrix of dissimilarity values is then entered into an ANOSIM to produce a significance test for the differences between years (Clarke and Gorley 2006). Replicate k-dominance curves across years that tend to be further apart from each other than replicates within years will give a global r-value >0.

The k-dominance plot of abundance for meadows sampled in each year is shown in Figure 40. Each line represents the data from one meadow. An ANOSIM on the dissimilarity matrix indicates (Table 38) that there are no significant changes in dominance among the three years. The vast majority of sites in all years had abundance of species equitably distributed, and hence as considered to have low dominance and high diversity.

Many biodiversity studies focus only on species richness and evenness. Surprisingly few studies focus on the taxonomic relatedness of a group of species. For example, two sites may have the same number of species (10) but closer examination may reveal that the first site contains 10 species from the same family, while the second site contains two species from each of five different families. Obviously, the second site would be of higher conservation value for biodiversity representivity. Further, a loss or reduction in species relatedness at a site would be a cause for further investigation. One statistic that has recently been developed and widely applied in marine biodiversity studies is the average taxonomic distinctness (avTD; Clarke and Gorley 2006). AvTD has been shown to be independent of the number of species in a sample, and it is based on the taxonomic distance through the Linnean classification tree between every pair of species (Clarke and Gorley 2006). The avTD of a site is the average taxonomic distance apart of all

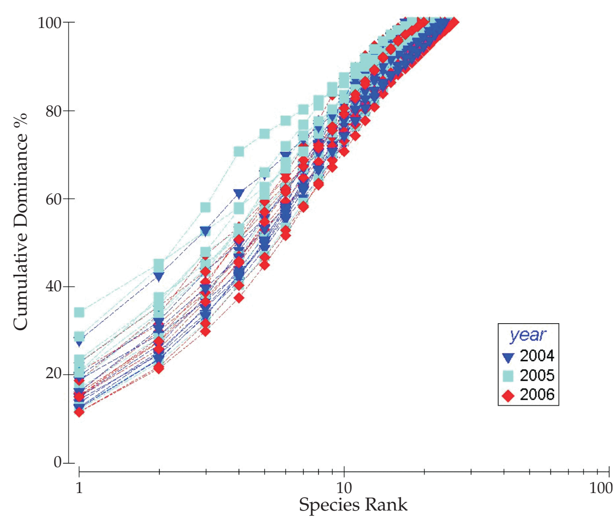


Figure 40. A plot of k-dominance curves for eelgrass beds sampled in Gwaii Haanas during July, 2004-2006. Each curve represents one eelgrass bed sampled in a particular year. Flatter curves indicate that the fish assemblage is dominated in abundance by fewer species. Dominance does not differ significantly among years.

Table 38. Results of an analysis of similarities in dominance of fish assemblages from Gwaii Haanas eelgrass meadows, July 2004, 2005, 2006.

Global Test

Sample statistic (Global R): -0.021
Significance level of sample statistic: 61.5%
Number of permutations: 999 (Random sample from a large number)
Number of permuted statistics greater than or equal to Global R: 614

	Pairwise Tests				
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
2004, 2005	0.032	25.0	92378	999	249
2004, 2006	-0.069	90.7	92378	999	906
2005, 2006	-0.024	54.7	92378	999	546

its pairs of species. Thus, the statistic summarizes the average taxonomic breadth of a site.

The assessment tool used here has the null hypothesis that a species list from one eelgrass site has the same taxonomic structure as the regional species list from which it is drawn. A randomization process is used to compare the observed site avTD with "expected" regionally derived avTDs. For example, if there are s species observed at a site, make repeated drawings at random of s species from the regional species list and compute the avTD for each drawing. Typically, 1,000 random drawings of s species are made from the regional list, and a 95% probability range of derived (expected) avTD values is constructed (Clarke and Gorley 2006). The observed value of avTD from a site can now be compared to the randomly generated distribution of avTDs. Values below the lower probability limit (5%) suggest that the biodiversity at that site is below expectation for the region. Closer examination of the sites' species list will reveal the cause of the reduced taxonomy.

The plot of avTD (Figure 41) shows that all sites in all years contained the expected taxonomic relatedness when compared to the regional species pool. Only Murchison eelgrass meadow in 2005 is considered marginal in its taxonomic distinctness (on the border of the funnel plot) and this site is deficient in plated species (missing three-spine sticklebacks and tube snouts) and in seaperches (shiners and striped seaperches). Murchison was also the most variable site sampled in Gwaii Haanas, most likely because of its small size and patchiness (Figure 33). Murchison was within the expected range in 2006, and hence the 2005 result may have been a sampling artefact.

In summary, when information from all assessment aspects are combined in Table 39, Gwaii Haanas' eelgrass meadows' status is "good" and the trend "stable." The only cautionary note is the 15% year-over-year increase in epiphyte percent load on intertidal portions of the eelgrass meadows. Overall, the assessments represent a good baseline of information for the ecological integrity of coastal ecosystems, as represented by the sentinel, eelgrass. It remains to be determined if fish assemblages and eelgrass properties observed in Gwaii Haanas can be used as reference values for meadows 600 km to the south in Pacific Rim and Gulf Islands National Park Reserves.

2.6.2. Measure 5 - Spawning Pacific Herring

Monitoring Question

What are the trends in estimated annual herring spawn biomass and distribution along Gwaii Haanas' coast?

Context

Pacific herring (*Clupea pallasii*) is important around Haida Gwaii culturally and economically. Islanders have always been active in this nearshore subsistence and commercial fishery (Sloan 2006). Herring is ecologically important as a "forage" species eaten by a wide range of fish, bird and mammal predators. Thus, herring transfers energy from lower trophic levels of their plankton food to the higher trophic levels of their many predators.

Given that the Haida Gwaii stock is managed by DFO as separate from all other stocks coast-wide, and that stocks are in decline, herring has become symbolic for local concerns over fisheries management and marine ecosystem health. The present era of stock decline with

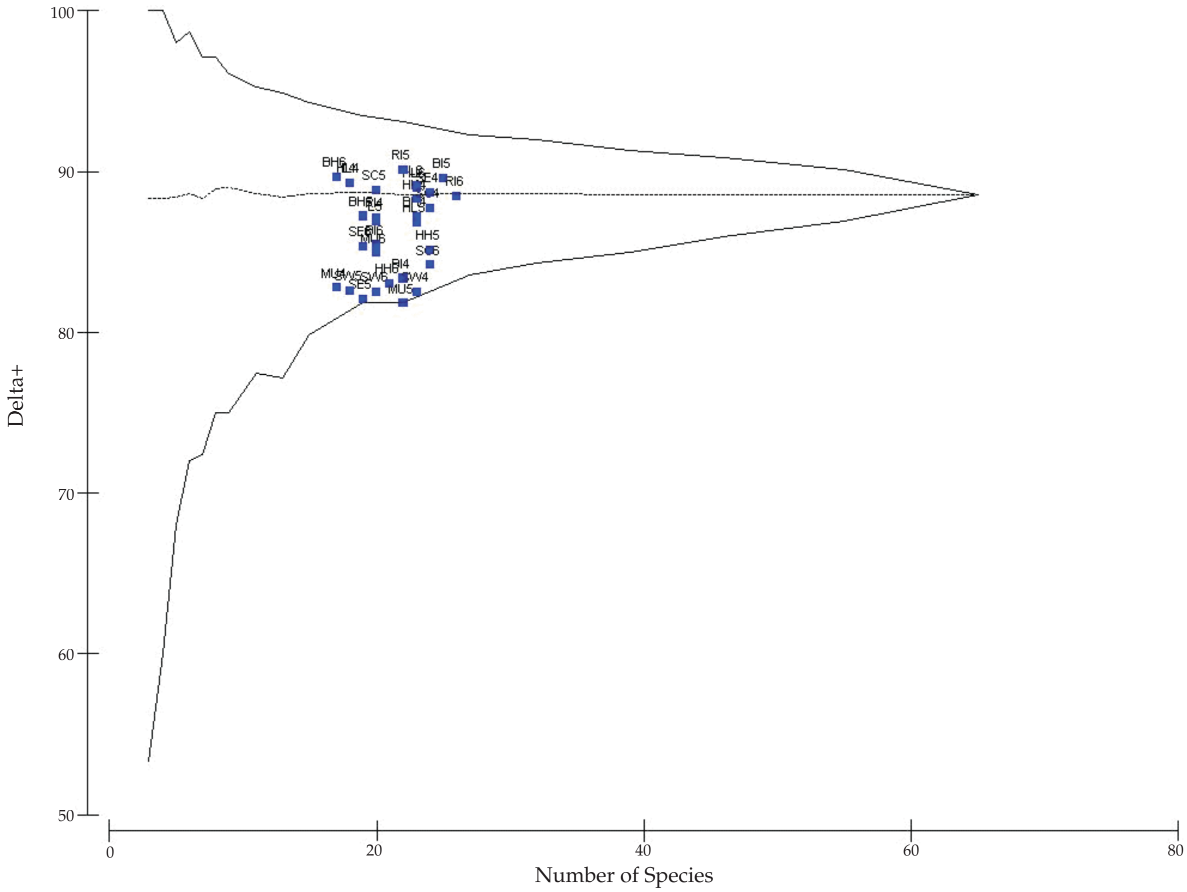


Figure 41. Funnel plot of taxonomic distinctness scores for eelgrass beds sampled in Gwaii Haanas during July of 2004-2006. All sites fall within the funnel and thus have an expected taxonomy based on the regional species list.

consistently low recruitment, for reasons that are poorly understood, heightens concerns in the local community over sustainability.

Pacific herring rarely exceeding 25 cm in length and occur in large schools in inshore and offshore waters. Locally, herring mature and spawn by age three with most living six years within their potential 12-year life span. Migrations to inshore spawning grounds occur during October to December from offshore feeding grounds in Hecate Strait. Herring spawn in huge aggregations in late winter to early spring between about 1.5 m above to perhaps 18 m depth below chart datum on rock, but mostly on attached plants. Inshore waters can turn white with the "milt" (sperm) of the millions of male spawners trying to fertilize the deposited

Table 39. Summary of the status of the seven measures of the CHAP metric for Gwaii Haanas, 2004, 2005, 2006. Eelgrass meadows are considered very healthy and relatively unchanged in the past three years. The only exception is the year-over-year increase in epiphyte load for intertidal portions of meadows. 2006 was the second year of increase and is marked as yellow. Subtidal video was not collected in 2004, and results were not available for 2006.

Year	Anthropogenic Disturbance Index	Environmental Assessment		Eelgrass Health Assessment		Fish Assemblage Assessment			Overall
		Regional	Local	Intertdial	Subtidal	Similarity	Dominance	Relatedness	
2004					ND[1]				
2005									
2006					ND				

1 ND = no data
Green = "*good*" status / Yellow = "*fair*" status

eggs. The local spawning season ranges from February to July with a March-to-April peak.

Herring spawning is one of the most dramatic coastal events, providing an enormous pulse of biomass and energy into nearshore food webs and attracting many predators to gorge on the spawners and their eggs (Willson and Womble 2006). Given that spawning is related to depth, substrate and vegetation characteristics, this is a valuable shoreline reference dataset. As well, the spawn time series data are used as a foundation for herring stock assessment.

Key context for herring is that their populations fluctuate widely. Further, factors affecting recruitment to commercial stocks remain unpredictable and poorly understood. Recruitment is thought to rely upon the size of the parent stock and environmental conditions (themselves highly variable annually) during the first year of life, such as sea surface temperature and salinity in the stock area. In summary, forecasting the potential catch taken sustainably from stocks requires an assessment of current abundance and determination of the factors affecting their dynamics.

The history of the Haida Gwaii fishery is recounted in Sloan (2006). The commercial herring fishery dates from the late 1930s with intermittent landings until the large "reduction" fishery from 1951 to 1965 that ended in stock collapse and fishery closure in 1968. The reduction era, during which huge catches were rendered into low-value fishmeal and oil, saw some dramatic takes (e.g., 77,500 tonnes in 1956), but also some annual landings of <1,000 tonnes. The inherent instability of stocks has, therefore, long been known. The modern era of limited-entry (fixed number of licenses) local herring fisheries essentially began in the 1970s. This was for the higher-value products of "roe" (ovaries removed from the body cavity and salted) or spawn-on-kelp (laid eggs attached to kelp fronds that are trimmed and brined) - both exported to Asia.

The local fishery is managed on-island by DFO in consultation with DFO's Science Branch, the Haida Fisheries Program and industry. For example, the Haida Fisheries Program cooperates with DFO annually to complete boat-based and diving surveys of spawn distribution and abundance. Fishery activities and management decisions are documented annually by DFO's on-island Resource Management Coordinator in the form of the Record of Management Strategies (RMS). Since 1994, the same individual has completed the RMSs that comprise the detailed local history of this fishery.

Within the commercial allocation, spawn-on-kelp has precedence over roe because of the former's lower biological effects on stocks. The areas from which herring stocks are assessed are shown in Figure 42. It is critical to note that this "major" stock assessment area includes all of Gwaii Haanas' east coast and some of its southwest coast. Also, there are numerous "minor" stock areas, particularly along the west coast. Between Port Louis and Englefield Bay, for example, there may be eight small spawning groups.

For roe, fish are either seined or gill-netted. In 2002, the most recent year of a roe fishery, only

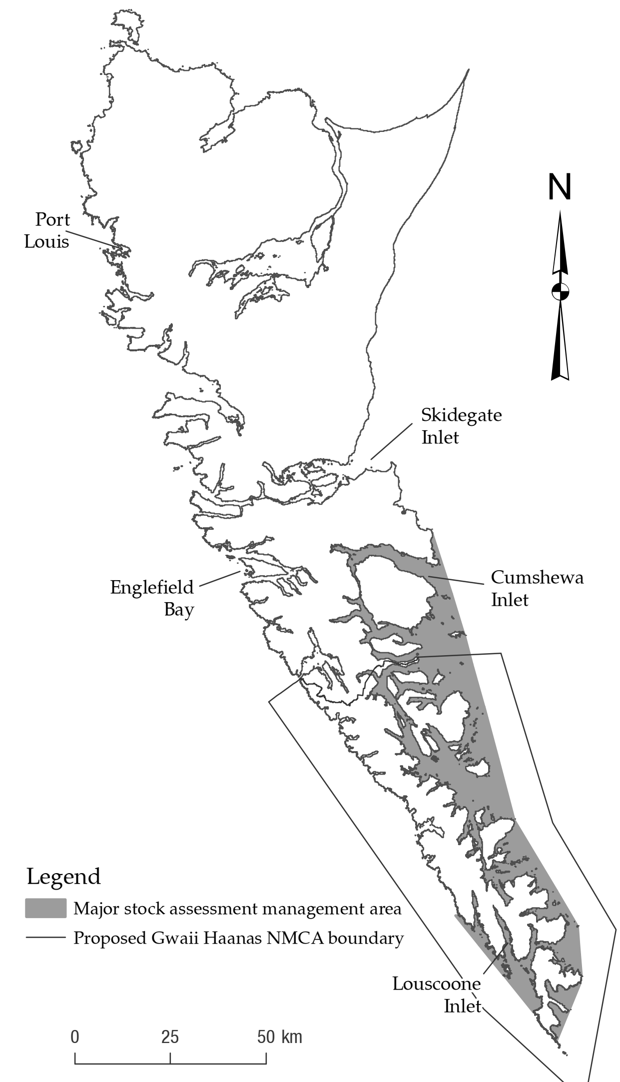


Figure 42. The region of Haida Gwaii within which Fisheries and Oceans Canada assesses the "major" Pacific herring stocks. "Minor" herring stock areas are along the west coast such as Port Louis or Englefield Bay.

about 700 tonnes were landed. Stocks have been so low that, from 1994 through 2007, there were ten years with no roe fishery at all.

The British Columbia spawn-on-kelp fishery originated within the proposed Gwaii Haanas NMCAR in 1971 - full commercial production by 1975. Kelp is the substrate for egg attachment for the commercial product and traditional Haida food (k'aaw). Giant kelp (*Macrocystis integrifolia*) is the dominant species used and is mostly gathered within the proposed NMCAR along the east coast of Moresby Island. Kelp is either deployed in pens, into which captured herring are placed ("closed-ponding") and then released, or suspended from float lines set in active spawning areas ("open-ponding"). Closed-ponding has been the main method used.

The March-April spawn-on-kelp fishery has always had a high proportion of Haida participation. Until 2003, the relatively high value spawn-on-kelp product has been worth about $2 million annually. Until 2004, the fishery occurred mostly along the SE coast of Moresby Island. In 2005, declining stocks along the east coast led to an industry shift to the west coast. In 2006 and 2007, all commercial herring fishing was closed due to low stocks. This closure has led to much local concern for the long-term well-being of herring.

<u>Results</u>
The core of herring monitoring by DFO is spawn mapping - recording spawn deposition according to intensity, space and time to underpin population ("stock") status assessments. Spawn mapping began in 1928 in southern British Columbia and the database of spawn frequency and magnitude per unit shore length is one of the longest annual time-series of marine population data coast-wide [http://www.pac.dfo-mpo.gc.ca/sci/herring/herspawn/pages/default5_e.htm]. For each of about 100 geographical sections of the British Columbia coastline, time-series maps were constructed to delineate annual spawn deposition along each kilometer of shoreline from 1930 to 2001, as described in detail in Hay et al. (1989 – updated 2006). Also mapped is cumulative spawn deposition in multi-coloured plots that rank and classify each km of spawn habitat according to the long-term frequency and magnitude of spawning. In this coast-wide analysis, therefore, any km can be compared with any other km. Some 5,260 km (18%) of the 29.500 km British Columbia coastline has been so ranked and classified. In a typical year, 450 to 600 km (1.8% of the shoreline) is used by spawners coast-wide.

The linear extent of shoreline along which herring spawning has historically occurred in Gwaii Haanas is shown in Figure 43. Generally, spawning starts earlier in southern Haida Gwaii and occurs later going northward as the season progresses. An example is the relatively late (May) spawn often occurring in Skidegate Inlet. This typifies small area-stocks (with distinct biological characteristics) around the archipelago. Genetic analyses have revealed a distinct population in Skidegate Inlet compared to the southeast coast of Moresby Island. No stock sub-structure was revealed from sample locations along the west coast, although Louscoone Inlet herring "may be" distinct from herring of the rest of the east coast management unit (Figure 42). Differences in timing of spawning have

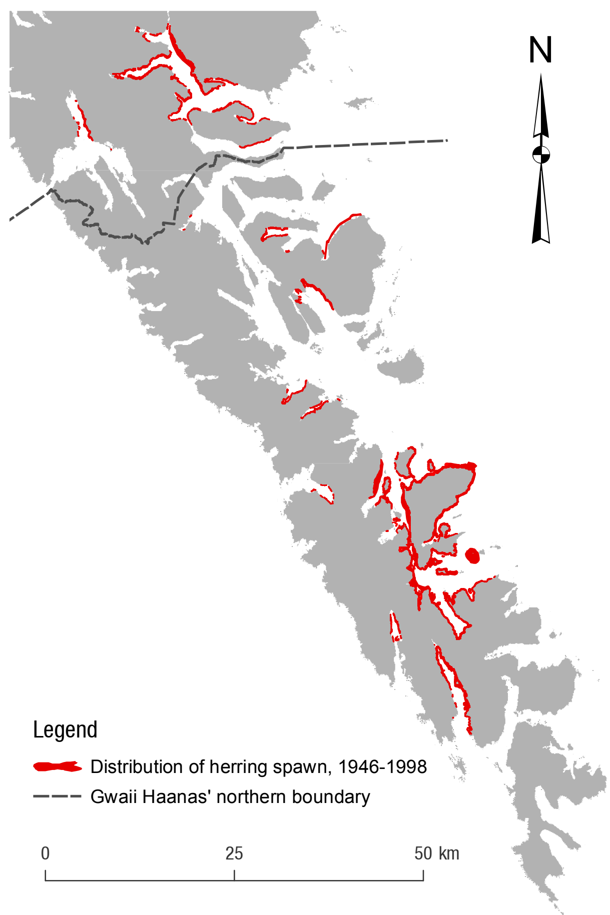


Figure 43. Distribution of Pacific herring spawn deposition (http://www.pac.dfo-mpo.gc.ca/sci/herring/herspawn/pages/default6_e.htm). The map includes overlapped data from 1946 to 1998. The distribution shows the approximate linear extent and breadth of spawning areas, as not all sections of coastline have been surveyed.

been suggested as the main isolating mechanism in areas where genetically distinct populations occur. It was concluded overall, however, that the weak genetic differentiation among herring stocks coast-wide coincides with appreciable straying rates between stocks sufficient to homogenize genetic traits coast-wide. For example, herring tagged on the central mainland coast have been collected from the west coast of Haida Gwaii.

Within the "major" stock area of Haida Gwaii, assessment data are gathered that including biological sampling for age and weight-at-age composition, historical spatial catch data and diver and surface surveys for distribution and intensity of egg deposition. As part of the annual cycle of local data gathering, a vessel makes sonar soundings and nets samples within the stock assessment area to estimate fish size frequencies for the age-structure model for stock size - early in the season (usually March). Also, there are spawn biomass estimates from annual surface and dive surveys that evaluate egg deposition on marine plants as an estimate of escapement (i.e., numbers of spawners). The spawn habitat index rates the shoreline as to its importance in terms of long-term production. The index derived is a function of the number of layers, the extent, and the frequency of spawn.

An assessment of current abundance is obtained using the revised age structure model for the "major" stock area (V. Fradette, DFO, personal communication). The estimate of spawning escapement is adjusted for survival and growth as well as recruitment to forecast abundance for the next year. The revised age-structured model uses the available 55-year time series of catch, spawn, weight-at-age, and age structure information. Forecasts of stock abundance are now calculated in two ways: (1) The number of fish at age prior to the fisheries are the numbers estimated at the beginning of the season multiplied by survival for the first period and the estimated availability at age. Recruitment is based on the survival and availability of the age 1+ fish estimated for the previous season. This recruitment is added to the estimated returning adults to project total abundance. (2) Recruitment is also calculated for three scenarios based on estimated numbers-at-age 2+ for the 1951 to 2005 time series. Poor, average, and good recruitment levels are calculated as the mean of the lowest 33%, the mid 33%, and the highest 33% of the estimated historic age 2+ abundance. These three recruitment estimates are then added to the projected adult biomass to provide abundance forecasts.

Central to DFO's management is setting a fixed annual quota for the "major" stock area based on a take of 20% (or less) of the forecast mature stock biomass (determined from previous years' spawn assessments). For Haida Gwaii, the "equilibrium" herring stock biomass (i.e., the undisturbed, unfished stock) is set at 42,800 tonnes for the "major" stock area. As well, there must be a minimum spawning stock biomass ("cutoff") of 25% (10,700 tonnes for the "major" stock area) <u>before</u> any biomass is available to industry that year. Think of this cutoff is a current <u>minimum</u> herring population threshold. The 25% rate is based on an analysis of stock dynamics, which indicate the level at which both catch and spawning biomass will stabilize while foregoing minimum yields over the long-term. For "minor" stock areas, such as along the west coast, a more conservative annual take is set at 10% of the forecast mature stock biomass.

Monitoring herring spawn within the proposed NMCA would be an important technical issue connecting Parks Canada, DFO, the Haida Fisheries Program and industry. Opportunities exist for collaborating towards more ecosystem-based fishery approaches with attendant social learning through a new and innovative management partnership. Although power analysis has not been done on this monitoring dataset, the fact that stocks are low with the commercial fishery closed for the last two years indicates that stock condition is "poor." The trend is pitched as "stable," with perhaps an indication of stock rebuilding (V. Fradette, DFO, personal communication).

2.7. PARK-WIDE

<u>2.7.1. Measure 1 - Species at Risk</u>

<u>Monitoring Question</u>
The question is not yet determined.

<u>Context</u>
In December 2002, the *Species at Risk Act* (SARA) was passed by Parliament (in force 2004) with a goal to protect species at risk and their habitats in Canada. Under SARA, a wildlife species is defined as "a species, subspecies, variety or geographically or genetically distinct population of animal, plant or other organism…" (Canada 2002). For a wildlife species to receive protection under SARA, it must first be assessed and classified by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as *extirpated, endangered, threatened*, or *of special*

concern (species classified as extinct, data deficient or not at risk are not covered by SARA) and then added to Schedule 1 of SARA (the legal list), a decision made by the Government in Council. Once a species is on the legal list, it is afforded some immediate protection and recovery planning is set into motion.

Seven wildlife species that inhabit Gwaii Haanas are on the legal list as extirpated, endangered, threatened or of special concern (Table 40). An additional three species are listed by COSEWIC, but have not yet been legally listed. This includes only those species that rely on lands within Gwaii Haanas at some time during their life cycle. Of the species known to inhabit the surrounding ocean within the proposed Gwaii Haanas NMCAR, an additional fifteen are legally listed under SARA, with one more listed by COSEWIC and undergoing public consultation for legal listing.

Of the seven legally listed species that rely on the lands within Gwaii Haanas, four have a recovery strategy, action plan or management plan completed or in progress. Recovery implementation activities are not currently underway within Gwaii Haanas for any of these species.

Methods and Analysis

The Pacific Bioregion monitoring group is planning to develop administrative metrics for species at risk that will track the change in status and recovery activity for listed species that inhabit the national parks on the Pacific coast.

Table 40. Species at risk found in Gwaii Haanas that are listed as endangered, threatened or special concern by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), as of January 2007. Those species that rely on the lands within Gwaii Haanas at some time during their life cycle are in bold. Species that do not rely on the lands of Gwaii Haanas but are found within the proposed NMCAR are in normal font. An asterisk (*) indicates those species for which a recovery strategy, action plan or management plan is complete or in progress.

Common Name	Scientific Name	COSEWIC Designation[1]	SARA Status[2]
*Blue Whale, Pacific population	*Balaenoptera musculus*	Endangered	Endangered
*Sei Whale, Pacific population	*Balaenoptera borealis*	Endangered	Endangered
*Leatherback Seaturtle	*Dermochelys coriacea*	Endangered	Endangered
*North Pacific Right Whale	*Eubalaena japonica*	Endangered	Endangered
***Ermine haidarum subspecies**	*Mustela erminea haidarum*	Threatened	Threatened
*Sea Otter	*Enhydra lutris*	Threatened	Threatened
Humpback Whale, North Pacific population	*Megaptera novaeangliae*	Threatened	Threatened
*Killer Whale, Northern resident population	*Orcinus orca*	Threatened	Threatened
Killer Whale, NE Pacific transient population	*Orcinus orca*	Threatened	Threatened
*Fin Whale, Pacific population	*Balaenoptera physalus*	Threatened	Special Concern
*Short-tailed Albatross	*Phoebastria albatrus*	Threatened	Threatened
***Marbled Murrelet**	*Brachyramphus marmoratus*	Threatened	Threatened
***Northern Goshawk laingi subspecies**	*Accipiter gentilis laingi*	Threatened	Threatened
*Pink-footed Shearwater	*Puffinus creatopus*	Threatened	Threatened
Northern Saw-whet Owl brooksi subspecies	*Aegolius acadicus brooksi*	Threatened	
Bocaccio Rockfish	*Sebastes paucispinis*	Threatened	
*Northern Abalone	*Haliotis kamtschatkana*	Threatened	Threatened
Steller Sea Lion	*Eumetopias jubatus*	Special Concern	Special Concern
Killer Whale, NE Pacific offshore population	*Orcinus orca*	Special Concern	Special Concern
Grey Whale, Eastern North Pacific population	*Eschrichtius robustus*	Special Concern	Special Concern
Harbour Porpoise, Pacific Ocean population	*Phocoena phocoena*	Special Concern	Special Concern
Ancient Murrelet	*Synthliboramphus antiquus*	Special Concern	Special Concern
*** Peregrine Falcon pealei subspecies**	*Falco peregrinus pealei*	Special Concern	Special Concern
Short-eared Owl	*Asio flammeus*	Special Concern	Special Concern[3]
Great Blue Heron fannini subspecies	*Ardea herodias fannini*	Special Concern	Special Concern[3]
Western Toad	*Bufo boreas*	Special Concern	Special Concern

1 COSEWIC produces a national list of wildlife species at risk, which classifies species into categories based on level of risk. The risk categories seen here are:
Endangered - facing imminent extirpation or extinction
Threatened - likely to become endangered if limiting factors are not reversed
Special Concern - has characteristics that make it particularly sensitive to human activities or natural events

2 For a species to be afforded protection under the Species at Risk Act (SARA), it must be listed on Schedule 1. Unless otherwise noted, the SARA Status is as it appears on Schedule 1. Schedule 2 and 3 species are awaiting a regulatory amendment to be added to Schedule 1.

3 Listed on Schedule 3 of SARA

2.7.2. Measure 2 - Non-native Mammals

Monitoring Question
Are the distributions of introduced species and their effects on ecosystems changing?

Context
As an isolated archipelago, Haida Gwaii is well known for its distinct flora and fauna evolved through over 10,000 years of isolation (Cowan 1989). Although biodiversity on the archipelago is lower than the adjacent mainland, a high proportion of island species are either endemic or disjunct. Most are uniquely adapted to function in ecosystems where competition is reduced and many of the mainland's top predators are absent. This renders local species vulnerable to introduced species.

Next to habitat destruction and fragmentation, introduced species are considered the most important global threat to biodiversity (Vitousek et al.1997). A new arrival is often freed from the predators or parasites that could normally control it - allowing the newcomer to thrive and disrupt/displace native species.

Ten mammals, 280 birds, one amphibian and 538 vascular plants are native to Haida Gwaii (Hamel and Hearne 2001; Burles et al. 2004; Cheney et al. 2007). Since European contact (late 1700s), at least 205 alien species have been introduced either intentionally or accidentally. Most are vascular plants (see previous Sections 2.1.2., 2.2.2., 2.3.2.), but 10 mammals, three birds and two amphibians have also been introduced. A further five domestic mammals have established feral populations. Of the introduced wild mammals, Sitka black-tailed deer, Norway and black rats, raccoon and red squirrel have established populations within Gwaii Haanas. Goat, rabbit and cat were formerly present, but are no longer. The remainder have been slowly expanding their ranges, but have not yet become established within Gwaii Haanas.

Sitka black-tailed deer were introduced in the late 1800s to early 1900s. Because of the absence of predators or competitors, coupled with the area's mild winter conditions and abundant food sources, deer have spread throughout the archipelago and have reached densities as high as 30 per km^2 on some offshore islands (Gaston et al. 2007 a,b and Sections 2.1.3., 2.2.3., 2.3.3.). Deer have colonized most islands within Gwaii Haanas with the exception of a few small offshore islands and small rocky islets (Table 41).

Deer pose a significant and ongoing threat to Gwaii Haanas' EI and the native ecosystems of Haida Gwaii. While early references often commented on Haida Gwaii's thick understory (e.g., Dawson 1880), over-browsing has since drastically depleted the shrub and herb layers of forests, and prevented the regeneration of new plants. This loss of structure is, in turn, affecting native fauna such as ermine and forest birds that are dependent on understory for protection, foraging and nesting.

Because of the significance of these effects, Gwaii Haanas developed a deer management strategy (Johnston 2006 a). This strategy recognizes that, although it is not logistically feasible to completely eliminate deer at present, a realistic goal is "protecting representatives of each major ecosystem type found in Gwaii Haanas from the effects of deer browsing in order to maintain or restore examples of the native biodiversity and ecosystem processes." Our approach has been to build a series of deer exclosures in each of the major ecosystem types and monitor vegetation changes. The metrics and protocols related to this approach are detailed previously in Sections 2.1.1., 2.2.1. and 2.3.1.

The second approach to restoring representative ecosystems is to prevent deer from reaching islands that are currently deer-free, and to cull deer from other islands. Islands within Gwaii

Table 41. Islands within Gwaii Haanas that are either deer-free, currently being culled, or have been designated as a high priority for culling (from Johnston 2006).

Island and Size (ha)	Status
Kerouard - north (15)	deer-free
Kerouard - south (10)	deer-free
Gordon (38)	high
SGang Gwaay (189)	maintain cull
Langtry (4)	deer-free
Rankine (25)	deer-free
Slug (1)	deer-free
Copper group (100)	high
Bolkus (82)	high
Howay (25)	deer-free
All Alone Stone (1)	deer-free
Sivart (8)	high
Ramsay (1,623)	high
Hotspring (17)	maintain cull
House (33)	maintain cull
Kawas-Agglomerate (41)	high
Tar Island group (23)	deer-free
Tuft Islet (3)	deer-free
Lost (10)	deer-free
Skaga (2)	deer-free

Haanas were evaluated based on their size, ease of access, proximity to other islands, ecosystem type and presence of other introduced species and ranked according to their priority for deer removal. Based on this analysis, 10 islands that are currently deer free, and three islands on which deer densities are being kept low by periodic culls were identified for monitoring and culling (Table 41). Seven other islands were identified as high priority candidates for possible future culls.

Two species of rat have been introduced to Haida Gwaii since European contact. The black or ship rat (*Rattus rattus*) is from the early sailing ships, likely the 1800s (Golumbia et al. 2002). During the early- to mid-1900s, black rats were introduced to some islands within Gwaii Haanas, largely by ships or industrial float camps. Recently, the Norway rat (*R. norvegicus*) was introduced. Being a larger and more aggressive, these displace black rats where their ranges overlap. Norway rats also drastically alter ecosystems on islands that they colonize.

Rats are omnivorous and eat a wide variety of plants and animals material. On Langara Island for instance, Norway rats ate nesting seabirds and their eggs, and were having a significant impact on nesting success (Harfenist 1994). Of the seven species of seabirds that once nested there in large numbers, only three species continued to nest there after the introduction of rats, and an Ancient Murrelet colony that once numbered >200,000 pairs (Gaston 1992) had been reduced to ~15,000 pairs. Similarly, within Gwaii Haanas, an Ancient Murrelet colony at Dodge Point, Lyell Island of >10,000 pairs in 1982 had been reduced to <8,000 pairs by 1992 (Lemon 1993).

Norway rats on Langara Island ate at least 34 different food items, including berries, seeds, plant shoots, fungi, terrestrial and marine invertebrates, fish and seabirds (Drever and Harestad 1998). They were also suspected of preying on songbird eggs, and perhaps responsible for the extirpation of Keen's Mice and a reduction in Dusky Shrews (Harfenist 1994).

Within Gwaii Haanas, rats have been found on 14 islands with an industrial history, or are near traditional anchorages, as well as in pockets on Moresby Island (Table 42). Rats have recently been eradicated from one of these islands (St. James) and plans are currently underway to eradicate rats from Bischof and Ellen Islands. While rats are not strong swimmers (Taylor 1993), they can swim up to 400 m (Russell et al. 2005). Some vessels visiting Gwaii Haanas may also harbour rats, although an active program exists to inform vessel owners of this concern and of pest control. The risk remains, however, that more islands could become infested.

Since their release in the early 1940s, raccoons have flourished and spread throughout the archipelago, including at least five islands other than Moresby Island (Table 42). When they have colonized islands that support seabirds, their impact is known to be devastating (Gaston and Lawrence 1993; Hartman et al. 1997). Little is known on what effects raccoons are having on other native species. Raccoons feed primarily on intertidal organisms (Hartman 1993), but predation on western toads in known in both Naikoon Provincial Park and Gwaii Haanas, as has predation on Red-throated Loon, Sandhill Crane, and Canada Goose nests (Reimchen 1992). These observations are largely anecdotal and the scale of these impacts is unknown.

Table 42. Distribution of rats, raccoon and red squirrel according to islandwithin Gwaii Haanas.

Island and Size (ha)	Rats	Red Squirrel	Raccoon
Arichka (10)	X[1]	–[2]	–
Bischoffs (66)	X	X	–
Boulder (6)	ND[3]	X	X
Burnaby (6,465)	X	X	X
de la Beche (27)	ND	X	ND
Ellen (20)	X	–	–
Faraday (316)	X	–	–
Harriet (6)	?[4]	X	ND
Haswell (13)	ND	ND	X
Hoskins (4)	ND	X	ND
Huxley (614)	X	X	?
Kat (71)	ND	X	ND
Kunga (450)	X	–	ND
Kunghit (12,330)	X	–	–
Lyell (17,301)	X	X	?
Marco (28)	ND	X	ND
Moresby (261,074)	X	X	X
Murchison (425)	X	–	–
Nakons (3)	ND	X	ND
Nest (2)	ND	X	ND
Richardson (945)	ND	X	ND
Ross (21)	ND	–	X
Shuttle (241)	X	X	?
Sea Pigeon (5)	ND	–	X
Section (12)	–	X	X
St. James (19)	X	–	–
Swan (30)	X	X	X
Tanu (2,193)	X	–	–
Topping-SE (3)	ND	X	ND
Wanderer (87)	–	X	?

1 X = known to be present
2 – = not known to be present
3 ND = no data
4 ? = presence suspected but not confirmed

Red squirrel (*Tamiascurius hudsonicus*) was first introduced to Haida Gwaii in 1950 to facilitate the collection of spruce cones and seed for silviculture purposes. With the help of a number of relocations facilitated by the British Columbia Forest Service to islands such as Lyell, Talunkwan and Limestone, they have quickly spread throughout the archipelago (Golumbia et al. 2002). Squirrels are now found throughout Moresby Island, as well as on 18 other islands within Gwaii Haanas (Table 42).

Red squirrel is having a significant effect on Gwaii Haanas ecosystems. Haida ermine, which were relatively abundant pre-squirrel, are now extremely rare. Their decline has been linked to increased predation and/or competition from marten, whose densities increased greatly after the introduction of squirrel prey (Edie 2001). Squirrels are also effective predators on songbird nests. They can have an added advantage in deer-browsed areas because bird nests are more vulnerable. This combined effect of deer browse and squirrel predation are linked to the decline of songbird populations (Martin et al. 2001).

<u>Results</u>

The EI metrics for non-native mammals are not established. As five introduced mammals do occur within Gwaii Haanas, the status of this measure is "poor." Although historical data are limited, it is known that once introduced, all of the above species have thrived and spread. For example, deer have spread to almost all islands capable of supporting them and raccoon and red squirrel have also spread rapidly on the major islands, but have only recently reached some of the outer islands. Rats have populated entirely the islands on which they have been introduced, but do not appear to be spreading to other islands. Overall, the effects of these introduced species are increasing, so the trend is "deteriorating."

3. CONCLUSIONS

Based on analyses of Gwaii Haanas' research and monitoring programs, the status and trend of our indicator ecosystems vary individually (SoPR Section 3.8) and cannot be coalesced into single, park-wide values. Given that this is Gwaii Haanas' first Technical Compendium, and that the monitoring program is still under development, data gaps do limit the completeness of this overview. Forest occupies ~90 % of Gwaii Haanas' landscape and the rolled-up status and trend from it's measures is "fair" and "deteriorating" respectively. The next largest indicator ecosystem (8.9 %) is non-forested, for which the status and trend are both "unknown."

The greatest biotic stressor is intense browsing by hyperabundant introduced deer that profoundly affects plant species composition, successional patterns and distribution in forested and non-forested ecosystems (>98 % of the park's area) from the shoreline to the alpine. Introduced mammal predators (raccoon, rats) also threaten our EI, but are more localized and not at the same ecological scale as deer. The other major stressor will likely be climate change. This could cause a rise in sea level, increased frequency and intensity of storms, shifts in species ranges and a rise in the treeline. Alone, any one of these effects could have a significant influence, but in combination the consequences could be profound for Gwaii Haanas' EI. |

A key consideration for the next review period, besides the deer and climate issues, will be melding the terrestrial EI mandate with the multiple sustainable use mandate from the proposed NMCAR. The extent of the land-sea linkage with its mountain top-to-sea-bottom scope will be unique in Canada and provide new types of monitoring and management opportunities for Gwaii Haanas and the Agency as a whole.

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