Prince of Wales Island Amphibian Surveys 2013 and 2014

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Abstract

Understanding a species' ecological role and predicting the effect of habitat change on a species requires awareness of habitat preferences. To our knowledge, our study is the first in Southeast Alaska to investigate habitat usage of karst-influenced wetlands by amphibians. Studies elsewhere in the country have indicated that old growth loss has had negative implications on amphibians. Karst systems on Prince of Wales Island were of particular interest to us in relation to amphibian ecology, because they often support old growth and are highly productive ecosystems. Timber harvest on Prince of Wales Island has been disproportionately higher in these landscapes due to the presence of large, dense forest stands. If amphibians exhibited a preference for karst wetlands, this could have implications for forest management practices.

During 2013 and 2014 we surveyed 220 wetlands for amphibians on Prince of Wales Island, in both karst and non-karst influenced habitats. Species composition included western toads and rough-skinned newts, which were detected in juvenile and adult life stages. Both species have been previously described on the island, with most historical observations from the central part of the island. We added 28 western toad and 56 rough-skinned newt detection sites, with an emphasis on the northwest corner of the island, increasing our overall knowledge of amphibian distribution on Prince of Wales Island.

Rough-skinned newts were observed more frequently (25% of sites) and at higher elevations (up to 530 meters) than western toads (13% of sites; up to 200 meters). Detection rates at monitoring sites established by Pyare et al. in 2005, and resurveyed during this study, were 25% for western toads and 67% for rough-skinned newts.

Both western toads and rough-skinned newts were found widely distributed throughout the northern and central parts of the island in each of the 10 wetland types surveyed. Western toad detections were highest in needleleaf forested peatlands, followed by palustrine (emergent-aquatic bed), and herbaceous peatlands and anthropogenic sites. Rough-skinned newts were also commonly associated with needleleaf forested peatlands and anthropogenic sites, as well as lacustrine littoral habitats. Wetland types at the five sites where both species co-occurred included herbaceous peatlands, needleleaf forested peatlands, and palustrine (emergent-aquatic bed).

Based on the habitat preferences we observed in our study, both western toads and rough-skinned newts were frequently associated with karst wetlands, but did not exhibit a preference for karst sites over non-karst wetlands. Rather, water temperature, wetland type (especially peatlands and aquatic beds), and a high proportion of terrestrial vegetation cover likely play a large role in habitat selection by the two species. An absence of physical abnormalities and an abundance of amphibian observations suggest that amphibian populations on Prince of Wales Island are healthy. Periodic resurveying of the sites reported here can give an indication of future changes to population status in the area.

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Table of Contents

Figures

Tables

Introduction

Amphibian diversity in Alaska is relatively low compared to temperate and tropical regions, with the majority of species concentrated in the southeast region of the state. Six amphibian species are native to Alaska: the western toad (*Anaxyrus boreas*), wood frog (*Lithobates sylvaticus*), Columbia spotted frog (*Rana luteiventris*), rough-skinned newt (*Taricha granulosa*), northwestern salamander (*Ambystoma gracile*), and long-toed salamander (*Ambystoma macrodactylum*). Only the western toad and wood frog have been documented outside Southeast Alaska. The western toad has been recorded throughout the Southeast Panhandle and along the mainland coast as far north as Prince William Sound. The wood frog, which is the most hardy and widespread species of frog in North America, has been found from the mainland of Southeast Alaska northward to the Brooks Range. There have also been localized introductions of two non-native species, the Pacific chorus frog (*Pseudacris regilla*) and red-legged frog (*Rana aurora*). These non-native species apparently have viable but restricted populations in the Alexander Archipelago of Southeast Alaska on Revillagigedo Island and Chichagof Island, respectively (ADF&G 2006).

Despite low diversity and restricted ranges, little is understood about the distribution and population status of Alaska's amphibians. Previous surveys have been limited to small geographic areas with little or no systematic resampling over time. Anecdotal reports suggest that western toads may be decreasing throughout their range from Ketchikan to Haines (ADF&G 2006), yet these claims have not been substantiated. Accurate statewide abundance and population trends are unknown, which is a concern as amphibians are rapidly declining and disappearing from other parts of their range. For example, western toads are now absent throughout large areas of their former distribution in Colorado and southern Wyoming and may be extinct in New Mexico. These toads are classified as "endangered" by Colorado and New Mexico and are designated as a protected non-game species in Wyoming (Carey et al. 2005).

Amphibians are good indicators of significant environmental changes. They are sensitive to environmental factors such as habitat destruction, fungal infection, intensified predation by introduced fish and nonnative frogs, climate change, increased presence and diversity of pathogens, and combinations of these factors. Their declines can also indicate looming threats to other organisms (Blaustein et al. 1995). Globally, many reptiles and amphibians are experiencing range reductions and population declines due to climate change, invasive species, chemical pollution, overharvesting, and habitat destruction (Blaustein et al. 1995). Additionally, amphibians in many parts of North America, including some areas of Alaska, have unusually high occurrences of malformed limbs (Reeves et al. 2013). In light of these growing conservation concerns, and the importance of amphibian habitats for other fish and wildlife species, there is a need for basic information about amphibians in Alaska. This requires an understanding of species distribution, habitat needs, current status, and population trends of specific species (AFG&G 2006).

Between 2004 and 2006, Pyare et al. conducted a pilot study in the Prince of Wales Island, Admiralty Island, and Upper Lynn Canal regions of Southeast Alaska to establish monitoring sites that could be used to calculate initial occupancy rates for western toads. To our knowledge, the western toad breeding sites identified during this initial survey on Prince of Wales Island have subsequently not been revisited. The Pyare et al. study focused on the central part of the island, where wetlands are relatively abundant. By contrast, large wetland complexes overlaying karst habitat features in the northern portion of Prince of Wales Island have never been systematically surveyed for amphibians (Carstensen et al. 2003).

Karst occurs in parts of Southeast Alaska and is well represented on the west coast and northern portion of Prince of Wales Island (Soja 1990, Busch 1994, Baichtel and Swanston 1996). Karst refers to the chemically eroded landscape that develops on soluble bedrock, usually composed primarily of calcium carbonate ($CaCO_3$) such as limestone or marble. As water dissolves, the limestone increases the alkalinity of adjacent waterways, causing karst wetlands to develop unique hydrologic dynamics and have a higher pH compared to non-karst wetlands. For example, Bryant et al. (1998) surveyed karst and non-karst influenced streams on Prince of Wales, and found the alkalinity of the former to be more than twice that of non-karst (1500-2300 μeq/L and 750-770 μeq/L, respectively). Average pH (>7.8) and conductivity (127.8) were higher at karst sites than non-karst sites (<6.9 pH and 67.5 conductivity). Water temperature varied between sites, but was thought to be more strongly influenced by logging, riparian cover, and water source (e.g. upwelling from cave versus a lake) than bedrock. It is likely that bogs supporting anoxic soils and high acidity contribute to the dissolution of limestone and formation of karst landscapes (Bryant et al. 1998).

While surface water is much less abundant on internally-drained karst topography, forests are often highly productive as a result of soil nutrients and formation of well-developed drainages, and streams and ponds on karst are exceptionally productive for invertebrates and fishes. Bryant et al. (1998) found a positive relationship between alkaline limestone habitats and fish populations. More alkaline waters supported higher densities of coho salmon (*Onchorynchus nerka*), and salmon in karst-influenced streams were larger and more abundant compared to non-karst streams. The same study found a positive relationship between karst-influenced streams and invertebrate diversity. Additionally, high alkalinity has been found to correspond with high growth rates for brown trout (*Salmo trutta*; Campbell 1961, Neophitou and O'Hara 1986). To our knowledge, the implications of karst topography on amphibian occurrence and habitat suitability have not been studied (Carstensen et al. 2003).

Since karst systems are highly productive, timber harvest has been disproportionately higher in these landscapes due to the presence of large, dense forest stands (Baichtal and Swanston 1996). Loss of old growth may have negative effects on amphibians, as studies elsewhere in the country have indicated. For example, in New York State, one study found significantly reduced numbers of salamanders in disturbed habitats relative to old growth (Pough et al. 1987). In the southeastern United States, salamander captures were five times higher in mature forests than in clearcuts, and it is estimated that clearcutting in North Carolina National Forests has led to a loss of 14 million salamanders annually (Petranka et al. 1993). On Prince of Wales, clearcut logging is widespread which complicates the relationships between karst, water geochemistry, and aquatic and amphibian populations (Bryant et al. 1998).

Known habitat preferences

Both rough-skinned newts and western toads are known to occur regularly on Prince of Wales Island. The rough-skinned newt is the only member of the Salamandridae family that occurs in the Pacific Northwest. This newt ranges from central California to Southeast Alaska, west to the Cascade Range and up to 2800 m elevation, with a few outlying populations. It utilizes a variety of terrestrial and aquatic habitats, including ponds, lakes, streams, forests, woodlands, grasslands, and farmland (Blaustein et al. 1995), and is often more abundant in lakes and ponds relative to streams. Aquatic vegetation is needed for breeding habitat, and aquatic habitats surrounded by vegetation are preferred (Pimentel 1960). Several studies have reported rough-skinned newts more abundant in old growth forests in Washington, Oregon, and California (Aubry and Hall 1991, Corn and Bury

1991, Welsh and Lind 1991), while other studies in Oregon have shown the opposite, with newts equally dispersed or more abundant in young forests (Corn and Bury 1991, Gilbert and Allwine 1991). Factors other than forest age apparently have a strong influence on rough-skinned newt presence. For example, some studies have found them in greater abundance at lower elevations (Aubry and Hall 1991, Bury et al. 1991), while other have reported higher abundance at higher latitudes (Corn and Bury 1991, Gilbert and Allwine 1991). The microhabitat preferences of newts are not well understood, as they vary considerably between studies. Soil moisture, rainfall, temperature, and microtopography (decaying wood, leaf litter, logs, rocks, talus) likely all play a part in creating suitable habitat. Newts display high site fidelity, spending the winter in upland sites in proximity to lower wetland breeding sites, and require adequate dispersal corridors and riparian buffers for migration routes (Blaustein et al. 1995).

Western toads are widespread in Southeast Alaska, and range northward along the coast to Prince William Sound, including Montague and Hawkins Islands; the edge of their range is a short distance north to the Tasnuna River (a tributary of the Copper River) and west to the Columbia Glacier (MacDonald 2003). The species can be found at elevations over 3600 m (Hodge 1976, MacDonald 2003). The western toad is a terrestrial and wetland species found in humid open forests with moderate to dense undergrowth, including old fields and meadows, often near surface water. Primarily terrestrial, they enter water to breed in a variety of permanent or temporary quiet pools of streams and sloughs, wetlands, lakes, and ponds, including brackish pools. The occasional use of brackish pools for breeding is very unusual for North American frogs and toads. The species' tolerance for brackish and sea water have enabled it to disperse widely in Southeast Alaska and colonize islands (The Shipley Group 2009). Egg laying sites include shallow areas of ponds, lakes, or reservoirs, or pools of slow-moving streams. Tadpoles are habitat generalists but typically seek the warmest part of a waterbody (O'Hara 1981). They hibernate in burrows below frostline in forested cover adjacent to aquatic habitats (MacDonald 2003).

Objectives

Our project aimed to fill basic information voids on distribution and habitat requirements of amphibians in Southeast Alaska. Given the known presence of rough-skinned newts and western toads, and abundant karst topography on Prince of Wales Island, we investigated amphibian distribution in this unique and productive habitat type.

The specific goals of the project were to:

1) Gather baseline data on amphibian distribution, species composition, and habitat use, with a focus on wetlands within or downstream of karst topography.
2) Resurvey monitoring sites established by Pyare et al. (2004-2006) to document continued presence or absence of amphibians in non-karst habitats.
3) Compare the extent to which amphibians utilize karst and non-karst habitats and the suitability of site characteristics as they relate to amphibian occupancy between the two habitat types.
4) Collect specimens for future use in research, teaching, and outreach, and for inclusion in a national bio-contaminants database.

This study occurred during the amphibian breeding season (June/July) of 2013 (Year I) and 2014 (Year II). During Year I we revisited a subset of monitoring sites established by Pyare et al. to document occupancy rates, and conducted a pilot survey for amphibians in karst habitats (Walton et al. 2014). During Year II we focused our surveys in both karst and non-karst sites to assess the influence of karst topography on amphibian distribution and abundance and to gain a better understanding of overall habitat preferences.

Methods

Study area

Prince of Wales Island (6,674 km^2) is located in coastal temperate rainforest in the southern portion of the Alexander Archipelago in Southeast Alaska. The climate is characterized by high annual precipitation (average ca. 250 cm in Craig) with mild winters (January average temperature range -1°C to 4°C) and cool summers (July average range 10°C to 17°C; WRCC 2015). The coniferous rainforest on Prince of Wales Island is dominated by Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*), and interspersed with western red cedar (*Thuja plicata*), yellow-cedar (*Callitropsis nootkatensis*), red alder (*Alnus rubra*), and shore pine (*Pinus contorta* var. *contorta*; Harris et al. 1974). Elevations range from 0 to 1,092 m above sea level. The majority of the island is managed by the USDA Forest Service, Tongass National Forest. During year II of the study we included Kosciusko Island (447 km^2) in our survey area. Approximately 10% of the area of these two islands is underlain by karst features.

Site selection

We selected a variety of wetland types located in both karst and non-karst habitats for amphibian surveys (Figure 1). To identify potential survey sites, we extracted wetland types that contained possible amphibian breeding habitat from the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory (NWI) spatial data layer. These included lacustrine littoral wetlands (primarily aquatic bed) and palustrine wetlands (primarily unconsolidated bottom, aquatic bed, emergent, scrub-shrub, and forested). Using ArcGIS 10.1, we overlaid the NWI spatial data layer with the Alaska Department of Natural Resources (ADNR) 1:63,600 roads layer (containing primary, secondary, and logging roads) and applied a two km buffer so that any wetlands greater than two km from roads were removed. This excluded wetlands from our study area that would have been potentially difficult to access. We then overlaid the "accessible wetlands layer" with a map of surficial karst features (Albert et al. 2008) to identify wetlands within or downstream of karst formations, which we considered most likely to be impacted by karst topography. We then randomly selected 205 sites for our final set of survey sites, stratified by wetland class and presence/absence of karst, with a focus in the northwest and central quadrants of the island (Figure 1). In 2014, we added Kosciusko Island to the sampling area.

In addition to surveying the randomly generated sites described above, during the 2013 field season we also revisited 34 sites of known amphibian occurrence (Pyare unpubl. data). These sites were primarily western toad breeding sites in non-karst habitats that were concentrated within watersheds in the central portion of the island (Figure 1).

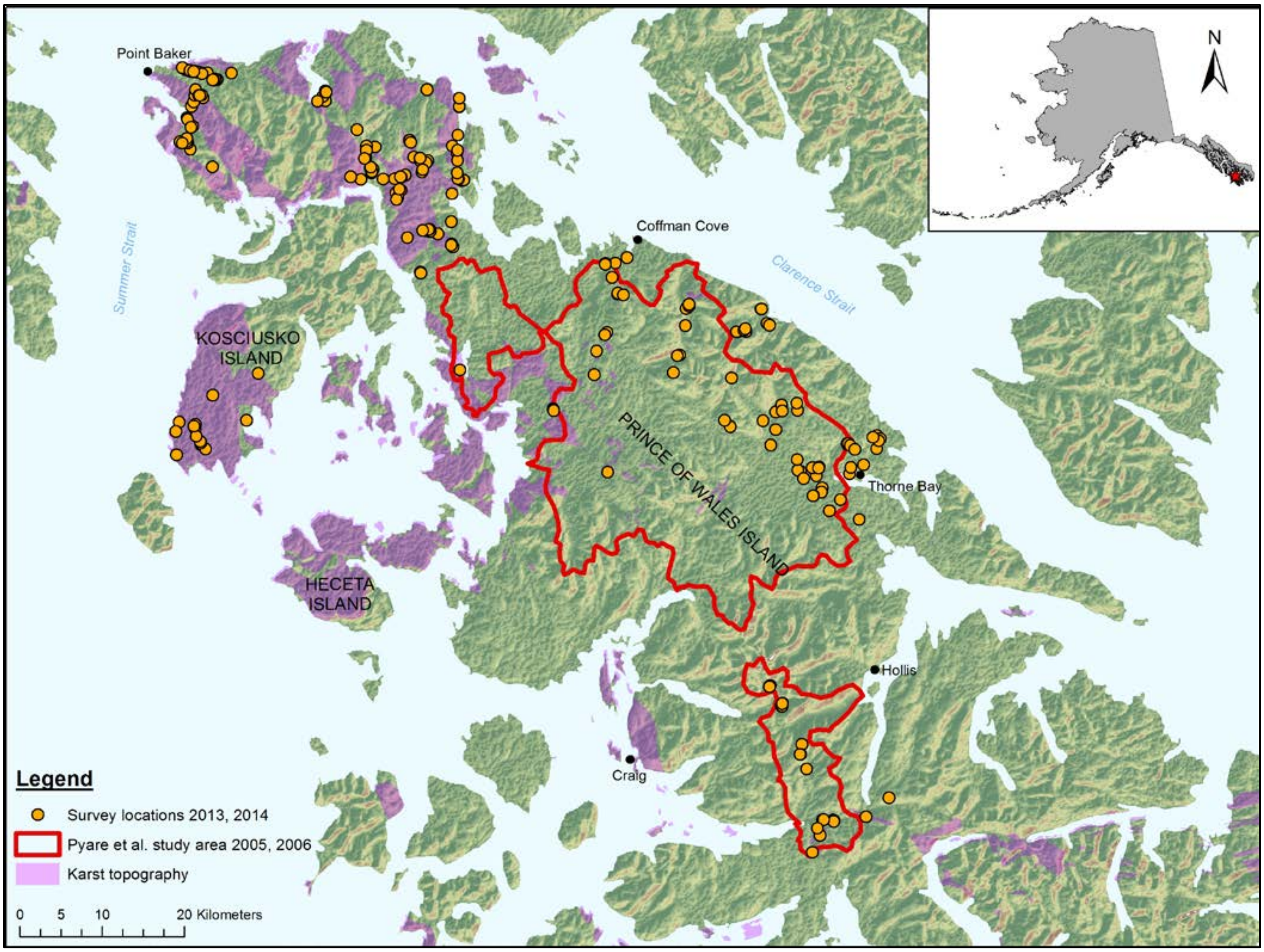

Figure 1. Location of 2013 and 2014 survey sites, Pyare et al. 2005-2006 survey area, and limestone/karst delineations on Prince of Wales Island.

Amphibian sampling

We used diurnal Visual Encounter Surveys (VES) to search for both western toads and rough-skinned newts at each sample location (Crump and Scott 1994, Carstensen et al. 2003). For distinct waterbodies we searched the shoreline, including the shallow aquatic zone and near-shore habitat, for evidence of amphibians using visual scans and net sweeps using dip-nets. We used a long-handled dip-net to sample for rough-skinned newts in small, deep, or mucky ponds that could not be visually searched due to poor water clarity. For relatively small lakes and wetlands, we surveyed the entire near-shore zone. For larger lakes, we surveyed a subsection of the shoreline.

On large peatlands we walked transects to systematically scan the entire wetland. When four observers were available, we divided the peatland into either four or five rectangular transects with the minimum possible width (i.e. long edges of rectangles parallel to long axis of survey area) to cover the entire survey area. We then walked one person to a transect at the same pace until we reached the end of the survey area. When only two observers were available, we generally walked lines perpendicular to the long axis of the survey area, moving approximately 5 to 10 m along the edge of the survey area parallel to the long axis between each perpendicular segment. Net sweeps in the waterbodies were completed after walking transects.

When amphibians were detected we recorded species, life stage, and number of individuals. For individuals captured, we also recorded standard measurements (snout to vent length and total length) and checked for deformities. In addition, we recorded detailed information on location, weather (air temperature, precipitation, and wind), wetland dimensions (length, width, surface area, percent shallow, and stream connections), water quality (pH, temperature, salinity, and clarity), habitat (terrestrial and aquatic vegetation, and aquatic substrate), disturbance, and presence/absence of fish and other predators. We took photographs in each cardinal direction. During Year I of the study, site characteristics were only recorded at sites where amphibians were detected. In Year II, detailed site characteristics were recorded at all survey sites. Where relevant data were lacking, sites were omitted from further analysis.

We characterized the surrounding habitat in the immediate vicinity of each wetland. For peatland wetland complexes with a series of small pools, the habitat was characterized for the entire peatland area, thus the terrestrial vegetation was a major component of the habitat. For ponds and lakes, we characterized the vegetation within the waterbody, as well as the terrestrial habitat bordering the shoreline (within approximately 5 m). As a result, the habitat often included a large water component (with or without aquatic vegetation) and a much smaller terrestrial component.

Specimen collection

During Year I, we collected a limited number of specimens using a buffered overdose of MS-222 to euthanize up to 10 western toad tadpoles (not subadults or adults) per site and up to 10 rough-skinned newts (of any life stage) per site. No more than 10% of the observed population was euthanized at any site. Specimens of western toads were only collected at sites with estimated or observed populations of over 100 tadpoles. Specimens were submitted to the Fishes Collection at the University of Alaska Museum (UAM) in Fairbanks, Alaska or the U.S. Geological Survey (USGS) Rangeland and Ecosystem Science Center in Corvallis, Oregon, for inclusion in a bio-contaminants database. All specimens were frozen immediately after collection and during transport. No specimens were collected during Year II of the survey.

Results

Survey effort and conditions

We surveyed a total of 239 sites during July 3 to 17, 2013 (n = 106) and June 10 to 29, 2014 (n = 133). Of these sites, 74 were on or adjacent to karst features (within 50 m buffer), and 165 were not associated with karst. Habitat information for 19 sites was categorized as “unclassified” because we only performed cursory checks for amphibians at those sites and detailed habitat information was not recorded. These 19 sites were omitted from our habitat assessment, leaving a total of 220 sites (64 karst, 154 non-karst) for final analysis.

Habitat types surveyed were classified as: seasonally flooded needleleaf forest wetlands (no waterbodies), seasonally flooded needleleaf forest wetlands (with waterbodies), beaver ponds and sloughs, lacustrine littoral (emergent – aquatic bed), riverine lower perennial (emergent – aquatic bed), palustrine (emergent – aquatic bed), needleleaf forest peatlands, herbaceous peatlands, tidal ponds and sloughs, and anthropogenic (See

Appendix 1 for detailed descriptions of each habitat type). Forty-six percent of surveyed wetlands were misclassified by NWI when compared to classifications observed in the field (n = 91; see Appendix 1 for details). For our analysis we used the empirical wetland classifications obtained in the field, not the mapped NWI class. In addition, we opportunistically surveyed several wetlands that were not identified by the NWI.

During the 2013 surveys, weather conditions were warmer and drier than normal (Table 1), resulting in partial desiccation of some wetlands and below average water levels. Air temperature recorded at survey sites ranged from 12.4°C to 32.0°C (mean: 17.2 ± 3.9°C). During 2014 surveys, the weather was more characteristic of Southeast Alaska, although wetter than average (Table 1). Air temperature recorded at survey sites ranged from 10.4°C to 24.9°C (mean: 14.6 ± 2.9°C).

Table 1. Temperature and precipitation averages for Craig, Alaska (NOAA National Weather Service - Alaska Climate Database).

	Monthly Normals (1981-2010)	2013	2014
June Temperature	11.8°C	13.4°C	12.1°C
July Temperature	14.2°C	14.8°C	14.6°C
June Precipitation	97 mm	66 mm	122 mm
July Precipitation	115 mm	76 mm	138 mm

Species composition and distribution

Amphibians were detected at 84 (38.18%) of the 220 total wetland sites surveyed. Western toads were recorded at 28 (12.73%) sites and rough-skinned newts at 56 (25.45%) wetland sites (Figure 2). Western toads were found at elevations between sea level and 200 m, while rough-skinned newts were found at sea level to 530 m. The majority of detections were single species, although both newts and toads were detected together at five sites (Figure 2). No other amphibian species were observed.

Both western toads and rough-skinned newts were found widely distributed throughout the northern and central parts of the island in each of the wetland types surveyed. Western toad detections were highest in needleleaf forested peatlands, followed by palustrine (emergent-aquatic bed), and herbaceous peatlands and anthropogenic sites. Rough-skinned newts were also commonly associated with needleleaf forested peatlands and anthropogenic sites, as well as lacustrine littoral habitats (Table 2). Both species occured at five sites that were categorized as herbaceous peatlands, needleleaf forest peatlands, and palustrine (emergent-aquatic bed; Table 3).

Table 2. Summary of survey sites by wetland type and total number of site detections for western toad (ANBO) and rough-skinned newt (TAGR) on Prince of Wales Island, 2013 and 2014. Percent detection was calculated by dividing the total number of sites in which amphibians were detected by the total number of sites surveyed.
*Survey sites where both ANBO and TAGR were detected.

		Total site detections			Percent detection		
Wetland type	Total sites surveyed	Total amphibian sites	Total ANBO sites	Total TAGR sites	All amphibian sites	ANBO	TAGR
Anthropogenic	9	10	3	7	4.55%	1.36%	3.18%
Beaver ponds and sloughs	40	8	2	6	3.64%	0.91%	2.73%
Herbaceous peatlands	19	9*	3	6	4.09%	1.36%	2.73%
Lacustrine littoral (emergent - aquatic bed)	31	10	2	8	4.55%	0.91%	3.64%
Needleleaf forest peatlands	48	24*	7	17	10.91%	3.18%	7.73%
Palustirne (emergent -aquatic bed)	19	10*	4	6	4.55%	1.82%	2.73%
Riverine lower perennial (emergent - aquatic bed)	8	1	1	0	0.45%	0.45%	0.00%
Seasonally flooded needleleaf forest wetlands (no waterbodies)	18	3	2	1	1.36%	0.91%	0.45%
Seasonally flooded needleleaf forest wetlands (with waterbodies)	21	6	2	4	2.73%	0.91%	1.82%
Tidal ponds and sloughs	7	3	2	1	1.36%	0.91%	0.45%
Total	**220**	**84**	**28**	**56**	**38.18%**	**12.73%**	**25.45%**

Sixty six (30%) of the 220 total sites surveys were in karst habitats, 154 (70%) sites were in non-karst habitats (Table 3). Within karst, we recorded the presence of western toads at 9 (13.63%) sites and rough-skinned newts at 15 (22.7%) sites. Detection values were similar for toads (n= 19; 12.3%) and newts (n = 41; 26.6%) in non-karst areas (Table 3). When detections were combined for both species, detection probabilities in karst (36.36%) were similar to those in non-karst sites (38.96%; Table 3; Figure 2). Amphibians were detected in each of the 10 wetland types surveyed in non-karst areas but were absent from riverine lower perennial and seasonally flooded needleleaf wetlands (with and without waterbodies) in karst areas (Table 3). There was no significant difference between western toad (t-test, df=169, t=-0.1170, p>0.05) or rough-skinned newt presence (t-test, df=169, t=0.7718, p>0.05) in karst versus non-karst influenced sites, when considering all habitat types combined. Small sample sizes precluded analysis within wetland type for differences in amphibian occurrence between karst and non-karst habitats.

Table 3. Summary of survey sites in karst and non-karst habitats, with wetland type and presence of western toads (ANBO) and rough-skinned newts (TAGR) on Prince of Wales Island, 2013 and 2014.

Wetland type	**Karst sites**			**Non-karst sites**		
	Total sites surveyed	ANBO	TAGR	Total sites surveyed	ANBO	TAGR
Anthropogenic	5	1	5	4	2	2
Beaver ponds and sloughs	13	2	2	27	0	4
Herbaceous peatlands	7	1	2	12	2	4
Lacustrine littoral (emergent - aquatic bed)	11	0	1	20	2	7
Needleleaf forest peatlands	10	1	3	38	6	14
Palustrine (emergent - aquatic bed)	8	3	2	11	1	4
Riverine lower perennial (emergent - aquatic bed)	1	0	0	7	1	0
Seasonally flooded needleleaf forest wetlands (no waterbodies)	5	0	0	13	2	1
Seasonally flooded needleleaf forest wetlands (with waterbodies)	4	0	0	17	2	4
Tidal ponds and sloughs	2	1	0	5	1	1
Total	**66**	**9**	**15**	**154**	**19**	**41**

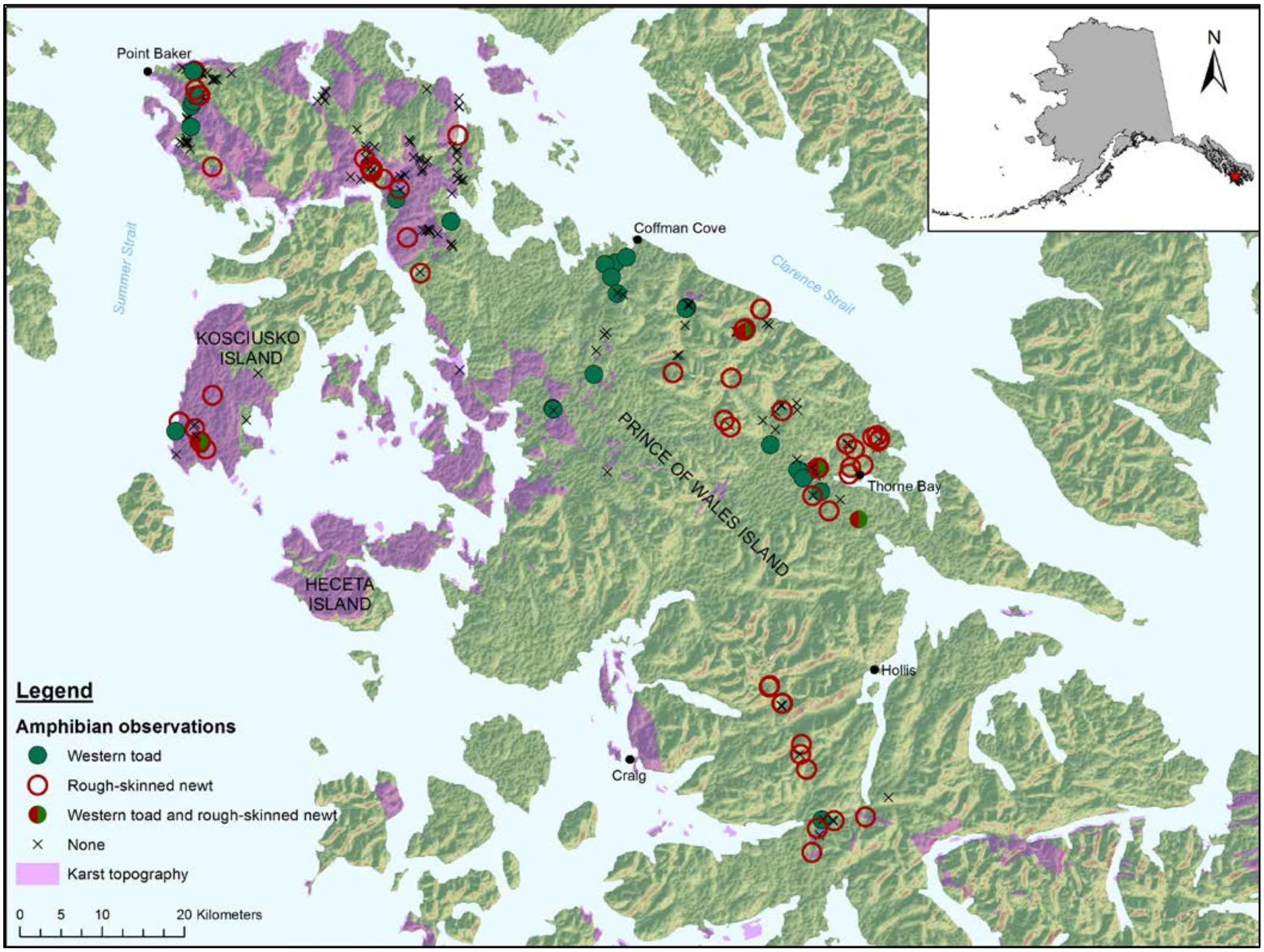

Figure 2: Location of western toad and rough-skinned newt observations in karst and non-karst influenced habitats on Prince of Wales Island, Alaska, during surveys in 2013 and 2014.

We detected rough-skinned newts at 67% of sites (n = 12) where they were previously described by Pyare et al. in 2005 and 2006 (this number is not corrected for by a probability of detection rate). We also found newts at three additional sites where Pyare et al. previously recorded the presence of western toads. Conversely, we only encountered western toads at 25% (n = 16) of resurveyed sites.

Both western toads and rough-skinned newts were detected during all active life stages. Yearling toads (25 to 45 mm) were observed more frequently than subadults (45 to 65 mm) and adults (>65 mm; Table 4; also see Appendix II for life stage descriptions) and were generally found very close to their natal ponds. Adult rough-skinned newts were detected more frequently than larval and metamorph forms, with all forms being found in or very close to shallow water bodies. There was no significant difference in rough-skinned newt adult or metamorph size (total length) at karst versus non-karst influenced sites (adult: t-test, df=183, t=0.56, $p>0.05$; metamorph: t-test, df=47, t=-1.35, $p>0.05$). Sample sizes for toads by age class were too small to compare total length at karst versus non-karst sites. No egg masses or deformities were observed in either year of the study.

Table 4. Counts and measurements of amphibians sampled in karst and non-karst sites on Prince of Wales Island during 2013 and 2014.

	All areas surveyed			Non-karst sites			Karst sites		
Life stage	Count	Avg. length (mm ± SD)	Range (mm)	Count	Avg. length (mm ± SD)	Range (mm)	Count	Avg. length (mm ± SD)	Range (mm)
Western toad									
Adults	4	81.3 ± 12	70-98	2	84 ± 20	70-89	2	78.5 ± 6	74-83
Subadults	7	52.1 ± 5	46-60	4	55.3 ± 4	50-60	3	48 ± 2	16-49
Yearlings	72	33.6 ± 4	26-45	69	33.4 ± 4	26-45	3	38.8 ± 9	28-44
Metamorphs	3	20.6 ± 4	16-23	3	20.6 ± 4	16-23	0	*NA*	*NA*
Larvae	3	19 ± 3	16-22	3	19 ± 3	16-22	0	*NA*	*NA*
Rough-skinned newt									
Adults	185	146.1 ± 22	63-200	117	146.8 ± 21	75-200	68	114.9 ± 24	63-183
Metamorphs	49	50.9 ± 9	32-70	46	50.5 ± 9	32-70	3	58 ± 8	50-66

Habitat characteristics

Water characteristics

Water characteristics were highly variable across sites (Table 5). Because many of the water bodies sampled were relatively shallow, water temperature was closely aligned with air temperature. There was no significant difference in water temperature between karst and non-karst influenced sites (ANOVA $F_{3,1}$ = 6.36, p = 0.01). However, water temperature was significantly higher at sites with amphibians present than sites without amphibian detections (ANOVA, $F_{3,1}$ = 5.73, p = 0.02). As expected, pH levels were significantly higher at karst versus non-karst sites (t-test, df=223, t=-3.9827, p<0.01), while there was no significant difference in pH levels in waterbodies with and without amphibians (ANOVA, $F_{3,1}$ = 0.01, p = 0.92). Salinity levels were also significantly higher at karst-influenced sites than non-karst sites (t-test, df=132, t=-1.88, p=0.03), but not significantly different between sites with and without amphibian observations (t-test, df=129, t=-1.8321, p=0.73; Table 5).

Table 5. Water characteristics summary table including mean values (±SD) and ranges (in parentheses) for air temperature, water temperature, pH, and salinity in karst and non-karst influenced wetlands, with amphibian presence (ANBO = western toad, TAGR = rough-skinned newt) and absence, Prince of Wales Island, 2013 and 2014.

	Karst-influenced sites				Non-karst sites			
Habitat characteristic	All sites (n=67)	ANBO sites (n=9)	TAGR sites (n=15)	Amphibians absent (n=45)	All sites (n=153)	ANBO sites (n=20)	TAGR sites (n=44)	Amphibians absent (n=93)
Air Temp (°C)	15.5 ± 3.5 (10.4-25.2)	16.2 ± 3.5 (11.4-21.9)	15.7 ± 3.6 (11.4-25.0)	15.2 ± 3.4 (10.4-25.2)	15.7 ± 3.6 (10.6-32)	15.8 ± 2.8 (10.6-23.3)	15.8 ± 3.6 (10.8-32)	15.6 ± 3.3 (10.7-24.9)
Water Temp (°C)	15.8 ± 3.5 (9.1-24.9)	16.9 ± 3.1 (12-22.2)	15.9 ± 3.6 (11.1-24.9)	15.6 ± 3.6 (9.1-24.9)	15.2 ± 3.6 (8.2-26.6)	15.3 ± 3.4 (10.3-24.3)	15.2 ± 3.6 (10.1-26.6)	15.1 ± 3.6 (8.2-24)
pH	7.2 ± 1.2 (4.34-9.14)	7.2 ± 1.3 (4.34-8.7)	7.2 ± 1.2 (5.93-9.14)	7.1 ± 1.2 (4.52-8.63)	6.6 ± 1.0 (4.24-9.61)	6.5 ± 1.0 (4.63-8.1)	6.6 ± 1.0 (4.81-7.63)	6.6 ± 1.0 (4.24-9.61)
Salinity (PSU)	51.6 ± 41.8 (2.0 -145.0)	39.1 ±41.1 (2-83.3)	64.8 ± 35.5 (14.5-125)	49.2 ± 43.2 (3.7-145)	35.2 ± 48.4 (1.6-342)	53.0 ± 87.9 (17.8-342)	30.5 ± 46.0 (1.6-247)	33.5 ± 34.2 (2.87-159)

Predator presence

Amphibian observations were positively correlated with aquatic predators, both fish and invertebrates, at survey sites (t-test, df=238, t=-3.79, p<0.01). Presence of predators was not significantly different for fish (t-test, df=238, t=1.91, p=0.06) or aquatic invertebrates (t-test, df=238, t=-0.50, p=0.61) between karst and non-karst influenced sites.

Predacious fish were most often found in riverine and lacustrine habitats, and were also common in tidal ponds and sloughs (Figure 3). The most commonly occurring fish species included adult and juvenile coho salmon (*Onchorhynchus kisutch*), cutthroat trout (*O. clarkii*), and three-spine stickleback (*Gasterosteus aculeatus*). Fish were absent from anthropogenic sites, which is to be expected, as most anthropogenic sites were isolated rock quarries. Invertebrate predators were found most frequently at lacustrine littoral, beaver ponds and sloughs, and anthropogenic sites (Figure 3). Major invertebrate predators included giant diving beetles, odonate naiads, and skimmers. Overall, lacustrine littoral habitats had the highest frequency of both fish and aquatic invertbrate predator species.

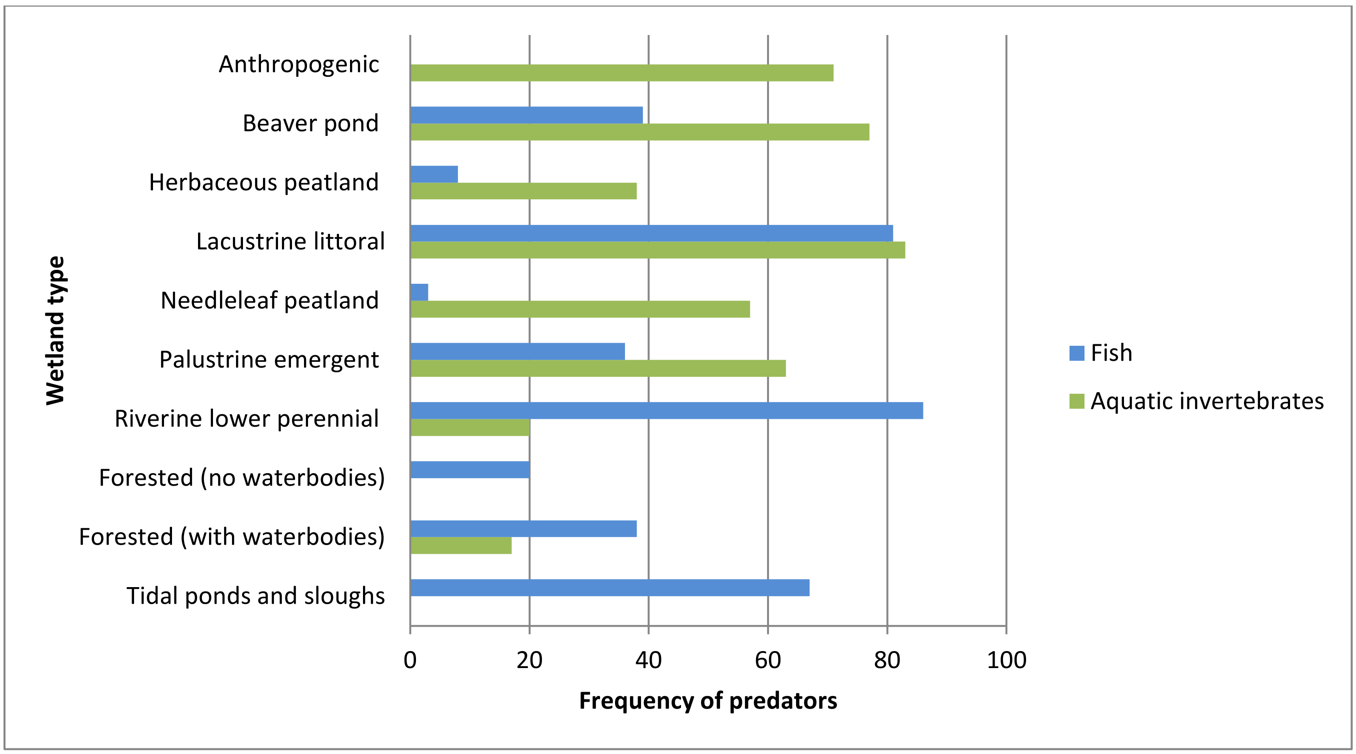

Figure 3: Percent frequency of predators across wetland habitat types sampled, Prince of Wales Island 2013 and 2014.

Wetland characteristics

Surface area of wetlands surveyed ranged from 5 to 45,000 m^2 (mean: 5311 ± 8213 m^2). There was no significant difference between wetland size at sites with and without amphibian observations (t-test, df=135, t=0.4784, p=0.6) or between karst and non-karst influenced sites (t-test, df=135, t=-0.95, p=0.34).

Water flow and drainage patterns were highly variable between wetlands. Wetlands with large waterbodies ranged from very slow moving water in river bends to stagnant water in lakes and ponds. Depth of water ranged from several centimeters in the shallowest littoral zones of lakes to several meters in abruptly deep beaver ponds and sloughs. Wetlands with small waterbodies or no waterbodies ranged from poorly drained, saturated soils interspersed with small pools in needleleaf forest peatlands to moist soils without obvious surface water in seasonally flooded needleleaf forest wetlands (no waterbodies).

Wetland vegetation

Most amphibians spend their early life stages undercover of aquatic vegetation and then move into more terrestrial habitats as they mature. We characterized both aquatic and terrestrial habitats at each survey site, assuming that amphibians detected in or near waterbodies would likely disperse into adjacent terrestrial areas. We classified aquatic vegetation into three categories: emergent, floating, and submerged. Terrestrial habitats, which become more important to anurans during subadult and adult life stages, were also subdivided into three categories: herbaceous, shrub, and tree.

For the aquatic vegetation, emergent plant cover was the greatest in palustrine and riverine lower perennial habitats, while floating plants had the highest average percent cover in lacustrine littoral and palustrine habitats (Figure 4). The dominant emergent plants included *Menyanthes trifoliata* (buckbean) and *Equisetum* spp. (horsetail). The major floating plants were *Nuphar polysepala* (water lily)*, Potamogeton* spp. (pondweed), and *Sparganium* spp. (bur-reed; Table 6). Submerged vegetation had the highest percent cover of all the aquatic vegetation types combined, and had the highest overall cover in anthropogenic and tidal ponds and slough habitat types (Figure 4). Submerged plants were dominated by *Chara* spp. (stonewort), which is a green or gray-green alga that has been positively correlated with amphibian breeding sites elsewhere in Southeast Alaska (Carstensen et al. 2003). A full list of aquatic vegetation observed during surveys is presented in Table 6.

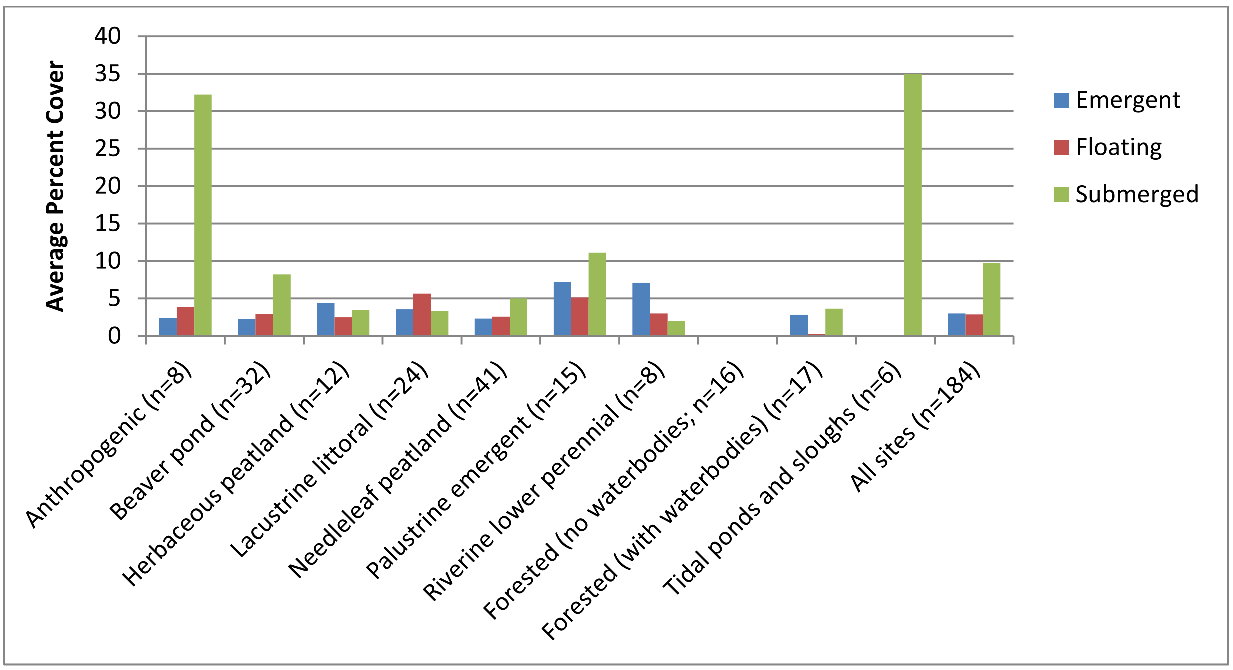

Figure 4. Average percent cover of aquatic vegetation in different wetland types surveyed. Some wetland class names are abbreviated but correspond to those discussed within this report. In 2013, vegetation was only recorded at sites where amphibians were observed.

Table 6. Aquatic vegetation species list including percent cover documented for all sites combined and the total number of sites each taxa was observed. The list is arranged in descending order of total cover within emergent, floating, and submerged vegetation categories.

Scientific name	Sum of Percent Cover	Number of Sites
Emergent vegetation		
Menyanthes trifoliata	224	61
Equisetum spp.	187	26
Carex spp.	60	13
Ranunculus trichophyllus	40	1
Ranunculus flammula	23	11
Caltha palustris	12	15
Lysichiton americanus	8	3
Phalaris arundinacea (non-native)	6	2
Comarum palustre	5	21
Cicuta douglasii	4	7
Hippuris spp.	3	2
Juncus spp.	2	7
Lycopodium annotinum	2	1
Scirpus spp.	2	1
Floating vegetation		
Nuphar polysepala	235	64
Potamogeton spp.	140	33
Sparganium spp.	114	39
Utricularia spp.	27	9
Submerged vegetation		
Chara spp.	379	30
Myriophyllum spp.	72	18
Ruppia maritima	30	1
Ranunculus trichophyllus	17	2
Utricularia spp.	12	11
Stuckenia filiformis	5	2
Sparganium spp.	5	4
Epilobium ciliatum	2	1
Hippuris vulgaris	2	1
Carex viridula	1	1
Potamogeton richardsonii	1	1

While aquatic vegetation may have the greatest influence on amphibian breeding site selection, across all wetland types surveyed, the majority of wetland vegetation was comprised of terrestrial herbs, shrubs and trees (Table 7). Where western toads were detected average cover was 44.3% for herbaceous plants, 10.5% for shrubs, and 19.8% for trees. Average cover values for sites with rough-skinned newts were similar to sites with toads, with average values of 48.1% for herbaceous plants, 9.1% for shrubs, 17.4 for trees (Table 7).

Table 7. Average cover and total site presence for aquatic and terrestrial vegetation for survey sites with western toads and rough-skinned newts present and sites without amphibian detections, Prince of Wales, 2013 and 2014. Sites that did not have vegetation data collected were omitted from analysis.

	Western toads present (n=28)		**Rough-skinned newts present** (n=59)		**Sites without amphibian detections** (n=99)	
Vegetation cover type	Total site presence	Average cover (% ± SD)	Total site presence	Average cover (% ± SD)	Total site presence	Average cover (% ± SD)
Aquatic emergent	18	7.1 ± 12	44	5.4 ± 6.8	44	5 ± 8.9
Aquatic floating	15	5.7 ± 6.1	37	7.6 ± 8.3	36	5.2 ± 6.7
Aquatic submerged	13	13.5 ± 16	23	14.4 ± 18	25	3.7 ± 4.8
Terrestrial herbaceous	28	44.3 ± 16.4	58	48.1 ± 21.6	91	42.2 ± 21.7
Terrestrial shrub	18	10.5 ± 11.3	46	9.1 ± 7	85	14.1 ± 13.7
Terrestrial tree	26	19.8 ± 16.6	59	17.4 ± 11.1	91	26 ± 18.7

Specimens collected

We collected a total of 76 voucher specimens: 20 western toads and 56 rough-skinned newts. All western toad specimens were tadpoles, collected from two sites (10 tadpoles per site). Of the 56 rough-skinned newts, 34 were adults, one was a metamorph, and 21 were larvae. All western toad specimens and 29 rough-skinned newt specimens were sent to the Fishes Collection at the University of Alaska Museum in Fairbanks, Alaska. The remaining 27 newt specimens were sent to the USGS Rangeland and Ecosystem Science Center in Corvallis, Oregon for contaminants analysis.

Discussion

Understanding a species' ecological role and predicting the effect of habitat change on a species requires awareness of habitat preferences. To our knowledge, our study is the first in Southeast Alaska to investigate habitat usage of karst-influenced wetlands by amphibians. During two field seasons in 2013 and 2014 we surveyed over 220 wetlands for amphibians on Prince of Wales Island in both karst and non-karst influenced habitats. Studies elsewhere in the country have indicated that old growth loss has had negative implications on amphibians. Karst systems on Prince of Wales Island were of particular interest to us in relation to amphibian ecology, because they often support old growth and are highly productive ecosystems. Timber harvest on Prince of Wales Island has been disproportionately higher in these landscapes due to the presence of large, dense forest stands (Baichtal and Swanston 1996). If amphibians exhibited a preference for karst wetlands, this could have implications for forest management practices.

Western toads and rough-skinned newts were detected in juvenile and adult life stages across Prince of Wales during our study. Both species have been previously described on the island, with most historical observations from the central part of the island. We added 28 western toad and 56 rough-skinned newt detection sites from central and northern parts of the island, increasing our knowledge of amphibian distribution on Prince of Wales Island.

Our surveys were designed to capture amphibians post-breeding and prior to metamorphosis – a time when larvae would be highly visible in breeding ponds. Therefore, the habitat associations we provide relate to that time period only. All detections were either in or within 15 m of breeding ponds, even though we scanned forested and non-forested areas farther away from waterbodies and also looked under dead wood and logs when we encountered them. Overall, we did not encounter amphibians far from breeding sites.

Rough-skinned newts were observed more frequently (25% of sites) and at higher elevations (up to 530 meters) than western toads (13% of sites; up to 200 meters). Although not directly comparable because we did not calculate occupancy rates, our detection rates for western toads were similar to those reported by Pyare et al. in 2005 (14% occupancy) and higher for rough-skinned newts (15-17% occupancy).

Detection rates for amphibians at the resurveyed sites (previously describe by Pyare et al. 2005-2006) were variable between species. We detected rough-skinned newts at 67% of resurveyed sites and found newts at three additional locations where Pyare et al. had previously recorded the presence of western toads only. Conversely, western toad observations occurred at only 25% of resurveyed wetlands. Since survey locations were not exclusively breeding sites, and adults have a tendency to move across the landscape (Maxcy and Richardson 1999, Muths and Guyer 2003, Bartelt et al. 2004), these low detection rates are likely indicative of inter-annual variation in site use and/or low detectability rates rather than declining populations. In addition, the uncharacteristically warm and dry weather observed in 2013 may have reduced amphibian activity during the daytime while we were actively surveying. During several afternoons when air temperatures exceeded 25°C, we noticed rough-skinned newts were only found buried within the cool sediments of ponds and were not observed on the surface.

Amphibians were observed in a broad range of habitats and were detected in 35% of wetlands, comprised of 10 different wetland classes. The majority of amphibian observations occurred in needleleaf forested peatlands and herbaceous peatlands, lacustrine littoral habitats, and anthropogenic sites. Such variability in habitat use is consistent with observations elsewhere in Southeast Alaska (Waters 1992, Norman and Hassler 1996, Carstensen 2003).

At sites across Prince of Wales Island gravel extraction has created a large number of anthropogenic ponds. Western toads have been reported breeding in small gravel pits in the Juneau, Haines, and Gustavus areas, but the high amphibian encounter rates we had at anthropogenic sites were still surprising. Of the eight anthropogenic sites surveyed, six contained adult and larval rough-skinned newts and two contained adult western toads, with overlap of both species at one site. These areas were generally devoid of emergent vegetation for protective cover, but had been inactive long enough that submerged vegetation had established and was generally dominated by *Chara* species of alga (stonewort; 20 to 50% cover). All anthropogenic sites except one had fairly basic conditions (pH values >7.6), shallow water, and high numbers of aquatic invertebrates. Large western toad breeding populations have also been associated with *Chara* presence and basic water conditions in other parts of Southeast Alaska (Carstensen et al. 2003).

As expected, water conditions varied between karst and non-karst habitats, with higher pH and salinity levels recorded in karst-influenced waterbodies. However, amphibians were present in both karst and non-karst influenced habitats with no significant difference in observations between sites, and unlike the findings of Bryant et al. (1998), there was no significant difference in water temperatures between these site types. These results suggest that although some water characteristics vary between karst and non-karst areas, the range of pH and salinity levels measured at the two habitat types are within the tolerable threshold for amphibians and did not result in a preference for one habitat type over the other. However, water temperature was significantly higher at sites with amphibians, suggesting that water temperature has a stronger influence on amphibian presence than pH or conductivity. Amphibians were not observed in waterbodies with temperatures lower than 10°C and metamorphs and larvae were observed in waterbodies with an average temperature of 17.1°C (16 sites, range 11.1°-22°C). Larval amphibians require warmer water temperatures to boost metabolic rates and accelerate the growth of algal food sources.

Habitat loss and fragmentation are recognized as the greatest cause of amphibian imperilment; therefore, activities such as forest clearcutting and land conversion have great potential to affect amphibian populations (Todd et al. 2009). Due to time and accessibility constraints, we were unable to incorporate a statistical assessment of amphibian use of logged versus non-logged sites into our study design. However, whenever possible, we did survey in clearcut areas. We visited a total of 15 clearcut sites and six buffer sites (habitat directly adjacent to recently logged areas) and found amphibians at 40% of logged and 67% of buffer sites, with an equal number of detections for toads and newts at each habitat type. Based on these limited observations, it appears that toads and newts are using logged areas during the breeding season. To promote the persistence of amphibian populations, conservation efforts should focus on preserving forest habitat adjacent to reproduction sites. Such measures are especially important where forest habitat connects local populations, such as multi-pond complexes, or where it links reproduction sites to other habitat features necessary for amphibian growth, survival, or overwintering, such as neighboring upland sites.

In conclusion, our primary goal in this study was to gather baseline data on amphibian distribution and habitat use on Prince of Wales Island and determine if karst wetlands are important amphibian habitat. After two years of surveys we feel that we achieved our goal by adding 84 new breeding sites to our collective understanding of amphibian distribution on the island, with the majority of new sightings from the northern end of the island. Based on the habitat preferences we observed in our study, both western toads and rough-skinned newts were frequently associated with karst wetlands, but did not exhibit a preference for karst sites over non-karst wetlands. Rather, water temperature, wetland type (especially peatlands and aquatic beds), a high proportion of terrestrial vegetation cover, and the presence of aquatic predators likely play an important role in habitat selection. An absence of physical abnormalities and an abundance of amphibian observations suggest that amphibian populations on Prince of Wales Island are healthy. Periodic resurveying of the sites reported on here can give an indication of future changes to population status in the area.

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Appendix 1: Habitat classes

Habitat classes and characteristics

Wetland habitat types surveyed on Prince of Wales Island included predominantly terrestrial sites and predominantly aquatic sites. Habitat classes for the more terrestrial sites were derived from the vegetation classes of the Alaska Vegetation Map and Classification: Southern Alaska and Aleutian Islands (Boggs et al. 2013). Habitat classes for the predominantly aquatic sites were derived from the subclasses of the U.S. Fish and Wildlife Service National Wetlands Inventory (NWI; USFWS 2009). The "seasonally flooded needleleaf forest "wetland class of Boggs et al. (2013) was subdivided into those with and without waterbodies, since the presence of waterbodies potentially has a strong effect on suitability for amphibians. The "beaver ponds and sloughs" class was separated out from the palustrine (emergent – aquatic bed) subclass of the NWI because beaver activity often produces unique waterbodies that are abruptly deep and lack shallow water for much of the shoreline. In 2014 two habitat classes were added: anthropogenic, and tidal ponds and sloughs.

During the 2013 and 2014 field seasons, we surveyed the following habitats for amphibians:

- Seasonally flooded needleleaf forest wetlands (no waterbodies)
- Seasonally flooded needleleaf forest wetlands (with waterbodies)
- Beaver ponds and sloughs
- Lacustrine littoral (emergent – aquatic bed)
- Riverine lower perennial (emergent – aquatic bed)
- Palustrine (emergent – aquatic bed)
- Needleleaf forest peatlands
- Herbaceous peatlands
- Anthropogenic (2014 only)
- Tidal ponds and sloughs (2014 only)

The descriptions of habitat classes provided in this appendix are based on the descriptions of coarse scale vegetation classes from Vegetation Map and Classification: Southern Alaska and Aleutian Islands (Boggs et al. 2013), but have been modified to more closely identify the sites that we surveyed in 2013 and 2014 on Prince of Wales Island. The common vegetation listed for each habitat class therefore reflects the vegetation that we observed at survey sites rather than vegetation common to the vegetation classes across southern Alaska.

Seasonally flooded needleleaf forest wetlands (no waterbodies)

The seasonally flooded needleleaf forest wetlands (no waterbodies) habitat class includes freshwater wetlands in open to closed needleleaf forests that lack streams, ponds, or other significant bodies of water (Figure 5). Soils are poorly drained and typically saturated. There may be some shallow standing water but there are no permanent waterbodies or large ephemeral waterbodies. Total canopy cover of trees is normally 25% or higher and needleleaf trees dominate.

Tsuga heterophylla and *Picea sitchensis* are usually dominant or codominant in the forest canopy. *Thuja plicata* and *Callitropsis nootkatensis* are also common. Common shrubs include *Vaccinium ovalifolium*, *Oplopanax horridus*, *Menziesia ferruginea*, and *Rubus spectabilis*. Common forbs include *Lysichiton americanus*, *Gymnocarpium dryopteris*, *Phegopteris connectilis*, and *Thelypteris quelpaertensis*. *Sphagnum* moss is common. The presence of *Lysichiton americanus* is indicative of soils that are usually saturated or inundated.

Figure 5. Seasonally flooded needleleaf forest wetland (no waterbodies) from site TAGR79-SP. Tree cover at this site was dominated by *Tsuga heterophylla* and *Picea sitchensis*.

Seasonally flooded needleleaf forest wetlands (with waterbodies)

The seasonally flooded needleleaf forest wetlands (with waterbodies) habitat class includes freshwater wetlands in open to closed needleleaf forests with streams, ponds, or other significant bodies of water (Figure 6). Soils are poorly drained and typically saturated. There may be some shallow standing water in addition to permanent waterbodies or large ephemeral waterbodies. Total canopy cover of trees is normally 25% or higher and needleleaf trees dominate.

Tsuga heterophylla and *Picea sitchensis* are usually dominant or codominant in the forest canopy. *Thuja plicata* and *Callitropsis nootkatensis* are also common. *Alnus rubra* is sometimes present. Common shrubs include *Vaccinium ovalifolium*, *Oplopanax horridus*, *Menziesia ferruginea*, and *Rubus spectabilis*. Common forbs include *Lysichiton americanus*, *Gymnocarpium dryopteris*, *Phegopteris connectilis*, and *Thelypteris quelpaertensis*. *Sphagnum* moss is common. The presence of *Lysichiton americanus* is indicative of soils that are usually saturated or inundated.

Figure 6. Seasonally flooded needleleaf forest wetland (with waterbodies) from site TAGR78-SP. Small ponds and creeks were present in this forested wetland complex. Tree cover was dominated by *Tsuga heterophylla* and *Picea sitchensis*.

Beaver ponds and sloughs

This habitat class consists of seasonally flooded needleleaf forest wetlands (with waterbodies) or palustrine (emergent – aquatic bed) sites that have been modified by beaver activity (Figure 7). Both active and inactive dams are included. The beaver ponds and sloughs class has been separated out from the two previously mentioned habitat classes because beaver activity often creates networks of abruptly deep channels and ponds. Beaver dams range from small ponds to lake-sized waterbodies.

Vegetation is usually similar to the vegetation described for seasonally flooded needleleaf forest wetlands (with waterbodies) or palustrine (emergent – aquatic bed) habitat classes. Common tree species include *Picea sitchensis*, *Tsuga heterophylla*, *Thuja plicata*, and *Alnus rubra*. Standing dead trees and tall graminoid terrestrial vegetation are common, as is the non-native grass *Phalaris arundinacea.*

Figure 7. Beaver ponds and sloughs with typical standing dead trees and tall graminoid terrestrial vegetation. An adult western toad was found on the woody debris of the large beaver pond at site BUBO-2013-10 (left). Beaver sloughs are often abruptly deep, such as this beaver slough from northern Prince of Wales Island (right).

Lacustrine littoral (emergent – aquatic bed)

The lacustrine littoral (emergent – aquatic bed) habitat class extends from lake shallows to terrestrial wetlands and includes terrestrial, emergent, floating, and submerged vegetation (Figure 8). This class does not include lakes in peatlands, as these are included within the needleleaf forest peatland or herbaceous peatland classes. Vegetation at these sites is usually a combination of the herbaceous (wet-marsh) coarse vegetation class and herbaceous aquatic coarse vegetation class from Boggs et al. (2013).

Herbaceous (wet-marsh) vegetation dominates the terrestrial shorelines and shallow waters. The water table for most of the growing season ranges from just below the ground surface, to at the ground surface, to above the ground surface in shorelines with emergent wetlands. Soils are mineral soil or muck over mineral soil, or have an organic layer less than 40 cm thick. Organic material may be composed of *Sphagnum* moss, *Carex* spp., or other plant material and can occur over mineral soil or may be a floating root mat.

Along shorelines where the water table is typically above the ground surface (i.e. marsh or emergent wetland), common plants include *Carex sitchensis*, *Carex utriculata*, *Comarum palustre*, *Equisetum fluviatile*, and *Menyanthes trifoliata*. Species diversity is often low. Along drier shorelines where the water table is typically below the ground surface (i.e. terrestrial shorelines), common plants include *Carex sitchensis*, *Comarum palustre*, and *Nephrophyllidium crista-galli*. The invasive grass *Phalaris arundinacea* is present or common at some sites.

Floating and submerged aquatic vegetation occurs in deeper water of lakes. *Nuphar polysepala* is often present. Other common species include *Callitriche* spp., *Isoetes* spp., *Myriophyllum sibiricum*, *Potamogeton* spp., and *Sparganium angustifolium*. *Chara* spp. (algae) are common.

Figure 8. Lacustrine littoral (emergent – aquatic bed) shorelines are often dominated by *Carex sitchensis*. *Nuphar polysepala* is common in the deep littoral zone of the lake at site ABS-2013-124 (left). *Equisetum fluviatile* is common in the shallow littoral zone of the lake at site ABS-2013-25 (right).

Riverine lower perennial (emergent – aquatic bed)

The riverine lower perennial (emergent – aquatic bed) habitat class extends from river shallows to terrestrial wetlands and riverbanks of rivers with year-round flow at low elevations (does not include headwater streams). These sites include terrestrial, emergent, floating, and submerged vegetation (Figure 9). This class does not include rivers through peatlands. Vegetation at these sites is usually a combination of the herbaceous (wet-marsh) coarse vegetation class and herbaceous aquatic coarse vegetation class from Boggs et al. (2013). The riverine lower perennial sites that we surveyed generally had slow moving waters and included protected waters.

Herbaceous (wet-marsh) vegetation dominates the terrestrial riverbanks and shallow water of rivers. The water table for most of the growing season ranges from just below the ground surface, to at the ground surface, to above the ground surface on riverbanks with emergent wetlands. Soils are mineral soil or muck over mineral soil, or have an organic layer less than 40 cm thick. Organic material may be composed of *Sphagnum* moss, *Carex* spp., or other plant material and can occur over mineral soil or may be a floating root mat.

Along riverbanks where the water table is typically above the ground surface (i.e. marsh or emergent wetland), common plants include *Carex sitchensis*, *Carex utriculata*, *Ranunculus flammula*, and *Ranunculus trichophyllus*. Species diversity is often low. Along drier riverbanks where the water table is typically below the ground surface (i.e. terrestrial riverbanks), common plants include *Carex sitchensis* and *Carex utriculata*. *Calamagrostis canadensis*, *Rubus spectabilis*, and *Spiraea douglasii*, as well as the invasive grass *Phalaris arundinacea*, are common at some sites. Herbaceous aquatic vegetation occurs in deeper waters of rivers and is dominated by floating and submerged vegetation. Common species include *Callitriche* spp., *Isoetes* spp., *Nuphar polysepala*, *Potamogeton* spp., and *Sparganium angustifolium*. *Chara* spp. (algae) are common.

Figure 9. Riverine lower perennial (emergent – aquatic bed) site consisting of a shallow, heavily vegetated river bend at site BUBO-2013-36. Numerous western toad tadpoles were found in shallow waters protected from the river current by gravel bars.

Palustrine (emergent – aquatic bed)

The palustrine (emergent – aquatic bed) habitat class consists of small to large ponds and adjacent terrestrial shorelines and includes terrestrial, emergent, floating, and submerged vegetation (Figure 10). This class does not include ponds in peatlands (these are included within the needleleaf forest peatland or herbaceous peatland classes). Vegetation at these sites is usually a combination of the herbaceous (wet-marsh) coarse vegetation class and herbaceous aquatic coarse vegetation class from Boggs et al. (2013).

Herbaceous (wet-marsh) vegetation dominates the terrestrial shorelines and shallow zones of ponds. The water table for most of the growing season ranges from just below the ground surface, to at the surface, to above the ground surface in shorelines with emergent wetlands. Soils are mineral soil or muck over mineral soil, or have an organic layer less than 40 cm thick. Organic material may be composed of *Sphagnum* moss, *Carex* spp., or other plant material and can occur over mineral soil or may be a floating root mat.

Along shorelines where the water table is typically above the ground surface (i.e. marsh or emergent wetland), common plants include *Carex sitchensis*, *Carex utriculata*, *Comarum palustre*, *Equisetum fluviatile*, *Equisetum palustre*, *Menyanthes trifoliata*, and *Ranunculus flammula*. Species diversity is often low. Along drier shorelines where the water table is typically below the ground surface (i.e. terrestrial shorelines), common plants include *Carex flava*, *Carex livida*, *Carex sitchensis*, *Equisetum palustre*, *Juncus ensifolius*, and *Scirpus microcarpus*.

Herbaceous aquatic vegetation occurs in the deep zone of ponds and is dominated by floating and submerged vegetation. Common species include *Callitriche* spp., *Isoetes* spp., *Myriophyllum sibiricum*, *Nuphar polysepala*, *Potamogeton* spp., and *Sparganium angustifolium*. *Chara* spp. (algae) are common.

Figure 10. Palustrine (emergent – aquatic bed) sites consisting of small ponds and emergent wetlands. Numerous rough-skinned newt adults were found buried in the organic muck at the bottom of a shallow pond at site TAGR82-SP (left). One rough-skinned newt was observed among *Sparganium angustifolium* and *Chara* spp. at site TAGR-2013-104 (right).

Needleleaf forest peatlands

Peatlands include all sites with organic layers more than 40 cm deep, composed largely of *Sphagnum* moss. Needleleaf forest peatlands have tree cover ranging from 10 to 30% and often include small ponds and channels (Figure 11). Needleleaf forest peatlands occur on low to mid elevation headlands, uplifted marine deposits, piedmonts, and ancient inactive outwash deposits. Sites are usually flat to low-angled and poorly drained. Some needleleaf forest peatlands may develop on fairly steep sideslopes in areas with very high rainfall and low permeability. Soils are poorly drained and often saturated to the surface.

Pinus contorta var. *contorta* is usually the dominant needleleaf tree. *Tsuga heterophylla*, *Picea sitchensis*, and *Callitropsis nootkatensis* are also common. Trees growing within peatlands are typically stunted. Common shrubs include *Empetrum nigrum*, *Juniperus communis*, *Rhododendron groenlandicum*, and *Vaccinium uliginosum*. Common herbaceous species include *Carex limosa*, *Carex livida*, *Eriophorum angustifolium*, *Nephrophyllidium crista-galli*, and *Trichophorum cespitosum*. Sphagnum moss is usually abundant. *Carex sitchensis*, *Menyanthes trifoliata*, *Nuphar polysepala*, and *Lysichiton americanus* are common in small ponds within needleleaf forest peatlands.

Figure 11. Needleleaf forest peatlands on Prince of Wales Island dominated by *Pinus contorta* var. *contorta.* A single adult western toad was found in the saturated organic muck at site BUBO-2013-07 (left). This habitat class is often a complex of forested and non-forested peatlands, such as site ABS-2013-126 (right).

Herbaceous peatlands

The herbaceous peatlands habitat class consists of well-developed peatlands (bogs and fens) with herbaceous vegetation cover greater than 25% and an organic layer that ranges from 40 cm to over 2 m deep (Figure 12). The organic layer may be composed of *Sphagnum* moss, *Carex* spp., or other plant material and can occur over mineral soil or may be floating or submerged. Soils are poorly drained and often saturated to the surface. Tree cover is usually less than 10%, and shrub cover is usually less than 25%.

Bogs are acidic, nutrient-poor peatlands. Sedges and shrubs dominate; *Carex livida*, *Carex pauciflora*, *Carex sitchensis*, *Eriophorum angustifolium*, and *Trichophorum cespitosum* are most common. *Nephrophyllidium crista-galli*, *Microseris borealis* and *Drosera rotundifolia* are common forbs. Ericaceous shrubs are characteristic of bogs and common taxa include *Empetrum nigrum*, *Kalmia microphylla*, *Rhododendron groenlandicum*, and *Vaccinium ovalifolium*. *Sphagnum* moss is also abundant (Keddy 2010).

Fens are alkaline, mineral-rich peatlands. Grasses and sedges dominate; *Carex sitchensis* is most common (McClellan et al. 2003), although a variety of other sedges and forbs may be present, including *Eriophorum russeolum,* and carnivorous plants. Rather than *Sphagnum* moss, *Scorpidium* or *Drepanocladus* mosses dominate (Keddy 2010).

Small ponds and channels are common in herbaceous peatlands. Within these small waterbodies, *Carex sitchensis*, *Menyanthes trifoliata*, *Nuphar polysepala*, and *Lysichiton americanus* are common.

Figure 12. Herbaceous peatlands are often dominated by *Carex* spp., *Eriophorum* spp., and *Trichophorum cespitosum*. A single adult western toad was found in the saturated organic muck at site BUBO-2013-131 (left). A single adult rough-skinned newt was found sheltering underneath a rotted log at site TAGR-2013-105 (right).

Tidal ponds and sloughs

Vegetated tidal ponds and sloughs are highly productive systems subject to occasional to regular inundation. This habitat class extends from below mean high tide level to highest storm surge level. On Prince of Wales, these sites are found on both karst and non-karst; have 2-35% water cover; some host fish, newts, and/or toads; and have soils composed of muck, silt, sand, gravel, and cobbles. Our survey sites were characterized by brackish meadows with small to large pools, ponds, streams, and sloughs (Figure 13).

Wetlands in estuarine areas had no emergent or floating vegetation. Submerged plants include *Ruppia maritima*, and *Stuckenia filiformis*. Tree canopy cover ranged from 0-25%, and includes *Picea sitchensis*, *Tsuga mertensiana*, *Thuja plicata*, *Malus fusca*, *Pinus contorta*, and *Alnus rubra*. Shrubs were typically not present. Numerous graminoid and forb species were present, including *Carex* spp., *Deschampsia* sp., *Trisetum cernuum*, *Hordeum bracyantherum*, *Leymus mollis*, *Festuca rubra*, and *Bromus sitchensis*.

Figure 13. Tidal ponds and sloughs at sites C1.5-012, C3.1-001, and C3.8-013 (left to right).

Anthropogenic

The predominant feature in anthropogenic habitats is disturbance by humans. The parallel classification from Boggs et al. (2013) describes this as the Urban, Agriculture, or Road Class, with at least 50% of the area dominated by one of these disturbance types. In our surveys, these sites include rock quarries and disturbed roadside habitats. Limestone gravel pits frequently hosted newts and numerous aquatic invertebrates. Anthropogenic sites were located both on and off karst, and include waterbodies both with and without aquatic vegetation and algae. Terrestrial vegetation cover is highly variable (Figure 14).

Dominant tree cover includes *Picea sitchensis*, *Thuja plicata*, *Alnus rubra*, *Tsuga heterophylla*, and *Tsuga mertensiana*, with a total canopy cover of 2-45%. Shrubs include *Rubus spectabilis*, *Rubus parviflorus*, *Ribes laxiflorum*, and *Vaccinium ovalifolium*. Herbaceous and graminoid cover typically includes non-native species such as *Phalaris arundinacea*, *Phleum pratense*, *Taraxacum officinale*, *Hieracium aurantiacum*, *Leucanthemum vulgare*, and *Ranunculus repens*, as well as native *Lysichiton americanus*, *Equisetum* spp., *Carex* spp., *Bromus sitchensis*, and *Poa palustris*. Submerged *Chara* spp. (algae) and emergent *Equisetum* spp. were also present.

Figure 14. Anthropogenic site ANT-001 with a pool formed by a road embankment and draining through a culvert (left). Rock quarry at site C1.5-001 with common characteristics of anthropogenic quarry sites such as sheer cliff walls, cobble substrate, human detritus, and standing pools of water (right).

NWI classifications: ground truthing

The habitat classification system used in our surveys was based on a combination of the National Wetland Inventory (NWI) and the Boggs et al. (2013) classification maps, which we modified to specifically meet the needs of our study. During data collection in the field, we noticed that the wetland types delineated by NWI did not always match empirical observations. To reconcile these differences we used ArcGIS 10.1 to compare habitat classifications we expected to encounter (NWI) and those actually observed (empirical). A 50 m buffer was placed around each of our survey points to account for the total area surveyed at each site. This buffer was intersected with the NWI data layer and the NWI classes were extracted from buffered areas for comparison. Table 8 summarizes differences between habitat classes observed in the field and habitat classes designated by the NWI.

Table 8. Differences in habitat classifications between the National Wetland Inventory (NWI) and empirical observations. Grey cells indicate where the observed habitat class should correspond to NWI habitat class. The numbers in each cell indicate the count of sites observed that correlate with each NWI classification. Therefore, grey cells with a number value reflect affirmative ground truthing of NWI classes, while numbers in non-shaded (white cells) reflect empirical observations that differed from NWI classifications. There was no comparable NWI class for beaver ponds and sloughs and anthropogenic sites.

	Expected (NWI) habitat class							
Observed habitat class	Estuarine and Marine Wetland	Freshwater Emergent Wetland	Freshwater Forested/Shrub Wetland	Lake	Freshwater Pond	Riverine	Other	None
Tidal ponds and sloughs (n=7)	6	1						
Palustrine (n=19)		9	6	1	3			
Herbaceous peatlands (n=19)		12	5		2			
Needleleaf forested peatlands (n=46)		21	22	1	1			1
Seasonally flooded forested wetland (no waterbodies; n=18)		2	12	1				3
Seasonally flooded forested wetland (with waterbodies; n=21)		6	8					7
Riverine lower perennial (n=8)	1	1	3		1			2
Lacustrine littoral (n=31)		3	6	12	10			
Anthropogenic (n=8)	--	--	--	--	--	--	--	--
Beaver ponds and sloughs (n=26)	--	--	--	--	--	--	--	--

The observed habitat class “tidal ponds and sloughs” matched well with the NWI’s “estuarine and marine wetlands”, as coastal zones are relatively immutable. The observed “palustrine” wetlands most frequently correlated with the NWI’s “freshwater emergent wetlands” but was highly variable and was also represented by the NWI’s “forested/shrub wetlands,” “freshwater ponds,” and “lakes”. Our habitat type of “herbaceous peatlands” frequently correlated with “freshwater emergent wetlands”, as we expected. Although “needleleaf forested peatlands” were expected to match the NWI’s “freshwater forested/shrub wetland”, this was only the case about 50% of the time, as it was also frequently represented by “freshwater emergent wetland” delineations. “Seasonally flooded forested wetlands (no waterbodies)” most often matched appropriately with “freshwater forested/shrub wetlands”, while those “with water bodies” had little correlation with the NWI classes, often showing up as non-wetlands. Similarly, our “riverine lower perennial” habitat types had a weak correlation with NWI classes. Sites we classified as “lacustrine littoral” most often matched up with the NWI’s “lakes” and “ponds.” Discrepancies between our empirical observation and NWI delineations are likely due to successional changes over time and/or a lack of ground truthing by the NWI.

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Appendix 2: Amphibian life stages and photos in habitat

Rough-skinned newts (*Taricha granulosa*)

Figure 15, Figure 16, and Figure 17 show the larvae, metamorph, and adult stages of rough-skinned newts that we observed in 2013 and 2014 on Prince of Wales Island and list distinguishing traits between the life stages.

Figure 15. Rough-skinned newt larva captured on July 9, 2013. Larvae are distinguished from metamorphs by the presence of external gills, non-functional legs, and semi-translucent skin.

Figure 16. Rough-skinned newt metamorph captured on July 7, 2013. Metamorphs are distinguished from larvae by the lack of external gills, presence of functional legs, and nearly opaque skin. Metamorphs are distinguished from adults by the presence of smooth skin, poorly differentiated dorsal and ventral coloration, and tail with large skin margins.

Figure 17. Rough-skinned newt adult captured on July 8, 2013. Adults are distinguished from metamorphs by the presence of small bumps on the skin, strongly differentiated dorsal and ventral coloration, and tail with minimal skin margins.

Western toads (*Anaxyrus boreas*)

Figure 18, Figure 19, and Figure 20 show the tadpole, metamorph, and adult and subadult stages of western toads that we observed in 2013 and 2014 on Prince of Wales Island and list distinguishing traits between the life stages.

Figure 18. Western toad tadpoles captured on July 15, 2013 (left) and July 10, 2013 (right). Tadpoles are distinguished from metamorphs by the absence of fore-legs, poorly developed hind-legs, and tail with large skin margins.

Figure 19. Western toad metamorphs captured on July 15, 2013. Metamorphs are distinguished from tadpoles by the presence of fore-legs, well-developed hind-legs, and tail without large skin margins. Metamorphs are distinguished from adults by the presence of tails.

Figure 20. Western toad adults vary considerably in size and coloration, from the small subadult captured on July 13, 2013 (left) to the large adult captured on July 9, 2013 (right). Adults are distinguished from metamorphs by the lack of tails.

Appendix 3: Amphibians and non-native vegetation

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Introduction

The establishment, growth, and persistence of non-native[1] plant species pose a serious threat to native ecosystems. Even though not all non-native species cause significant economic or ecological harm, a small portion of these plants may be invasive[2] and may significantly alter community composition, successional pathways, nutrient cycling, hydrology, and fire regimes, and can also reduce or eliminate threatened and endangered native species populations (Ehrenfeld 2011).

While invasive plants constitute a major problem in the lower 48 states (Randall 1996), Alaska has remained much less affected. However, in recent decades there has been a marked acceleration in the rate of introduction of non-native plants to the state, probably driven by increases in population, commerce, development, gardening, and outdoor recreation activities (Carlson and Shephard 2007). Invasive species have become costly in Alaska, with an annual average of $5.8 million spent between 2007 and 2011 on management, prevention, education, restoration, monitoring, and research (Schwörer et al. 2012).

The susceptibility of native ecosystems to invasion is largely a function of the degree of natural or anthropogenic disturbance (Hobbs and Huenneke 1992). In Alaska, non-native plant occurrence is most strongly correlated with high-use, and therefore, highly disturbed areas such as urban centers and transportation routes (Carlson et al. 2014). Their abundance declines rapidly off trail and road corridors (Bella 2011). As native plant species are eliminated from an area (e.g. by logging and roads), habitats are opened up for more opportunistic species. Invasive plants establish in these types of areas because there are more opportunities for introduction, less competition from native plants, and plenty of disturbed substrates on which invasive plants thrive. Consequently, as disturbances take place, the chances that invasive species will be introduced and successfully establish increases (Byers 2002). In addition to disturbed and reclaimed areas, other habitats most vulnerable to plant invasions in Southeast Alaska include islands, ports, and waterways. Ports are continually exposed to ships and cargo from outside areas, and as such are prone to invasive propagule introduction by these vectors. Waterbodies and corridors of both salt and freshwater are susceptible to aquatic invaders of both flora and fauna (Sheley and Petroff 1999, Bella 2003).

In addition to direct anthropogenic factors, climate change may also affect non-native plant establishment (Carlson et al. 2014). At higher latitudes climate change is more pronounced (Holland and Bitz 2003), which may lead to a greater rate of non-native species establishment and accelerated population growth in the future. Non-native species are often more adaptable and better competitors relative to native species (Prentis et al. 2008), and are likely to have an advantage with changing weather, temperature, and disturbance patterns. Native

[1] Non-native plants are those whose presence in a given area is due to the accidental or intentional introduction by humans.

[2] Invasive plants are non-native plants that produce viable offspring in large numbers and have the potential to establish and spread in natural areas.

species have slower migration rates (Malcolm et al. 2002) and are likely to lag behind invasive species in their response to environmental changes.

Prince of Wales Island in Southeast Alaska has undergone extensive timber harvest, and areas disturbed by logging and logging access roads are highly likely to harbor non-native plants. Logged forests in remote locations can potentially provide opportunities for weeds to spread from urban centers to more remote areas and to develop large population sizes that then facilitate establishment in adjacent natural ecosystems. Logging projects inherently have a high rate of canopy clearing and soil disturbance, aiding non-native plants in establishing self-perpetuating populations. In this and other studies (e.g. Arhangelsky 2005) on Prince of Wales Island, invasive weeds have been documented moving off the human footprint into natural ecosystems. Since timber harvest on the island has often targeted highly productive karst habitats, and invasive plants follow logging, karst habitats may be particularly susceptible to invasion. Moreover, reseeding - even with native taxa - can initially make it difficult for locally established native populations to compete with seeded species; as natives have trouble establishing, invasives move in (Borchett 2004).

We were unable to ascertain the exact seed mix used on Prince of Wales logging roads, but elsewhere in the Tongass National Forest reseeding mixes used in the 1970s and 1980s included perennial ryegrass (*Lolium perenne*), alta fescue (*Festuca arundinacea* [*sic.*] current accepted taxonomy *Schedonorus arundinaceus*), reed canarygrass (*Phalaris arundincea*), white clover (*Trifolium repens*), and red clover (*Trifolium pratense*; Arhangelsky 2005), all of which are non-native. Reed canarygrass is used because it is quick to establish and provides excellent erosion control due to its sod-forming growth habit. It grows exceedingly well across the island and was often observed moving off the human footprint. Furthermore, it grows tall enough to impede visibility along roadsides, obscuring signage and road shoulders, creating a driving hazard around tight corners and has known negative impacts on amphibians. More recently reed canarygrass has been eliminated from reseeding mixes used around Coffman Cove on Prince of Wales, and instead perennial ryegrass is used, although this species is also non-native (Arhangelsky 2005).

Previous non-native plant studies on Prince of Wales

In 2005, Turnstone Environmental conducted a vegetation inventory of 553 miles of state, local, and Forest Service roads on Prince of Wales Island, totaling 2635 sites. They surveyed 50 x 8 m transects every quarter mile along roads, and additionally surveyed intersections, pull-outs, recreation sites, parking areas, and rock pits, with survey sites typically 0.1 – 4 acres. They found a total of 62 non-native taxa previously known to occur in Alaska, and only six sites were weed-free. Highest species diversity was observed at rock pits, along paved state roads, and in towns and residential areas; diversity of exotic species declined with distance to these habitat types. Similarly, on Forest Service roads, the highest diversity of weeds was observed at their intersection with paved state roads and declined away from main roads. A number of species were widespread throughout the island, most prevalent (>30% of sites) were reed canarygrass (*Phalaris arundincea*), common plantain (*Plantago major*), dandelion (*Taraxacum officinale*), white clover (*Trifolium repens*), and mouse ear chickweed (*Cerastium fontanum*). The high occurrence and densities of reed canarygrass and white clover is attributable to previous seeding of the roadways with these taxa. Orange hawkweed (*Hieracium aurantiacum*), and to a lesser extent hairy cat's ear (*Hypochaeris radicata*), were also present at high densities. Among the most aggressive species documented, reed canarygrass was at the top of the list, moving off roadways and into adjacent habitats, establishing along waterways under forest canopy, penetrating intact muskegs and forests, and outcompeting native vegetation. Clover species (*Trifolium* spp.), particularly white clover (*Trifolium repens*), were also seen moving off roadsides into adjacent habitats, establishing in muskegs and forests, and outcompeting natives. Canada thistle (*Cirsium arvense*) and orange hawkweed (*Hieracium caespitosum*) were observed outside of anthropogenically influenced areas. Common timothy (*Phleum pratense*) was also documented moving off roadways into neighboring intact habitats, as was bull thistle (*Cirsium vulgare*; although only encroaching on logged areas and open slopes), and hairy cat's ear (*Hypochaeris radicata*; seen growing in shady, albeit disturbed habitats; Arhangelsky 2005).

Another study conducted by the Forest Service State & Private Forestry and the National Forest System surveyed roadsides throughout Southeast Alaska from 2005 to 2007 (Lamb and Shephard 2007). Similar to Turnstone's study, roadsides were surveyed every quarter mile, in addition to surveying all campgrounds, rock pits, and high use areas. Invasive plants found included spotted knapweed (*Centaurea stoebe*), orange hawkweed (*Hieracium aurantiacum*), meadow hawkweed (*Hieracium caespitosum*), tansy ragwort (*Senecio jacobaea*), bull thistle (*Cirsium vulgare*), common tansy (*Tanacetum vulgare*), perennial sowthistle (*Sonchus arvensis*), hairy cat's ear (*Hypochaeris radicata*), Scotch broom (*Cytisus scoparius*), black medic (*Medicago lupulina*), reed canarygrass (*Phalaris arundinacea*), and common St. John's wort (*Hypericum perforatum*; Lamb and Shephard 2007).

In 2010 Kendrick Bay and Dotson Ridge on the south end of Prince of Wales were surveyed in conjunction with the Ucore Bokan Mountain mining environmental assessment (No author 2013). Their study only found non-natives along roadways. Relatively few taxa were observed: mouse-ear chickweed (*Cerastium fontanum*), oxeye daisy (*Leucanthemum vulgare*), black medic (*Medicago lupulina*), reed canarygrass (*Phalaris arundinacea*), common plantain (*Plantago major*), common dandelion (*Taraxacum officinale*), and white clover (*Trifolium repens*; no author 2013).

A limited amount of invasive plant control work has taken place on Prince of Wales. A 2006 effort targeted white sweetclover (*Melilotus albus*) and bull thistle (*Cirsium vulgare*), and in approximately one month removed these taxa from about 10 acres of land. The work was carried out by Serve Alaska Guidance Association (SAGA) crews and Community Connections, an organization that helps find employment for people with disabilities. This project was funded by USDA Forest Service Centennial funds and grants from SAGA and Community Connections (No author 2006). In 2015, the Alaska Department of Transportation plans to apply herbicide along parts of state owned right-of-ways near Thorne Bay, in addition to mowing and potentially handpulling weeds (Jenkins 2014, Viechnicki 2014).

Scope of vegetation and non-native plant assessment in the present study

During 2013 amphibian surveys on Prince of Wales Island, we noted the high incidence of invasive plant taxa, particularly conspicuous orange hawkweed and reed canarygrass, along roadsides. During the 2013 field season, vegetation data were typically only collected at sites where amphibians were encountered and invasive plants were generally recorded only as incidental occurrences and not incorporated into the habitat data collection protocol. In 2014, during the second year of surveys, we systematically recorded a complete vascular plant list at each survey site, including non-native taxa, even if amphibians were absent. When this was not possible due to limited time and/or personnel, we documented at least the dominant vegetation observed.

Collecting a full vascular plant species list of the existing vegetation at each site allowed us to assess habitat preferences by amphibians, as well as contribute data to ongoing research to better understand Alaska's rare plants and ecosystems (e.g. old growth karst); for more information see http://aknhp.uaa.alaska.edu/ecology/ecosystems-conservation-concern). This also allowed us to add records to the Alaska Exotic Plants Information Clearinghouse (AKEPIC; aknhp.uaa.alaska.edu/akepic), the state-wide database for non-native plant occurrences. Furthermore, we hoped that comparing amphibian occurrence with invasive plant populations might elucidate trends between these two ecosystem components. Figure 21 shows where survey sites, amphibians, and invasive plants intersected in our study area.

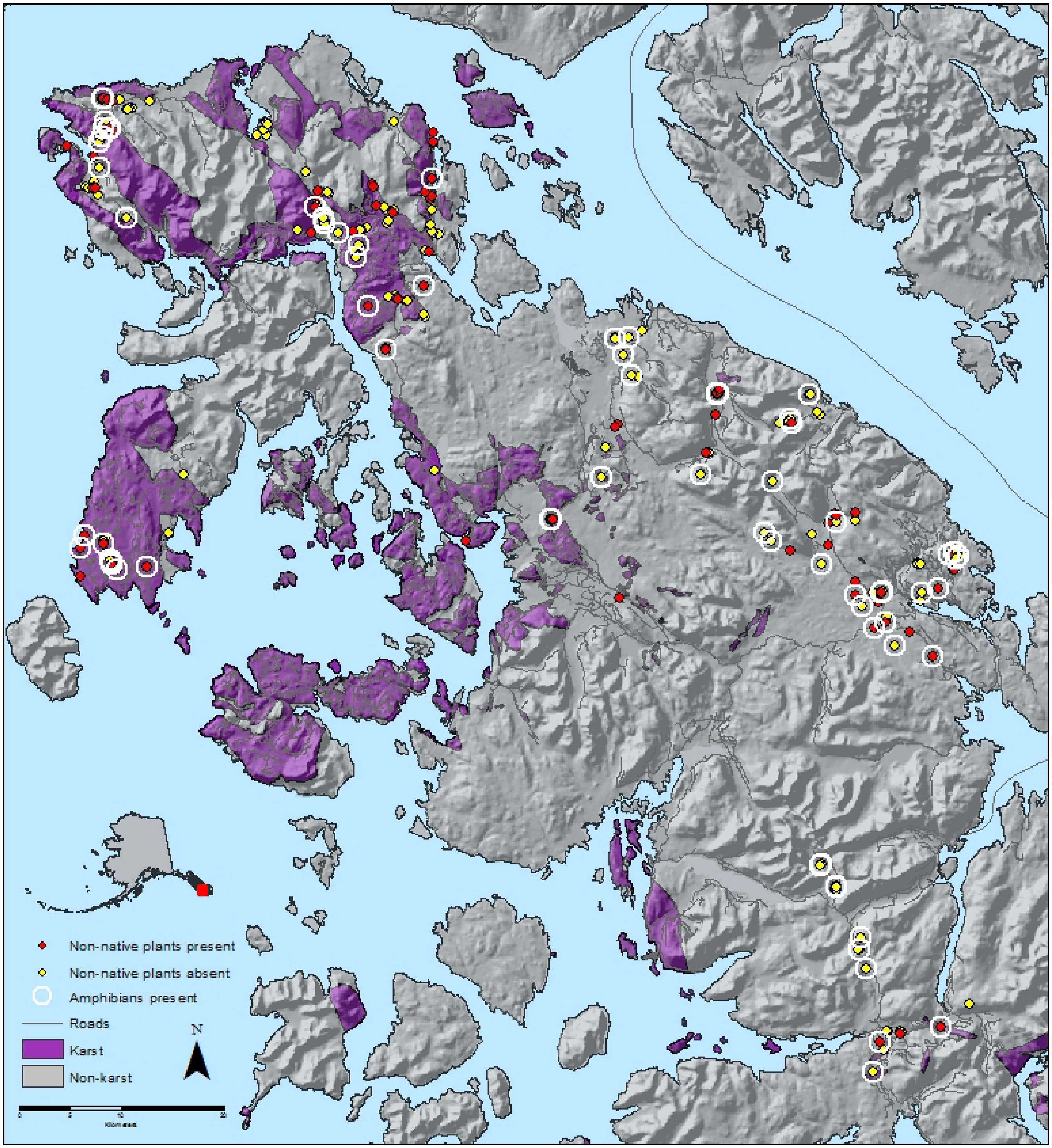

Figure 21. A comparison of amphibian and non-native plant presence in karst and non-karst environments, Prince of Wales Island 2013 and 2014.

Results

Of the 239 sites surveyed, 149 (62%) were weed-free, 73 (31%) had non-native plants, and at 17 (7%) the presence or absence of weeds was not recorded. Non-native plants were documented at an additional seven locations, although these were not official amphibian survey sites and were not included in further data analysis. A total of 13 non-native plant species were recorded during surveys (Table 9). Reed canarygrass was the most abundant and widespread; this taxon also has the highest Invasiveness Rank[3] of all species documented on the island. Common dandelion and orange hawkweed were also frequently observed, while white clover and creeping buttercup were observed at high densities, as indicated by relative biomass (infestation size multiplied by percent cover; Table 9).

Table 9. Non-native plants found on Prince of Wales Island at amphibian survey sites 2013 and 2014.

Common name	Scientific name	Invasiveness rank	Number of sites[a]	Infested acres	Biomass[b]	Percent frequency[c]
bull thistle	*Cirsium vulgare*	61	2	0.01	0.05	1.68%
purple foxglove	*Digitalis purpurea*	51	1	<0.01	0.01	0.84%
Japanese knotweed	*Fallopia japonica*	87	1	<0.01	<0.01	0.84%
orange hawkweed	*Hieracium aurantiacum*	79	11	1.98	0.95	9.24%
oxeye daisy	*Leucanthemum vulgare*	61	6	6.75	0.08	5.04%
reed canarygrass	*Phalaris arundinacea*	83	70	48.88	371.94	58.82%
timothy	*Phleum pratense*	54	1	0.15	0.15	0.84%
common plantain	*Plantago major*	44	4	1.80	0.041	3.36%
tall buttercup	*Ranunculus acris*	54	1	0.20	0.02	0.84%
creeping buttercup	*Ranunculus repens*	54	4	2.40	1.36	3.36%
common dandelion	*Taraxacum officinale*	58	13	9.60	0.22	10.92%
white clover	*Trifolium repens*	59	4	3.95	2.50	3.36%
bird vetch	*Vicia cracca* ssp. *cracca*	73	1	<0.01	0.01	0.84%
None	None	--	149	--	--	--
Total			268	75.73		

[a] Indicates how many sites each taxa was found at; the total number is higher than the number of sites surveyed because many sites hosted more than one invasive species.
[b] Infested area multiplied by percent canopy cover.
[c] Indicates relative frequency of each species out of all sites with non-natives.

Results from our study varied from those previously conducted on Prince of Wales roadsides. Similar to Turnstone Environmental (Arhangelsky 2005), we found about 30% of all sites surveyed had reed canarygrass, while their surveys showed additional taxa at ≥30% frequency that we did not detect, including common plantain, common dandelion, white clover, and mouse ear chickweed. Both studies found high occurrences and densities of orange hawkweed, while Turnstone found equally high biomass of hairy cat's ear that we did not

[3] Invasiveness Rank is calculated based on a species' ecological impacts, biological attributes, distribution, and response to control measures. The ranks are scaled from 0 to 100, with 0 representing a plant that poses no threat to natural ecosystems and 100 representing a species that poses a major threat to natural ecosystems (see Carlson et al. 2008 for more information).

observe. Both studies found reed canarygrass to be extremely aggressive and moving into otherwise intact ecosystems. Although we observed white clover at relatively few locations, it has a mat-forming growth habit, and where it was present it formed large, dense stands. Turnstone also recorded a lot of aggressive clover, and additional aggressive and widespread species that we did not detect in large numbers, such as thistles (*Cirsium* spp.), common timothy (*Phleum pratense*), and common plantain (*Plantago major*). When considering all invasive plant studies conducted on Prince of Wales, the fluctuation in number of taxa – from seven to 64 – reflects different survey methods and survey locations across the island. Our study found only 13 taxa. While we targeted a variety of wetland habitats, the other studies followed the road system and disturbed right-of-ways. The 13 non-native plant taxa we observed are likely well suited to wet environments and less anthropogenically influenced sites compared to weeds found in previous road-based surveys.

Non-native plants occurred at similar frequency (one third) at sites where western toads were detected, where rough-skinned newts were detected, and where no amphibians were detected (Table 10). Our study was designed to explore relationships between amphibians, karst, and non-karst habitats; to that end, we also looked at the relationship between non-native plants and areas underlain by karst features. Non-native taxa were found at 30% (n=49) of karst sites, and 32% (n=24) of non-karst sites.

Table 10. Incidence of non-native plant occurrence in relation to amphibian presence and absence, Prince of Wales Island 2013 and 2014.

	Total sites	Sites with non-native plants
Western toads	30	11 (37%)
Rough-skinned newts	59	19 (32%)
No amphibians	155	53 (34%)

In addition to considering overall presence of non-native species, reed canarygrass deserves special attention due to its known impacts on amphibians and abudance on Prince of Wales. Studies by Kapust et al. (2012) and Rittenhouse (2011) found that reed canarygrass degrades ovipositing habitat and decreases survival of some amphibian species, likely due to lowering dissolved oxygen in decomposition. We observed reed canarygrass at 70 survey sites; this was the only non-native plant found at 56 of these sites, and we often came across this grass in relatively remote locations, away from human disturbance. Of the 70 sites with reed canarygrass, 17 (24%) hosted rough-skinned newts, 10 (14%) had western toads, and at 45 (56%) amphibians were absent. Compared with amphibian presence across all sites (84 out of 239 sites surveyed, 35%), our data show little to no discernible reduction in amphibians where reed canarygrass is present. However, there is often a lag time between when an exotic plant is introduced and when the plant's negative impacts becoming apparent. Additionally, this grass was found most frequently at beaver ponds and sloughs (22 sites, 31%), indicating a strong relationship that may deserve future research. In all other habitat types, reed canarygrass was found at a relatively even distribution of 2-8 (3-11%) infestations per habitat class.

Overall, non-native plants were observed at 33% of survey sites, and relative frequency of introduced vegetation within each habitat type ranged from 6 to 100%. Invasive plants were most often seen at anthropogenic sites, and least often in needleleaf forested peatlands. Habitat types most invaded (>50% frequency) by non-native plants included anthropogenic, riverine lower perennial, and beaver ponds and sloughs. Average non-native plant biomass per site, as well as total biomass, was highest at beaver ponds and sloughs (Table 11).

We found that percent frequency and biomass of invasive plants were not necessarily correlated. For example, we found non-native species at every anthropogenic habitat type, but infestations were relatively small and limited to the disturbed area. The largest and most dense infestations occurred at beaver ponds, lake shores, along river corridors, and in seasonally flooded forests. Presence of invasive plants did not appear to have a direct impact on amphibian presence. For example, at anthropogenic sites there seemed to be a positive correlation, and at riverine sites there seemed to be a negative correlation between the two.

Table 11. Comparison of non-native plant presence and biomass with amphibian presence. Biomass is a relative number calculated by multiplying infestation size by percent cover. Percentages may not always total 100 because some sites hosted both newts and toads.

	Non-native plants present	Non-native plant biomass, total	Non-native plant biomass, average	Western toads	Rough-skinned newts	Amphibians absent
Anthropogenic	8 (100%)	8.106	0.45 ± 0.933	2 (25%)	6 (75%)	1 (13%)
Beaver ponds and sloughs	22 (59%)	189.178	4.399 ± 12.224	2 (5%)	6 (15%)	32 (80%)
Herbaceous peatlands	2 (11%)	0.011	.001 ± 0.002	3 (16%)	6 (32%)	11 (58%)
Lacustrine littoral	5 (16%)	85.105	2.432 ± 12.467	4 (13%)	8 (26%)	19 (61%)
Needleleaf forest peatlands	3 (6%)	3.501	0.073 ± 0.382	7 (15%)	17 (35%)	26 (54%)
Palustrine	5 (26%)	2.681	0.117 ± 0.358	4 (21%)	6 (32%)	10 (53%)
Riverine	7 (88%)	19.75	2.469 ± 2.728	1 (12%)	0 (0%)	7 (88%)
Seasonally flooded forest wetlands (no waterbodies)	7 (39%)	25.756	1.12 ± 3.169	2 (11%)	1 (6%)	15 (83%)
Seasonally flooded forest wetlands (with waterbodies)	8 (38%)	39.66	1.724 ± 6.086	2 (10%)	4 (19%)	15 (71%)
Tidal ponds and sloughs	3 (43%)	1.381	0.115 ± 0.279	2 (29%)	1 (14%)	4 (57%)
All sites	73 (33%)	339.641	1.408 ± 7.148	29 (13%)	58 (26%)	140 (63%)

Discussion

Next to human development, invasive species are the second leading cause of habitat loss globally. While previous non-native plant studies on Prince of Wales Island focused on right-of-ways and disturbed habitats, we explored their distribution and abundance in numerous wetland types across the island, often off the human footprint. The data presented here contribute to our knowledge of habitat invasibility and the status of some under-studied ecosystems in remote parts of Southeast Alaska. Our data analysis primarily looked at the frequency of occurrence, more so than intra- and inter-population dynamics, between amphibians and non-native plants. Future studies could further explore relationships between population size, density, life stages, etc. to elucidate trends not apparent through a frequency comparison alone.

In summary, we observed 13 species of non-native plants at about one-third of all sites surveyed. Non-native plants were also found at this same frequency at sites supporting western toads, rough-skinned newts, and sites without amphibian detections. Non-native plants were also observed at about one-third of karst and non-karst sites. Reed canarygrass was the most widespread and abundant non-native documented here and in previous studies on the island. The most invaded landscapes were anthropogenic, riverine lower perennial, and beaver ponds and sloughs. There seems to be little correlation between non-native plant frequency, overall biomass, and amphibian presence. However, there is often a lag time between introduction and measurable impacts of non-native species. Monitoring and management of invasive vegetation, particularly in wetland environments, is vital to limit their spread, help maintain amphibian and other native populations, and to prevent potential deleterious effects in the future.

Implications of non-native plants on amphibian ecology

Plant detritus from emergent and surrounding vegetation has an important impact on aquatic and semiaquatic ecosystems. Changes in chemistry of detritus can alter habitat, including water chemistry, decomposition, and the availability of zooplankton, periphyton, and algae (Maerz et al. 2005). Invasive plants can also impact disturbance regimes, nutrient dynamics, decomposition, physical and chemical habitat features, trophic structure, life history traits such as breeding and calling behavior, and can create toxicity in habitats (e.g. tannins and phenolic compounds). Additionally, invasive plants and climate change can have synergistic effects on wetland habitats (Cotten et al. 2012, Saenz et al. 2013).

Amphibians are susceptible to the concepts of the “ecological trap” or “evolutionary trap.” In the former, a species chooses to settle in low quality habitats due to environmental changes or stressors; in this case, the presence of invasive plants degrades high-quality amphibian habitat and leads them to settle elsewhere. In the latter, exotic species invade at such a quick rate that native species are unable to keep up with the changing environment, leading to declines in native populations (Cotten et al. 2012). Exotic plants can have confounding effects with climate change; new weather patterns can alter the phenology of amphibians, and often favor the establishment and spread of exotic species (Saenz et al. 2013). Overall, the effects of invasive plants on ecosystems can potentially remain cryptic for many years, until a threshold is reached in which changes to structure or populations become apparent and measurable (Brown et al. 2006).

The primary mechanism for non-native vegetation’s detrimental effect on aquatic organisms is changing water chemistry (Saenz et al. 2013). Plants produce water soluble phytochemicals (secondary compounds), phenolic compounds (tannins), and allelopathic chemicals (Watling et al. 2011). These chemicals serve a purpose in the life of a plant, such as defending against herbivory or inhibiting the growth of other vegetation in order to ensure their own success (Brown et al. 2006). However, these can become bioavailable to amphibians and other fauna (Watling et al. 2011). Plant compounds leach into water and aquatic species are immersed in them; they pass over gills and may be ingested during feeding. Amphibians are highly sensitive to phenols (Kerby et al. 2010), which are known to cause direct and indirect impacts to aquatic organisms. Tannins bind with proteins, inhibiting digestion of proteins by tadpoles (Brown et al. 2006), and they interfere with digestive enzymes and reduce nutritional value of food. Saponins and tannins cause gill lesions in fish (Temmink et al. 1989). Similar to fish, some amphibians, such as certain species of toad tadpoles, are obligate gill breathers and can suffer the same gill damage and subsequent asphyxiation as is known to occur in fish (Maerz et al. 2005). Some plant chemicals (e.g. emodin) are known to cause malformation in amphibians. Chemical changes precipitated by introduced vegetation have the greatest effect on small waterbodies, such as ephemeral pools, which are often used by amphibians as breeding habitat (Lincoln Park Zool 2013, Watling et al. 2011).

Plant chemicals may affect trophic systems. They can inhibit the growth of periphyton, phytoplankton, zooplankton, algae, and microorganisms that tadpoles feed on. Changes to the available quantity and quality of food may affect tadpole success and consequently have bottom-up impacts on local trophic relationships (Maerz et al. 2005).

Phenology is important for amphibians, as their development is synchronized with their local environment and food sources. Non-native plants often have different phenology and decomposition rates relative to native vegetation and can alter phenology in several ways. Tannins can inhibit the activity of aquatic invertebrates (reducers), consequently altering decomposition rates and nutrient cycles, affecting the timing of nutrient pulses into waterbodies. Phenological changes can interfere with amphibian feeding habits and availability (Cameron and LaPoint 1978, Saenz et al. 2013, Brown et al. 2006). Some introduced flora can speed up the drying of ephemeral pools and streams by increasing transpiration (Bucciarelli et al. 2014, Boyce et al. 2012).

The timing of leaf fall and quantity of biomass from invasive species can have strong effects on dissolved oxygen levels. Changes in dissolved oxygen can impact survival, reproductive capacity, activity, growth, and development of many species (Saenz et al. 2013). An adaptation to low dissolved oxygen is to increase visits to surface waters, where there is more oxygen. However, amphibians often limit their movement to avoid predators; if they are forced by a lack of oxygen to make more trips to the surface, they subject themselves to increased risk of predation. More frequent surfacing, in addition to raising predation risk, limits the amount of time spent engaging in other activities, like foraging (Watling et al. 2011). A lack of dissolved oxygen, and aforementioned damage to gills by phenols, is a major concern for species that are obligate gill breathers early in life (Maerz et al. 2005).

Purple loosestrife (*Lythrum salicaria*) has been known to occur in Alaska and can negatively impact amphibians. It tolerates inundation and thrives in littoral zones used by amphibians. Purple loosestrife drops its leaves in the fall and they decompose more rapidly than native species, while stems of loosestrife take a long time to breakdown and can add a large amount of standing litter to a wetland. The leaves of this herbaceous plant have high quantities of tannins, and when introduced to a wetland can reduce microbial conditioning of detritus, which in turn affects decomposition and detritivore communities (Brown et al. 2006, Maerz et al. 2005). Purple loosestrife changes sediment chemistry (Templer et al. 1998), decreases dissolved oxygen, and changes the diet of some species, such as American toads (*Bufo americanus*), consequently reducing fitness. Research on American toad larvae and tadpoles show in the presence of purple loosestrife, gill damage from phenolic compounds and decreased dissolved oxygen work in tandem to slow down development, increase mortality, and raise survival variability (Brown et al. 2006, Maerz et al. 2005).

Two species common to Southeast Alaska, garlic mustard (*Alliaria petiolata*) and Japanese knotweed (*Fallopia japonica*) are also culprits. The former has been implicated in the declines of terrestrial woodland salamanders (Maerz et al. 2009), and the latter reduces arthropod abundance, thereby reducing frog foraging (Maerz et al. 2005). Common reed (*Phragmites australis*) is not yet in Alaska, but is on our watch list, and also affects food availability and tadpole development (Bucciarelli et al. 2014); it has also caused loss of breeding habitat and declines in populations of Fowler's Toads (*Bufo fowleri*; Greenberg and Green 2013).

On Prince of Wales, we found reed canarygrass (*Phalaris arundinacea*) to be abundant and widespread throughout wetlands. This grass is known to degrade egg laying habitat and decrease survival for some amphibian species (e.g. the Oregon Spotted Frog, *Rana pretiosa*). There is apparently no direct toxic effects from this species; however, negative effects are likely due to decomposing grass causing anoxic environments (Kapust et al. 2012, Rittenhouse 2011).

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