

Project Title: Hydrologic & Ecological Monitoring of Montane Wetlands in North Cascades, Mount Rainier, and Olympic National Parks

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Note: Portions of this report are excerpted and condensed from Lee et al. in review and from our final report 2014 report to the Northwest Climate Science Center (available [here](#)). A full description of the broader wetlands & climate change research of which this report is a subset is available via the Northwest Climate Science Center website and public repository.

Project Abstract

Wetland ecosystems are ecologically important components of National Park landscapes. Montane wetlands may be particularly vulnerable to changing climates. Responsible and effective park protection of these areas relies on accurate inventories of sites, a detailed understanding of ecosystem functions and hydrologic cycles, and projections of changes based on future climates. Currently, parks have incomplete baseline inventories of montane wetlands and mainly qualitative information on hydroperiods of wetlands. The goals of this collaborative project were to collect hydrologic data to support the development of models, to collect GPS data to improve delineations of wetland maps, and to use these data to improve models projecting future wetland hydroperiods and function. The models will be used by park managers, scientists, and other interested parties to apply vulnerability assessments to park landscapes and develop adaptation strategies for future park management of these important components.

Introduction

Wetlands are ecologically important yet at-risk ecosystems (Dahl 1990, IPCC 2001, Comer & Goodin 2006, IPCC 2007). Wetland hydrology, structure, and function all respond dynamically to changes in temperature and precipitation such as those predicted for the coming century. As a result, wetlands are considered among the most sensitive ecosystems to climate change (Carpenter et al 1992, Poiani et al 1996, Burkett and Kusler 2000, IPCC 2001, 2007, Erwin 2009). Near-coastal areas of the Pacific Northwest are among the most sensitive regions in the western U.S. to climate change, amplifying consequences for wetlands (Cayan et al 2001, Mote 2003, Hamlet et al. 2005, Mote et al. 2005, Nolin and Daly 2006, Hamlet et al. 2007). Warming in all seasons, increasing precipitation in fall, winter, and spring, and decreasing precipitation in summer may all contribute to shifting patterns of wetland hydrology and resulting changes in ecological function (Mote and Salathé 2010, Lee et al. *in review*).

Within Washington's National Parks, montane wetlands are well protected from many human impacts, but may be uniquely vulnerable to climate change due to their reliance on diminishing snowpack (Mote et al. 2005, Hamlet et al. 2007). Effective assessment, management, and development of climate adaptation strategies for wetland ecosystems within North Cascades, Olympic, and Mount Rainier National Parks is not likely to be possible without accounting for climate change. The National Park Service itself has recognized this need and called for increased capacity in "forecasting climate changes at relevant scales" and "integrated assessment of impacts" in their *Climate Change Response Strategy*. However, until recently, few resources existed to incorporate climate change into wetlands management.

Our research group has been developing a suite of new scientific resources specific to North Cascades, Mount Rainier, and Olympic National Parks, which can be used to inform wetlands management in light of climate change. The major challenge in developing these products has been the severe dearth of empirical data to calibrate hydrologic models, validate wetland maps, and support vulnerability assessment for wetland classes and wetland-reliant species. To address this need, in 2012 we implemented hydrologic monitoring and collected new data on the ecology and distribution of wetlands within focal regions in each Park. In this project, we gathered a second season of hydrologic and ecological data (2013) to extend and refine our modeling and mapping methods and to enhance datasets for use in vulnerability assessment.

Methods

Funding for this project allowed us to hire a field team to work in North Cascades, Olympic, and Mount Rainier National Parks. We collected hydrologic data and ground-truthed areas of new wetland maps to improve remote sensing mapping and classification algorithms. We also conducted visual encounter surveys for amphibians to enhance the data resources used in vulnerability assessments of future climate impacts. While conducting visual encounter surveys, we gathered basic hydrologic data (e.g. wetland depth, surface area) and habitat attribute data (e.g. vegetation composition, substrate) at instrumented wetlands and all other wetlands (up to 50/site) within focal field areas. We focused on three regions per park. In North Cascades, we focused on wetlands in the Dagger Lake/Twisp Pass region, Pyramid Lake, and Big Beaver valley. In Olympic, we focused on wetlands around Deer Lake, Potholes, and 7 Lakes Basin. In Mount Rainier National Park, we focused on Mazama Ridge, Spray Park, and Palisades Lakes regions. The data collected during the project were combined with 2012 data collected using the same design.

Hydrologic monitoring

For our primary dataset used in developing the climate-hydrologic models presented below, we collected detailed data on wetland hydrology (wetland depths and spatial extent) for 121 montane wetlands in 2012, representing a mix of ephemeral, intermediate, perennial, and permanent ponds, through physical monitoring. In a subset of wetlands, we also estimated wetland depths using iButton temperature dataloggers. To do so, we installed iButtons along transects from the edge of the wetland to the deepest accessible point in the wetland. We identified the date at which wetland water levels dropped below each iButton based on changes in the variance in temperature (measured every two hours). Because air temperatures fluctuate more dramatically than water temperatures, it is possible to compare temperatures of iButtons along each transect to iButtons placed in the open air adjacent to the wetland to determine when the iButton was submerged or exposed to the air. Our physical depth measurements validated the estimates of water level derived from the iButton transects. This larger 2012 dataset supplemented several smaller historical datasets that included 1) measured wetland water depths for 7 montane wetlands in Seven Lakes Basin, Olympic National Park from the summer of 2000 and 2) wetland water volume estimates for 10 montane wetlands on Mazama Ridge, Mount Rainier National Park from June through September 1992 (Girdner & Larson, 1995). In 2013, we repeated hydrologic monitoring at these original sites to collect an additional year of data using the same methods. Models below are constructed using the 2012 data, and the 2013 data can be used to update these projections when the hydrologic (Variable Infiltration Capacity) model runs become available. (There is a ~2-year time lag in most cases in developing updated model runs.)

Amphibian monitoring: Sampling design, field surveys, and monitoring of breeding success

In the same suite of sites monitored for hydrologic change, we conducted amphibian visual encounter surveys in 2012 and 2013. Amphibian surveys focused on species presence and on key habitat attributes known or suspected to influence amphibian occupancy, habitat use (e.g. breeding versus foraging), and

recruitment. We developed protocols in accordance with the US Geological Survey Amphibian Research and Monitoring Initiative's approach. We focused on three common montane species: *Rana cascadae* (Cascades frogs), *Ambystoma gracile* (northwestern salamanders), and *Ambystoma macrodactylum* (long-toed salamanders). We noted any additional pond-breeding amphibian species of any life stage where they were present (*Taricha granulosa*, rough-skinned newt; *Rana luteiventris*, Columbia spotted frog; *Pseudacris regilla*, Pacific chorus frog; *Bufo boreas*, Western toad).

To conduct visual encounter surveys, two-person teams carefully walked the perimeter of each pond and checked all microhabitats for amphibians (e.g. in the pond, under banks, stream inlets, in submerged or adjacent terrestrial vegetation) between 08:00 and 20:00 hours. During each survey, field crews recorded pond coordinates and the presence of any amphibian life stage (eggs, larvae or tadpoles, metamorphs, juveniles, terrestrial adults, paedomorphs, or dead animals of any stage). Crews also recorded habitat attributes that may be associated with amphibian occupancy, including elevation, pond dimensions (length and width to estimate circumference), depth, wetland type (lake, pond, or wet meadow), hydrologic class (ephemeral, intermediate, perennial, permanent), fish presence, percent shallows (flooded habitat <0.5m in depth), presence of emergent vegetation, substrate, presence of cobble, presence of downed wood, presence of complex side habitat, percentage of surrounding area that was wooded, and dominant types of emergent and riparian vegetation. Crews also recorded additional data on environmental factors that might further influence amphibian detection, including date, time of day, sky conditions, wind conditions, air temperature, water temperature, water presence, water depth, water color, water transparency, presence of predatory birds or snakes or invertebrates, and percentage of the pond perimeter successfully searched.

The seasonal start to surveys was determined by ice-out of the ponds, which we tracked carefully at the beginning of each season via communication with National Park Service rangers and field crew reconnaissance. In 2012, surveys began in late June and concluded in late October. We surveyed each site for the presence of amphibians between one and six times, with the majority of sites receiving three or four surveys. A subset of our sites is part of a long-term demographic study of *Rana cascadae*, and we surveyed these sites up to six times. In 2013 surveys began in July and ended in late September. In 2013, we conducted a smaller number of visual encounter surveys and focused on tracking developmental rates of *Rana cascadae* tadpoles and evidence of mortality as ponds dried. To do so, we staged tadpoles according to their Gosner stage, and noted any evidence of mortality (e.g. dried egg masses or dried tadpoles). Amanda Kissel (PhD student at Simon Fraser University), with support of the same field crew from this project, intensively tracked *Rana cascadae* breeding effort and success, focusing on egg deposition, tadpole development, and tadpole survival or mortality in drying ponds. In all three years (2012-2014), our collective team from UW and SFU closely tracked developmental rates of *Rana cascadae* tadpoles and evidence of mortality of any species as ponds dried.

Analysis of Pond-breeding Amphibian Habitat Use

We have preliminarily analyzed amphibian visual encounter survey data from 2012, the year in which we had the greatest number of surveys, to relate amphibian presence and habitat use (breeding or adult/foraging) to our four hydrologic wetland classes and other habitat attributes. We focus on the three common focal species for which we have sufficient data for a meaningful analysis: *Rana cascadae*, *Ambystoma macrodactylum*, and *Ambystoma gracile*. Because we anticipate that some species use different habitats for breeding versus foraging, we analyzed our dataset in two ways, first looking at the presence of life stages that indicate breeding (eggs, larvae, and tadpoles), and second at the presence of adult stages of each species. Data collected in 2013 are being used to by Amanda Kissel to develop a demographic model of *Rana cascadae* for the Seven Lakes Basin population.

Using survey data, we constructed binomial logistic regression models to predict binary presence or absence of a) breeding evidence or b) adult life stages (terrestrial or aquatic) for the three focal species

from the 2012 dataset. Included in the analysis were individual ponds ($n = 219$) that we visited from 1 to 5 times each over the course of the breeding season. If we detected breeding efforts (egg masses or larval life stages) during any of those visits, this was coded as positive evidence of breeding. Our explanatory variables represent a suite of hydrologic and habitat variables: pond elevation (m), maximum pond size (circumference in m^2), maximum pond depth (m), pond hydroperiod (ordered factor, 4 levels: ephemeral, intermediate, perennial, permanent), presence of fish in the pond (binary factor), maximum percent of the pond that was shallows ($<0.5m$) during the breeding season, maximum percent of the pond occupied by emergent vegetation throughout the breeding season, presence of complex adjacent habitat that may be used by amphibians (binary factor), percent of the pond perimeter occupied by woods, substrate class (factor, 4 levels: muck, mud/clay/silt, sand/gravel, cobble/boulder), presence of cobbles in the substrate (binary factor), and presence of dead wood in the pond (binary factor). We examined all combinations of explanatory variables to test for co-variation and used principal component analysis (PCA) where necessary to transform strongly correlated variables into uncorrelated variables (PCA axes) for analysis. Due to the variety of survey efforts and types across the ponds, some ponds had to be dropped from the analysis due to data paucity, producing a smaller subset of ponds on which the analysis was performed ($n = 169$).

In order to avoid over-fitting, we limited the number of parameters included in models to 5 (including intercept; Burnham, Anderson 2002). We used Akaike's Information Criterion (Akaike 1974) to compare the degree of support for all combinations of 5-parameter models. This approach is generally used to explore data in the absence of any *a priori* hypothesis being considered more or less probable (Anderson, Burnham 2002). While we strongly suspected that hydroperiod and the presence of fish would be important drivers of amphibian adult presence and breeding, we used this approach to explore the data given the uncertain effects of most other variables (Symonds, Moussalli 2011). The results of this method can then be used to inform more traditional hypothesis tests, which we are conducting now and are a more appropriate way to check for interactions among parameters. These will be reported in future publications.

We performed analyses using *R* (v 3.0.2; R Core Team 2013), with the AIC analysis package *AICcmodavg* (Mazerolle 2013). We calculated Akaike weights (w_{Ak}) for each model (weight of support between 0 and 1, all w_{Ak} sum to 1), which can be interpreted as the probability that a given model is the best approximation (Symonds, Moussalli 2011). To estimate a 95% confidence set of models (a subset of candidate models that we are 95% sure contains the best model in the original set), we selected the top models whose cumulative w_{Ak} just surpassed 0.95 (Burnham, Anderson 2002). To compare the relative importance of individual variables, w_{Ak} for all models containing each variable were summed, resulting in a relative ranking of variable importance. The w_{Ak} was also used to calculate the weighted mean of variable coefficients across all models in which each variable was included. For visual ease of interpretation, we scaled and centered all variables (mean of 1, standard deviation of 1) so that their coefficient estimates are directly comparable and relative to their importance.

We are currently exploring these data with a number of additional approaches not reported here. Occupancy analyses are a common approach to amphibian habitat association studies, because they make it possible to estimate detection rates as well as species occupancy, thereby accounting for false absences in the data. Our dataset presents several challenges for occupancy analyses given the particular focus of our study. Most significantly, we are most interested in the relationship between hydrologic dynamics, species habitat use, and breeding success. However, due to the mechanics of occupancy modeling, sites that dry must be dropped from the analysis once dry. 2012 was a climate change analog year, hence many of our sites dried, which presents a number of methodological and interpretive challenges for the analysis. Additionally, many of our site-level attributes did not vary much over the course of the field seasons, which creates problems for the analysis of detection rates. Therefore, for the initial hydrologic assessments that are the focus of this study (i.e. to identify core relationships between species habitat use and pond hydrologic types), we use the methods described above, but moving forward are further

exploring the use of occupancy analysis to estimate variation in detection in relation to a range of habitat attributes that are shared among different hydrologic classes.

Synthesis of Climate-Hydrologic Models with Ecological Analyses

We evaluated the vulnerability to future climate impacts of our three focal species of pond-breeding amphibians (*Rana cascadae*, *Ambystoma macrodactylum*, *Ambystoma gracile*). Because these three species represent different life history and developmental requirements in the Cascade and Olympic mountains, we hypothesized that this variation would be associated with differential use of wetland habitats and therefore different levels of risk of climate-associated habitat loss.

Vulnerability to climate impacts combines sensitivity, exposure, and adaptive capacity (Glick et al. 2011). Sensitivity is a measure of whether or how a species or ecosystem is likely to be affected by climate change based on its biology and physiology (if a species) or other factors such as geographic location and associated processes (if an ecosystem). Exposure is the intensity of climate change impacts a species or ecosystem is likely to experience given where it lives or is located. Adaptive capacity is the range of ways a species or system might be buffered from climate impacts to reduce its sensitivity or exposure and enable it to cope without significant changes in viability or ecological function. Adaptive capacity includes biological responses such as migration, behavioral changes, or evolutionary adaptation. Also, in a management sense, adaptive capacity refers to opportunities to actively ameliorate impacts through conservation actions (Glick et al 2011).

We assessed sensitivity to climate change based on the strength of association of each focal species with the four wetland hydrologic classes, as determined by the ecological analyses above, and in relation to the relative vulnerability of each class of wetland to climate change, as determined by the climate-hydrologic models of future impacts.

To assess exposure, we overlaid the Variable Infiltration Capacity (VIC) hydrologic output maps, developed as part of the broader project of which this project is a subset. The VIC maps show changes in wetland drying rates by the 2080s with National Wetland Inventory hydrologic water regime modifier cross-walked to our wetland hydrologic classification. For example, we classified all wetlands with the NWI hydrologic modifier defined as “seasonally flooded” as intermediate wetlands. Seasonally flooded wetlands are defined as having “surface water present for extended periods especially early in the growing season, but absent by the end of the growing season in most years”. See the Northwest Climate Science Center report ([here](#)) for a detailed explanation of the NWI hydrologic modifiers and how they relate to our pond classifications. While the NWI is imperfect, this approach enables us to 1) relate the proportion of available habitats of different types within each VIC grid cell to projected changes in the level of climate-induced hydrologic risk, and 2) estimate changes in the distribution of wetland types within each VIC geographic cell by extrapolating from our focal site assessment of the proportion of ponds that will switch categories under future climates. We are in the process of conducting a separate analysis of Mount Rainier National Park where improved wetland mapping resources are available, not reported here.

Finally, we assessed management-related adaptive capacity by identifying regions where introduced fish are present and could potentially be removed as a means of restoring what appears to be more climate-resistant wetland habitat for amphibians (Ryan et al. 2014). Relatively little is known regarding biological responses of our three primary species to climate change. What information is available we discuss below.

Results (with overarching Northwest Climate Science Center project)

Below, we combine the results from our field surveys with existing data from 2012 to demonstrate the kinds of integrated resources that these data can provide. Beyond the survey results from 2013, existing resources used in presentation of these findings include: 1) Variable Infiltration Capacity (VIC) model projections of the difference in probability of drying for intermediate ponds (this can also be used as a proxy of risk to short-hydroperiod and perennial ponds as well) based on 2012 or earlier hydrologic surveys (VIC runs are not yet available for 2013), 2) locations of National Wetland Inventory wetlands mapped by the National Park Service (NPS), and cross-walked to our four hydrologic categories, and 3) distribution of introduced fish based on NPS records. Reports on the full suite of these related analyses can be found through our North Pacific Landscape Conservation Cooperative and the Northwest Climate Science Center reports (this report includes NPLCC findings and can be found [here](#)).

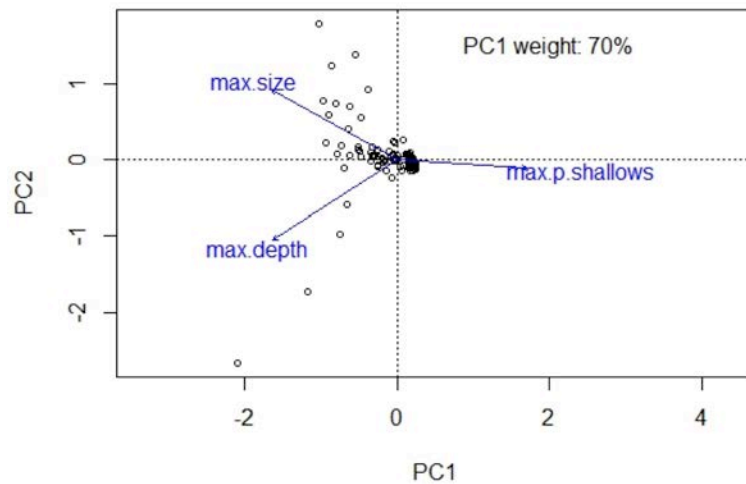


Figure 1. Above: *Rana cascadae* embryo within eggmass. Below: First and second axis of principal components analysis on pond max size, max depth, and percent shallows.

Wetland & Amphibian Analyses

Maximum pond depth was highly correlated with both maximum size (Spearman correlation coefficient = 0.80) and maximum percent shallows (-0.82), so we conducted a principal component analysis (PCA) on these three variables. The first PCA axis carried the majority of the loadings with a straightforward interpretation. Lower PCA values are bigger, deeper ponds with less shallows, while higher PCA values are smaller ponds with more shallows. We therefore used the first PCA axis in our logistic regression analyses in place of maximum pond depth, percent shallows, and maximum pond size.

Rana cascadae (Cascades frog): breeding evidence

For Cascades frog breeding evidence, AIC model selection produced a set of 55 models that held 95% of the weight from the candidate set of 210 models. No single top model was supported: $\Delta AIC_{max} = 8.26$, and maximum $w_{Ak} = 0.15$. Table 1 shows the top ranked models with $\Delta AIC_c < 4$. (ΔAIC_c of 2-4 is considered positive evidence of one model having a superior fit.) Hydroperiod class and elevation had the highest variable importance, as they were both included in the majority of top models (Figure 3.4.2). These were also the only two variables with significant parameter coefficient estimates, suggesting a higher incidence of Cascades frog breeding in intermediate and perennial ponds (relative to fast-drying ephemeral ponds) (Figure 2, Table 2). Fish had the third highest variable importance. The parameter

coefficient estimates suggest that, despite being non-significant (Table 3.4.2), this factor may have a large influence on whether or not *R. cascadia* breeds in a given pond. The lack of significance of this variable may be attributable to the small sample size of ponds with fish presence in this dataset ($n = 9/168$), all of which are in the permanent hydroperiod class.

Table 1. Top ranked models with $\Delta AIC_c < 4$ for *Rana cascadae* breeding evidence (BE). *elev* = elevation, *hydro* = hydroperiod, *emergent* = maximum percent of pond occupied by emergent vegetation, *fish* = fish presence, *side habitat* = presence of complex inlet or adjacent wetland habitat, *substrate* = pond bottom substrate, *PCA* = first axis of PCA, *cobble* = presence of cobbles in the substrate, *wooded* = percent of pond perimeter occupied by woods. For full set of top 55 models holding 95% of all model support, see Northwest Climate Science Center report (links above).

Rank	AIC	Model	ΔAIC_c	w_{Ak}
1	208.8	BE ~ <i>elev</i> + <i>hydro</i> + <i>emergent</i> + <i>fish</i>	0	0.15
2	209.4	BE ~ <i>elev</i> + <i>hydro</i> + <i>fish</i> + <i>side habitat</i>	0.56	0.11
3	210.9	BE ~ <i>elev</i> + <i>hydro</i> + <i>substrate</i> + <i>fish</i>	2.09	0.05
4	211.0	BE ~ <i>elev</i> + <i>hydro</i> + <i>PCA</i> + <i>emergent</i>	2.16	0.05
5	211.2	BE ~ <i>elev</i> + <i>hydro</i> + <i>cobble</i> + <i>fish</i>	2.39	0.04
6	211.2	BE ~ <i>elev</i> + <i>hydro</i> + <i>PCA</i> + <i>fish</i>	2.43	0.04
7	211.4	BE ~ <i>elev</i> + <i>hydro</i> + <i>wooded</i> + <i>fish</i>	2.57	0.04
8	211.6	BE ~ <i>elev</i> + <i>hydro</i> + <i>fish</i> + <i>downed wood</i>	2.75	0.04
9	211.6	BE ~ <i>elev</i> + <i>hydro</i> + <i>emergent</i> + <i>side habitat</i>	2.82	0.04
10	211.9	BE ~ <i>elev</i> + <i>hydro</i> + <i>PCA</i> + <i>side habitat</i>	3.11	0.04
11	212.1	BE ~ <i>hydro</i> + <i>emergent</i> + <i>fish</i> + <i>side habitat</i>	3.30	0.03
12	212.4	BE ~ <i>elev</i> + <i>hydro</i> + <i>emergent</i> + <i>downed wood</i>	3.60	0.02

Table 2. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels for *Rana cascadae* breeding. Variable codes are defined in Table 3.4.1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i> *	-1.79 (-3.20, -0.38)	0.717
<i>elevation</i> *	-0.47 (-0.87, -0.07)	0.204
<i>PCA</i>	0.34 (-0.24, 0.92)	0.296
<i>emergent</i>	0.34 (-0.04, 0.72)	0.193
<i>cobble(yes)</i>	-0.14 (-0.97, 0.69)	0.425
<i>wooded</i>	0.08 (-0.32, 0.49)	0.207
<i>fish(yes)</i>	-2.09 (-4.59, 0.42)	0.278
<i>hydro(intermediate)</i> *	2.14 (0.88, 3.41)	0.645
<i>hydro(perennial)</i> *	1.62 (0.50, 2.74)	0.569
<i>hydro(permanent)</i>	0.80 (-0.45, 2.06)	0.642
<i>side habitat(yes)</i>	0.57 (-0.26, 1.40)	0.423
<i>downed wood(yes)</i>	-0.29 (-1.14, 0.56)	0.434
<i>substrate(muck)</i>	1.48 (-0.18, 3.15)	0.850
<i>substrate(mud.clay.silt)</i>	0.96 (-0.71, 2.64)	0.852
<i>substrate(sand.gravel)</i>	1.75 (-0.34, 3.84)	1.067

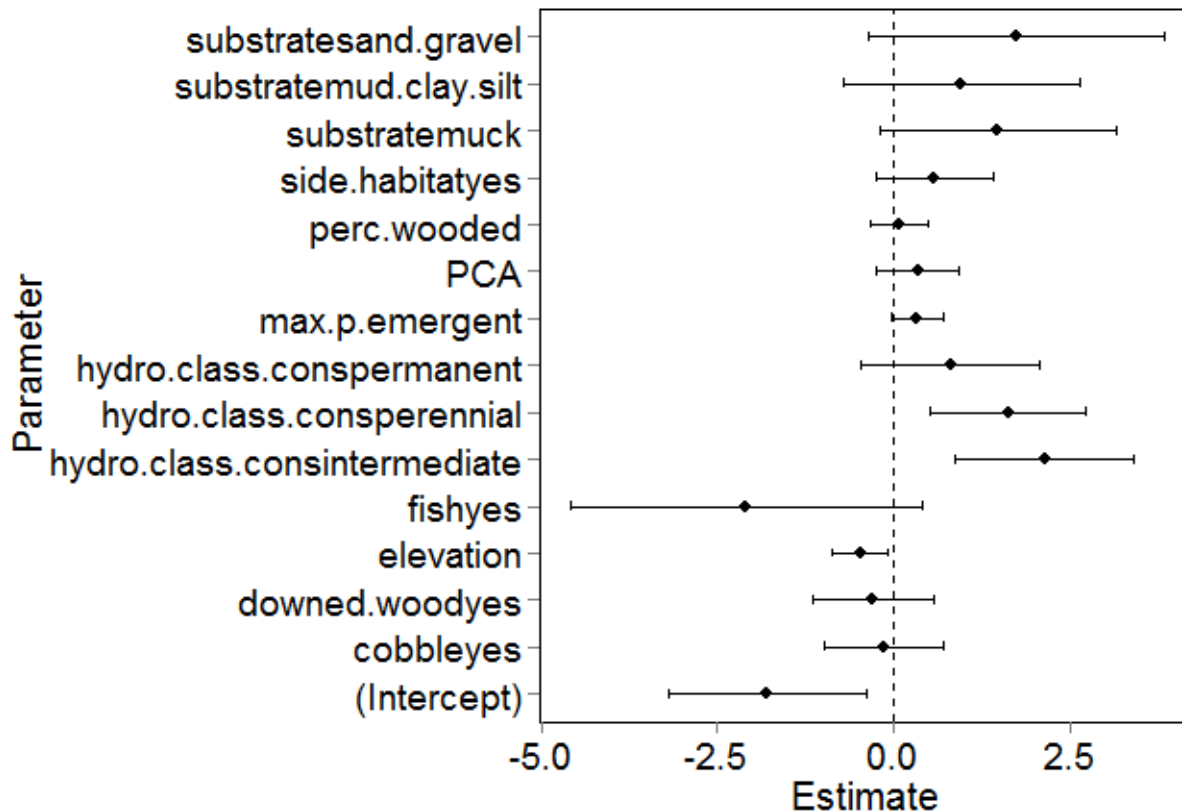
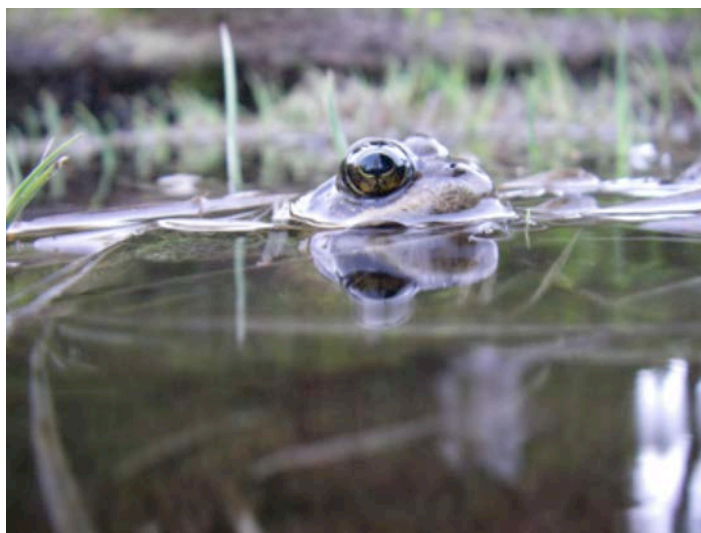


Figure 2. Scaled and centered model-averaged parameter coefficient estimates for the variables included in the set of models holding 95% of the weight of support based on Akaike weights for *Rana cascadae* breeding evidence. Hydro.class estimates are in reference to ephemeral wetlands.

Rana cascadae (Cascades frog): adult presence

For Cascades frog adult presence, AIC model selection produced a set of 11 models that held 95% of the weight from the candidate set of 210 models. The top model had relatively strong support (ΔAIC_c for second-ranked model = 2.83; $\Delta AIC_{max} = 8.59$; maximum $w_{Ak} = 0.59$; Table 3). The top model included hydroperiod class, the first PCA axis (a measure of pond shape or bathymetry), the percent of wooded perimeter, and fish presence, all of which had significant parameter coefficient estimates (Figures 4-5 and Table 4). Pond shape (PCA) and wooded perimeter had the highest



Rana cascadae adult in a perennial montane pond in Olympic National Park.

variable importance and were included in all top models, showing an association between Cascades frogs adults and smaller, shallower ponds (higher PCA values) with greater forest cover. These variables were followed in importance by fish and hydroperiod (Figure 3). Adult Cascades frogs were most strongly associated with intermediate and permanent ponds (relative to fast-drying ephemeral ponds) (Figure 4 Table 4). In this case fish had a significant negative influence on the presence of adult *Rana cascadae* (Table 4) despite the small number of sites in which they were present (n=9/168).

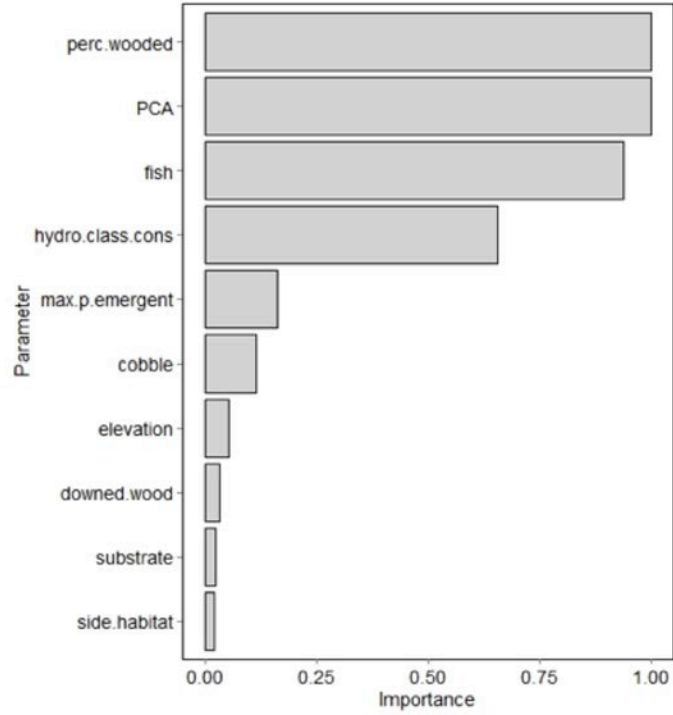


Figure 3. Relative ranking of variable importance for parameters included in the set of models holding 95% of the weight of support based on Akaike weights for *Rana cascadae* adult presence. Importance computed by summing Akaike weights ($\sum w_{Ak}$) for every model in which each variable was present.

Table 3. Top ranked models for the set of models holding 95% of all model support for *Rana cascadae* adult presence (AP). *elev* = elevation, *hydro* = hydroperiod, *PCA* = first PCA axis, *wooded* = percent of pond perimeter occupied by woods, *fish* = fish presence, *emergent* = maximum percent of pond occupied by emergent vegetation, *side habitat* = presence of complex inlet or adjacent wetland habitat, *substrate* = pond bottom substrate, *cobble* = presence of cobbles in the substrate, *downed wood* = presence of branches and downed wood in the pond.

Rank	AIC	Model	$\Delta AICc$	w_{Ak}
1	165.3	AP ~ <i>hydro</i> + <i>PCA</i> + <i>wooded</i> + <i>fish</i>	0	0.59
2	168.2	AP ~ <i>PCA</i> + <i>emergent</i> + <i>wooded</i> + <i>fish</i>	2.83	0.14
3	169.0	AP ~ <i>PCA</i> + <i>wooded</i> + <i>cobble</i> + <i>fish</i>	3.68	0.09
4	170.9	AP ~ <i>elev</i> + <i>PCA</i> + <i>wooded</i> + <i>fish</i>	5.58	0.04
5	171.7	AP ~ <i>PCA</i> + <i>wooded</i> + <i>substrate</i> + <i>fish</i>	6.36	0.02
6	171.8	AP ~ <i>PCA</i> + <i>wooded</i> + <i>fish</i> + <i>downed wood</i>	6.47	0.02
7	172.0	AP ~ <i>hydro</i> + <i>PCA</i> + <i>wooded</i> + <i>cobble</i>	6.67	0.02
8	172.0	AP ~ <i>PCA</i> + <i>wooded</i> + <i>fish</i> + <i>side habitat</i>	6.70	0.02
9	172.4	AP ~ <i>hydro</i> + <i>PCA</i> + <i>emergent</i> + <i>wooded</i>	7.11	0.02
10	172.5	AP ~ <i>elev</i> + <i>hydro</i> + <i>PCA</i> + <i>wooded</i>	7.21	0.02
11	173.8	AP ~ <i>hydro</i> + <i>PCA</i> + <i>wooded</i> + <i>downed wood</i>	8.51	0.01

Figure 4.

Scaled and centered model-averaged parameter coefficient estimates for the variables included in the set of models holding 95% of the weight of support based on Akaike weights.

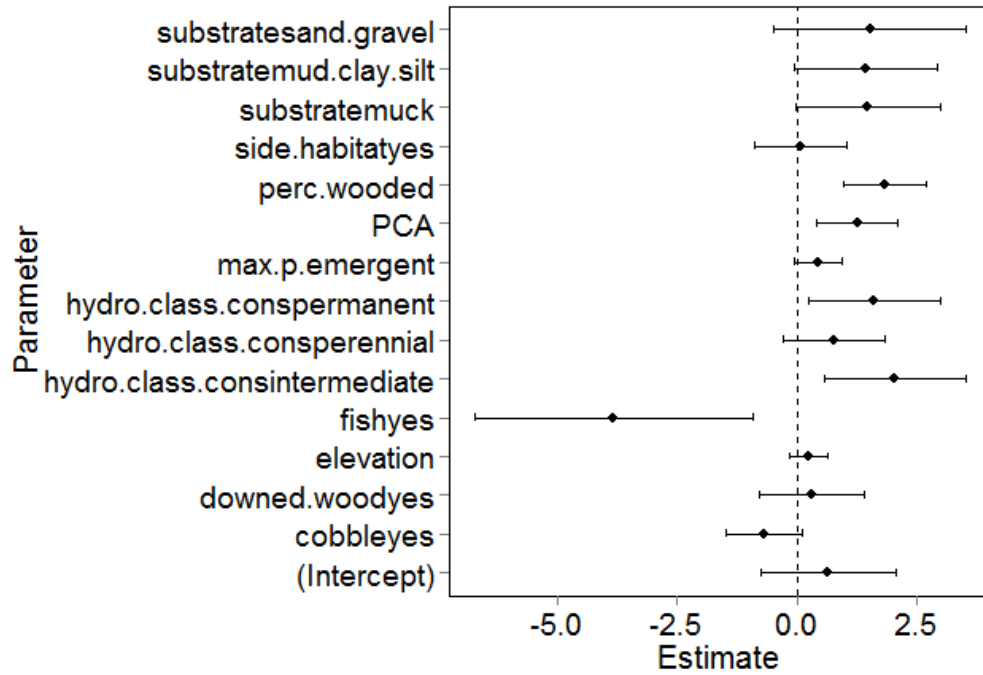


Table 4. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels. Variable codes are defined in Table 1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i>	0.65 (-0.76, 2.06)	0.720
<i>elevation</i>	0.23 (-0.17, 0.64)	0.207
<i>PCA</i> *	1.26 (0.42, 2.10)	0.429
<i>emergent</i>	0.44 (-0.04, 0.92)	0.246
<i>cobble(yes)</i>	-0.69 (-1.48, 0.10)	0.404
<i>wooded</i> *	1.82 (0.96, 2.68)	0.438
<i>fish(yes)</i> *	-3.83 (-6.73, -0.93)	1.480
<i>hydro(intermediate)</i> *	2.04 (0.57, 3.51)	0.750
<i>hydro(perennial)</i>	0.76 (-0.29, 1.82)	0.538
<i>hydro(permanent)</i> *	1.60 (0.22, 2.99)	0.705
<i>side habitat(yes)</i>	0.07 (-0.87, 1.03)	0.487
<i>downed wood(yes)</i>	0.31 (-0.79, 1.41)	0.562
<i>substrate(muck)</i>	1.48 (-0.02, 2.98)	0.767
<i>substrate(mud.clay.silt)</i>	1.43 (-0.07, 2.92)	0.765
<i>substrate(sand.gravel)</i>	1.51 (-0.47, 3.51)	1.017

Ambystoma macrodactylum (Long-toed salamander):
breeding evidence

For long-toed salamander breeding evidence, AIC model selection produced a set of 47 models that held 95% of the weight from the candidate set of 210 models. No top model was supported ($\Delta AIC_{max} = 8.59$; maximum $w_{Ak} = 0.21$; Table 5) and included hydroperiod class, the percent of wooded perimeter, substrate, and the presence of cobble as having the highest variable importance (Figure 5). Of these, all but substrate had significant parameter coefficient estimates (Figure 6 and Table 6), showing stronger evidence of breeding in intermediate and perennial hydroperiod sites with less wooded perimeter and cobble present. While not significant, the PCA coefficient aligns with hydroperiod observations in a trend towards occupying shallower, smaller sites.

Table 5 Top ranked models with $\Delta AIC_c < 4$ for *Ambystoma macrodactylum* breeding evidence (BE). *hydro* = hydroperiod, *wooded* = percent of pond perimeter occupied by woods, *PCA* = first axis of PCA, *emergent* = maximum percent of pond occupied by emergent vegetation, *substrate* = pond bottom substrate, *cobble* = presence of cobbles in the substrate, *side habitat* = presence of complex inlet or adjacent wetland habitat. For full set of top 47 models holding 95% of all model support, see Northwest Climate Science Center report.



Ambystoma macrodactylum eggs (top) and larva approaching metamorphosis (bottom)

Rank	AIC	Model	ΔAIC_c	w_{Ak}
1	181.1	BE ~ <i>hydro</i> + <i>wooded</i> + <i>substrate</i> + <i>cobble</i>	0	0.21
2	181.5	BE ~ <i>hydro</i> + <i>PCA</i> + <i>substrate</i> + <i>cobble</i>	0.44	0.17
3	183.7	BE ~ <i>hydro</i> + <i>wooded</i> + <i>substrate</i> + <i>side habitat</i>	2.62	0.06
4	184.0	BE ~ <i>hydro</i> + <i>emergent</i> + <i>wooded</i> + <i>substrate</i>	2.91	0.05
5	184.1	BE ~ <i>hydro</i> + <i>PCA</i> + <i>wooded</i> + <i>substrate</i>	3.01	0.05
6	184.7	BE ~ <i>hydro</i> + <i>PCA</i> + <i>cobble</i> + <i>side habitat</i>	3.62	0.03
7	184.8	BE ~ <i>hydro</i> + <i>wooded</i> + <i>cobble</i> + <i>side habitat</i>	3.74	0.03
8	184.8	BE ~ <i>hydro</i> + <i>substrate</i> + <i>cobble</i> + <i>side habitat</i>	3.75	0.03

Figure 5. Relative ranking of variable importance for parameters included in the set of models holding 95% of the weight of support based on Akaike weights for *Ambystoma macrodactylum* adult presence. Importance computed by summing Akaike weights (Σw_{AK}) for every model in which each variable was present.

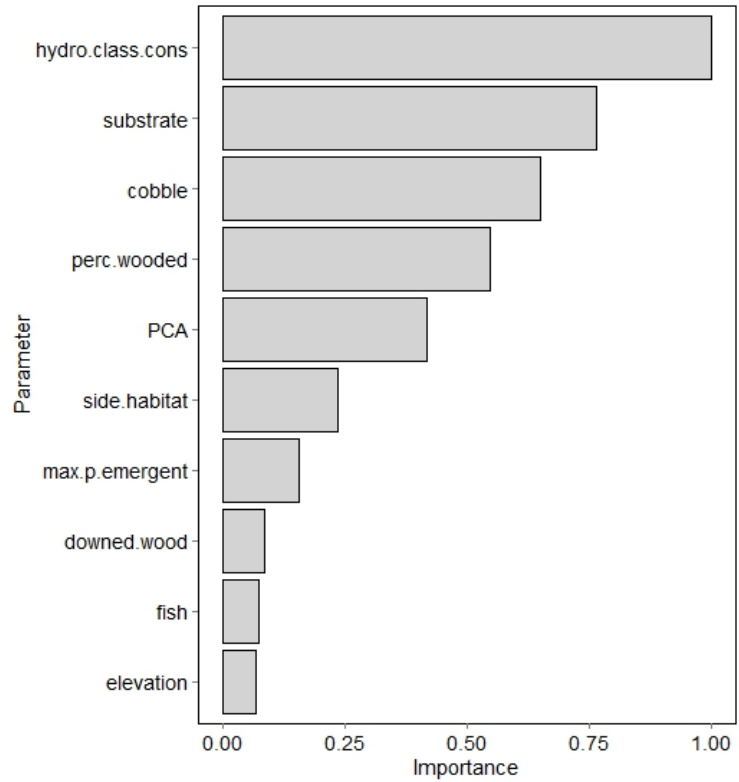


Figure 6. Model-averaged parameter coefficient estimates for variables included in the set of models holding 95% weight of support based on Akaike weights for *A. macrodactylum* breeding.

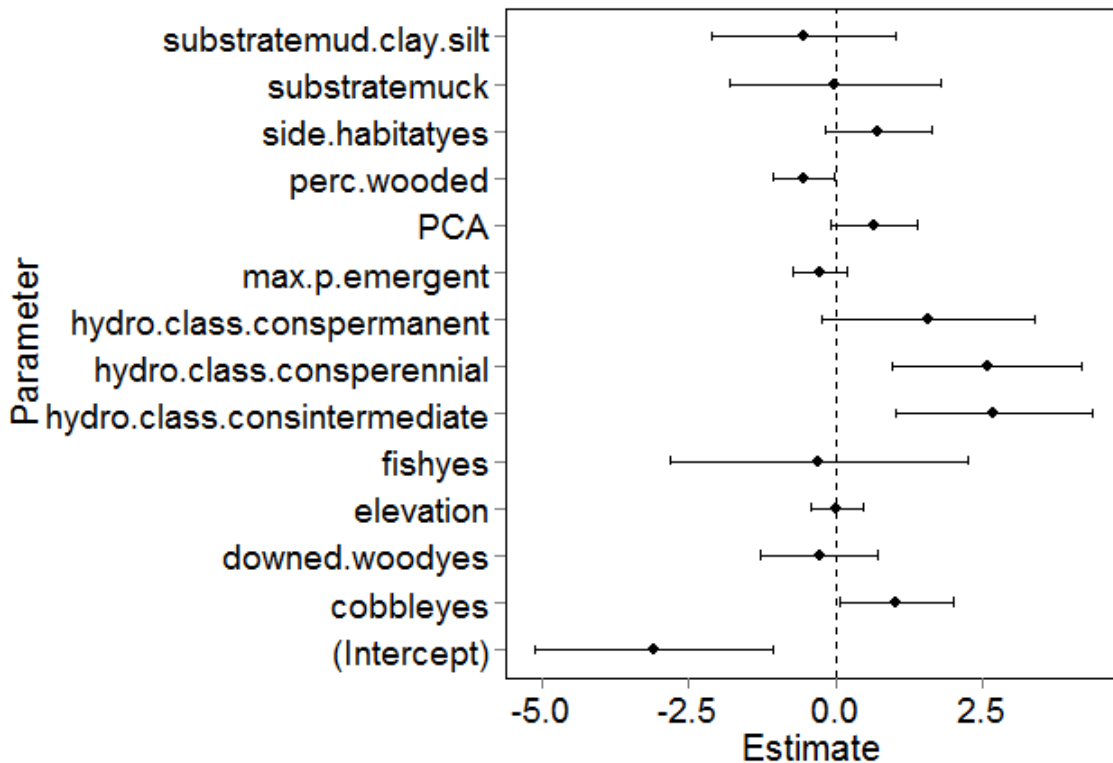


Table 6. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels for *A. macrodactylum* breeding. Variable codes are defined in Table 1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i> *	-3.09 (-3.10, -1.06)	1.035
<i>elevation</i>	0.01 (-0.43, 0.46)	0.229
<i>PCA</i>	0.65 (-0.08, 1.39)	0.374
<i>emergent</i>	-0.27 (-0.73, 0.19)	0.235
<i>cobble(yes)</i> *	1.03 (0.07, 2.00)	0.491
<i>wooded</i> *	-0.53 (-1.06, -0.01)	0.270
<i>fish(yes)</i>	-0.28 (-2.80, 2.24)	1.284
<i>hydro(intermediate)</i> *	2.69 (1.01, 4.36)	0.853
<i>hydro(perennial)</i> *	2.57 (0.96, 4.18)	0.821
<i>hydro(permanent)</i>	1.58 (-0.24, 3.40)	0.927
<i>side habitat(yes)</i>	0.72 (-0.19, 1.63)	0.462
<i>downed wood(yes)</i>	-0.28 (-1.28, 0.72)	0.512
<i>substrate(muck)</i>	-0.01 (-1.81, 1.79)	0.918
<i>substrate(mud.clay.silt)</i>	-0.53 (-2.09, 1.02)	0.795
<i>substrate(sand.gravel)</i>	-17.41 (-2540.03, 2505.20)	1287.070

Ambystoma macrodactylum (Long-toed salamander): adult presence

For long-toed salamander adult presence, AIC model selection produced a set of 42 models that held 95% of the weight from the candidate set of 210 models. The top model was moderately supported (ΔAIC_c for second-ranked model = 3.15; $\Delta AIC_{max} = 10.00$; top model maximum $w_{Ak} = 0.37$; Table 7) and included pond shape (PCA axis), the percent of wooded perimeter, fish presence, and side habitat having the highest variable importance (Figure 7). Of these, all but fish had significant parameter coefficient estimates (Figure 8 and Table 8), showing stronger association of *A. macrodactylum* adults with bigger, deeper ponds with less shallows, less side habitat, and more wooded perimeter. Fish presence was barely not significant, with a strongly negatively trend in response to fish presence.

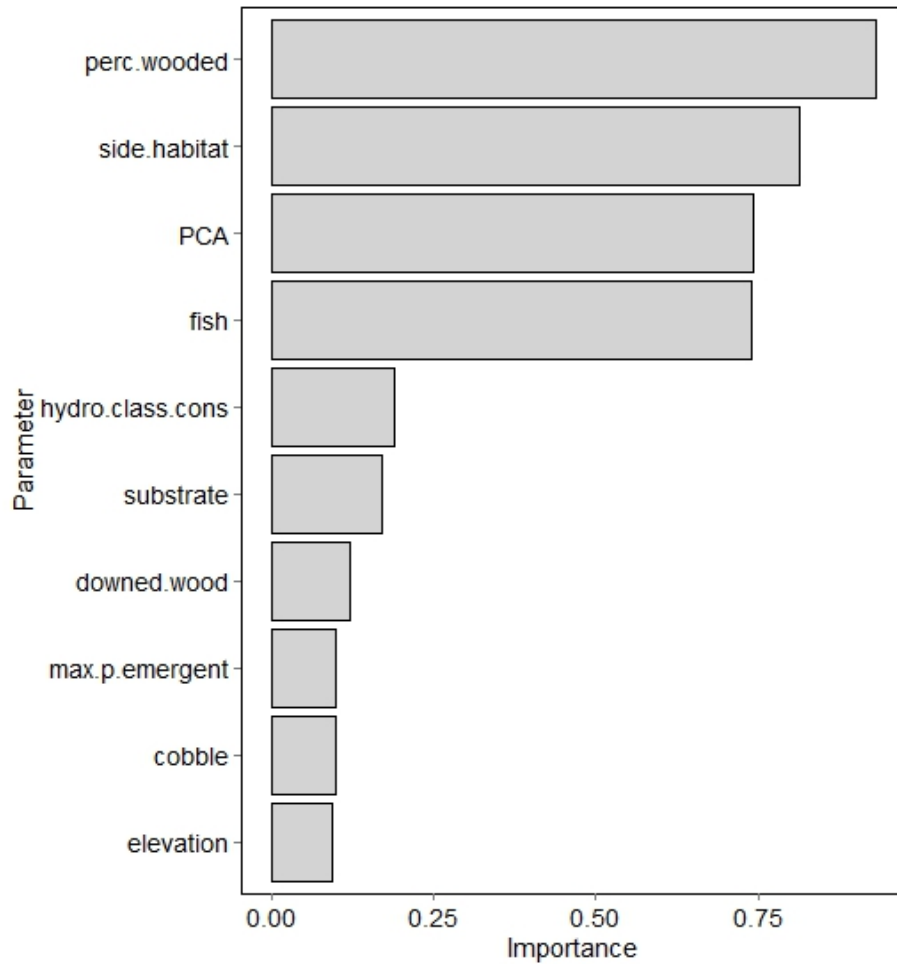


Ambystoma macrodactylum adults hiding in ledges in the pond bank (top) and adult underwater in an alpine pond (bottom)

Table 7. Top ranked models with $\Delta AIC_c < 4$ for presence of *Ambystoma macrodactylum* adults. AP = adult presence, PCA = first axis of PCA, hydro = hydroperiod, wooded = percent of pond perimeter occupied by woods, fish = fish presence, side habitat = presence of complex inlet or adjacent wetland habitat, For full set of top 42 models holding 95% of all model support, see Northwest Climate Science Center report.

Rank	AIC	Model	ΔAIC_c	w_{Ak}
1	146.4	AP ~ PCA + wooded + fish + side habitat	0	0.37
2	149.5	AP ~ hydro + wooded + fish + side habitat	3.15	0.08
3	150.3	AP ~ PCA + wooded + substrate + fish	3.93	0.05

Figure 7. Relative ranking of variable importance for parameters included in the set of models holding 95% of the weight of support based on Akaike weights for *A. macrodactylum* adult presence. Importance computed by summing Akaike weights ($\sum w_{Ak}$) for every model in which each variable was present.



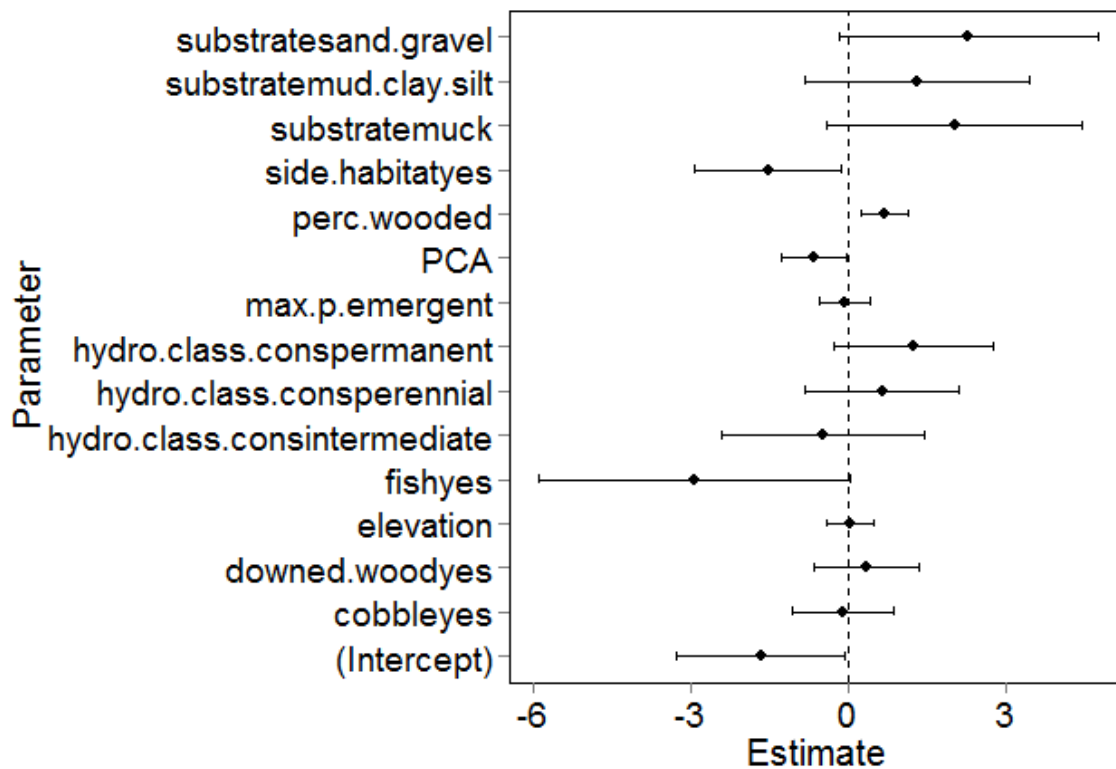


Figure 8. Model-averaged parameter coefficient estimates for variables included in the set of models holding 95% weight of support based on Akaike weights for *A. macrodactylum* adults.

Table 8. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels for *A. macrodactylum* adults. Variable codes are defined in Table 1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i>	-1.65 (-3.26, -0.05)	0.818
<i>elevation</i>	0.05 (-0.39, 0.50)	0.227
<i>PCA</i> *	-0.64 (-1.26, -0.02)	0.318
<i>emergent</i>	-0.05 (-0.53, 0.43)	0.245
<i>cobble(yes)</i>	-0.10 (-1.07, 0.87)	0.496
<i>wooded</i> *	0.68 (0.23, 1.14)	0.232
<i>fish(yes)</i>	-2.93 (-5.90, 0.03)	1.512
<i>hydro(intermediate)</i>	-0.49 (-2.42, 1.44)	0.985
<i>hydro(perennial)</i>	0.64 (-0.81, 2.10)	0.745
<i>hydro(permanent)</i>	1.25 (-0.27, 2.76)	0.774
<i>side habitat(yes)</i> *	-1.52 (-2.91, -0.13)	0.708
<i>downed wood(yes)</i>	0.36 (-0.64, 1.35)	0.506
<i>substrate(muck)</i>	2.02 (-0.40, 4.46)	1.240
<i>substrate(mud.clay.silt)</i>	1.32 (-0.81, 3.44)	1.082
<i>substrate(sand.gravel)</i>	2.29 (-0.17, 4.75)	1.257

Ambystoma gracile (Northwestern salamander): breeding evidence

For northwestern salamander breeding evidence, AIC model selection produced a set of 24 models that held 95% of the weight from the candidate set of 210 models. The top model was fairly strongly supported: ΔAIC_c for second-ranking model = 4.42; $\Delta AIC_{max} = 10.90$; maximum $w_{Ak} = 0.63$; Table 9). The top model included elevation, hydroperiod class, the percent of wooded perimeter, and fish presence (Table 9). Percent wooded, hydroperiod class, fish, and elevation had the highest variable importance (Figure 9). Of these, parameter coefficient estimates were significant for percent wooded, fish, and elevation (Figure 10 and Table 10), showing stronger evidence of breeding in lower-elevation sites with more wooded perimeter, and a negative relationship with fish. Data for hydroperiod class were highly skewed, creating statistical problems (e.g. standard errors >2000). Coefficient estimates are reported in the tables, but the relevant parameters are not included in Figure 10. However the pattern is clear from the raw data: 0/29 ephemeral hydroperiod sites and only 3/25 intermediate sites had signs of *A. gracile* breeding, whereas 11/42 perennial sites and 18/41 permanent sites had breeding activity, indicating a strong association with longer-hydroperiod classes of wetlands.

Table 9. Top ranked models with $\Delta AIC_c < 4$ for *Ambystoma gracile* breeding evidence. BE = breeding evidence, *elev* = elevation, *hydro* = hydroperiod, *wooded* = percent of pond perimeter occupied by woods, *emergent* = maximum percent of pond occupied by emergent vegetation, *fish* = fish presence. For full set of top 24 models holding 95% of all model support, see Northwest Climate Science Center report.

Rank	AIC	Model	ΔAIC_c	w_{Ak}
1	141.1	BE ~ <i>elev</i> + <i>hydro</i> + <i>wooded</i> + <i>fish</i>	0	0.63
2	145.5	BE ~ <i>hydro</i> + <i>emergent</i> + <i>wooded</i> + <i>fish</i>	4.42	0.07



Ambystoma gracile eggmass attached to a stick (top) and larva (bottom)

Figure 9. Relative ranking of variable importance for parameters included in the set of models holding 95% of the weight of support based on Akaike weights for *A. gracile* breeding evidence. Importance computed by summing Akaike weights ($\sum w_{Ak}$) for every model in which each variable was present.

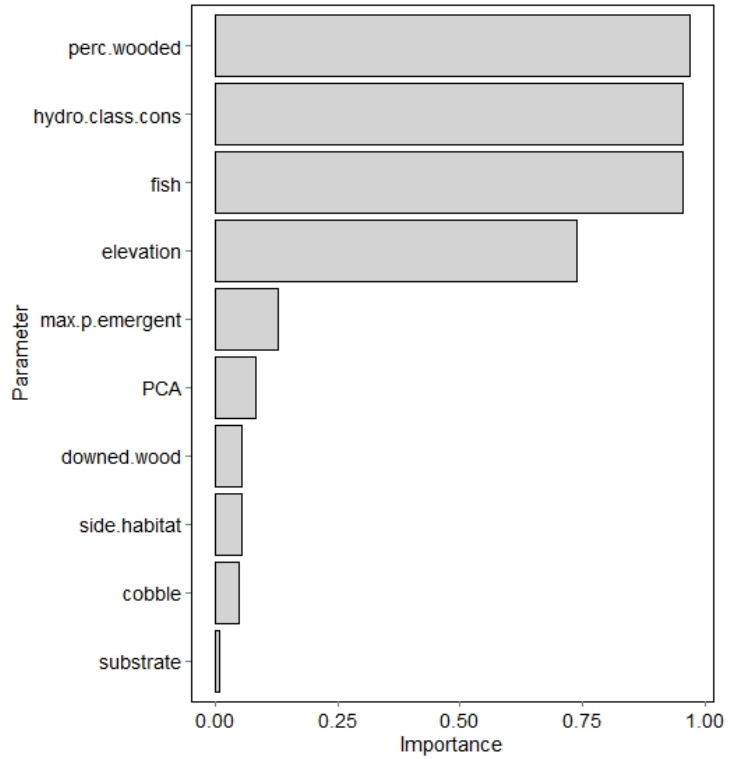


Figure 10. Model-averaged parameter coefficient estimates for the variables included in the set of models holding 95% of the weight of support based on Akaike weights for *A. gracile* breeding evidence. Hydrologic class is missing from the plot due to the scale of SE.

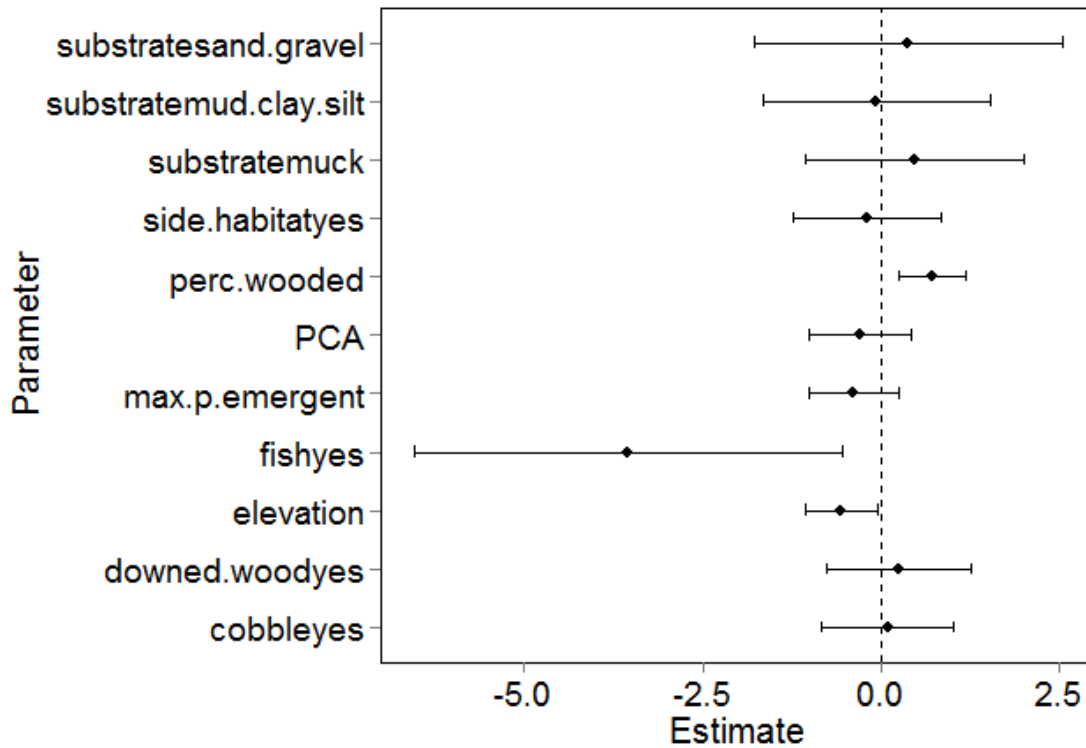


Table 10. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels. Variable codes are defined in Table 1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i>	-17.69 (-2226.82, 2191.44)	1127.130
<i>elevation*</i>	-0.56 (-1.06, -0.06)	0.256
<i>PCA</i>	-0.30 (-1.01, 0.41)	0.362
<i>emergent</i>	-0.39 (-1.02, 0.24)	0.320
<i>cobble(yes)</i>	0.09 (-0.83, 1.01)	0.471
<i>wooded*</i>	0.72 (0.25, 1.19)	0.240
<i>fish(yes)*</i>	-3.55 (-6.54, -0.55)	1.528
<i>hydro(intermediate)</i>	16.57 (-2252.29, 2285.44)	1157.607
<i>hydro(perennial)</i>	17.10 (-2251.77, 2285.97)	1157.607
<i>hydro(permanent)</i>	17.78 (-2251.09, 2286.65)	1157.607
<i>side habitat(yes)</i>	-0.20 (-1.23, 0.85)	0.533
<i>downed wood(yes)</i>	0.25 (-0.75, 1.26)	0.514
<i>substrate(muck)</i>	0.47 (-1.05, 2.00)	0.780
<i>substrate(mud.clay.silt)</i>	-0.06 (-1.66, 1.53)	0.813
<i>substrate(sand.gravel)</i>	0.38 (-1.78, 2.54)	1.100

Ambystoma gracile (Northwestern salamander): adult presence

For northwestern salamander adult presence, AIC model selection produced a set of 18 models that held 95% of the weight from the candidate set of 210 models. The top model was moderately supported: ΔAIC_c for the second-ranked model = 2.79; $\Delta AIC_{max} = 7.17$, maximum $w_{Ak} = 0.39$; Table 11). The top model included the percent of wooded perimeter, hydroperiod class, fish presence, and the presence of side habitat as having the highest variable importance (Figure 11). Of these, all had significant parameter coefficient estimates except hydroperiod class, due to the same statistical issues regarding data skew as for *A. gracile* breeding evidence (Figure 12 and Table 12). Adult *gracile* were associated with ponds with less side habitat and more wooded perimeter, and were negatively associated with fish. As with breeding evidence, adult *A. gracile* predominantly used longer-



Ambystoma gracile paedomorph (aquatic adult, top) and terrestrial adult (bottom)

hydroperiod sites: zero adults were detected in ephemeral ponds, 1/27 intermediate ponds had adults whereas adult *A. gracile* were found in 9/44 perennial and 16/43 permanent ponds. Note that in the case of *A. gracile*, “adults” refer to both terrestrial morphs and paedomorphs, which are the mature aquatic adult form.

Table 11. Top ranked models for the set of models holding 95% of all model support. *BE* = breeding evidence, *elev* = elevation, *hydro* = hydroperiod, *shallow* = maximum percent shallows, *wood* = percent of pond perimeter occupied by woods, *emergent* = maximum percent of pond occupied by emergent vegetation, *size* = maximum size of the pond during the breeding season, *cobble* = presence of cobbles in the substrate, *side* = presence of side habitat.

Rank	AIC	Model	$\Delta AICc$	W_{Ak}
1	113.5	<i>AP ~ hydro + wooded + fish + side habitat</i>	0	0.39
2	116.3	<i>AP ~ PCA + wooded + substrate + fish</i>	2.79	0.10
3	117.1	<i>AP ~ hydro + PCA + wooded + fish</i>	3.56	0.07
4	117.6	<i>AP ~ PCA + wooded + fish + side habitat</i>	4.02	0.05
5	117.6	<i>AP ~ hydro + wooded + substrate + side habitat</i>	4.11	0.05
6	117.8	<i>AP ~ elev + hydro + wooded + side habitat</i>	4.25	0.05
7	117.9	<i>AP ~ hydro + wooded + side habitat + downed wood</i>	4.36	0.04
8	117.9	<i>AP ~ hydro + wooded + cobble + side habitat</i>	4.36	0.04
9	118.0	<i>AP ~ hydro + PCA + wooded + side habitat</i>	4.43	0.04
10	118.0	<i>AP ~ hydro + emergent + wooded + side habitat</i>	4.50	0.04
11	119.3	<i>AP ~ hydro + wooded + substrate + fish</i>	5.81	0.02
12	119.4	<i>AP ~ elev + hydro + wooded + fish</i>	5.83	0.02
13	119.9	<i>AP ~ hydro + wooded + fish + downed wood</i>	6.41	0.02
14	120.0	<i>AP ~ hydro + emergent + wooded + fish</i>	6.42	0.02
15	120.0	<i>AP ~ hydro + wooded + cobble + fish</i>	6.46	0.02
16	120.1	<i>AP ~ PCA + wooded + fish + downed wood</i>	6.52	0.01
17	120.5	<i>AP ~ PCA + wooded + cobble + fish</i>	6.92	0.01
18	120.7	<i>AP ~ elev + PCA + wooded + fish</i>	7.17	0.01

Figure 11. Relative ranking of variable importance for parameters included in the set of models holding 95% weight of support based on Akaike weights for *A. gracile* adult presence (terrestrial morphs and pedomorphs). Importance computed by summing Akaike weights ($\sum w_{Ak}$) for every model in which each variable was present.

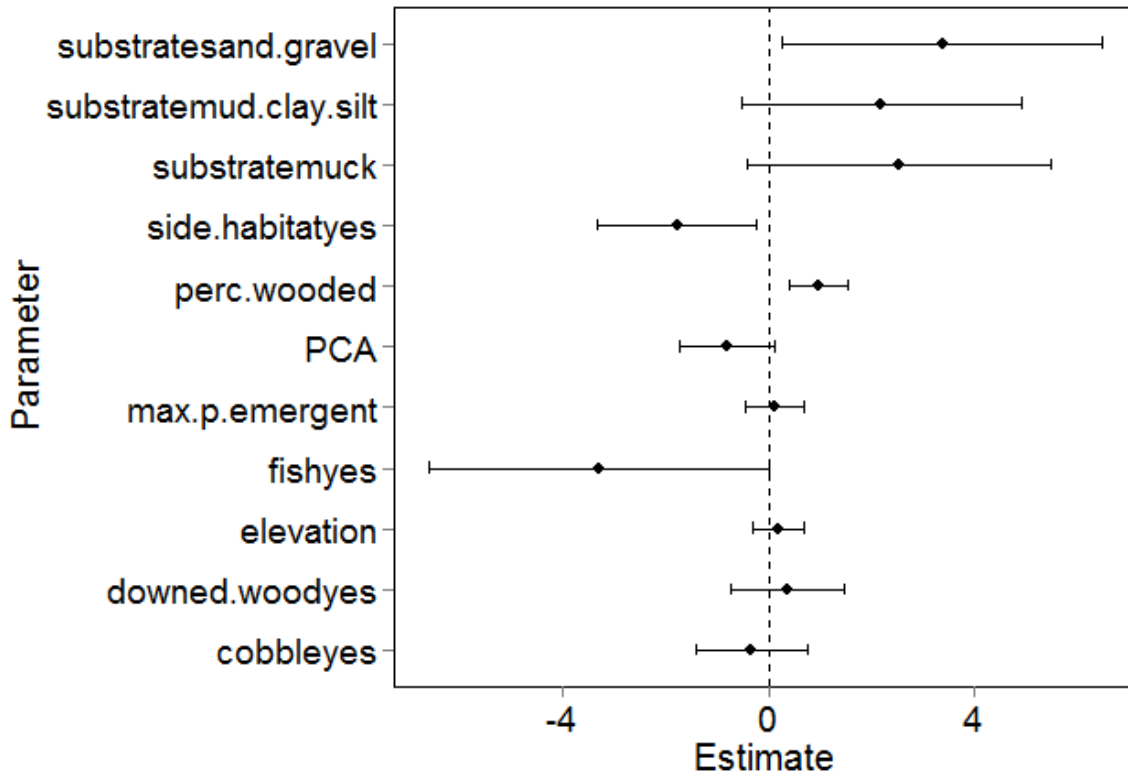
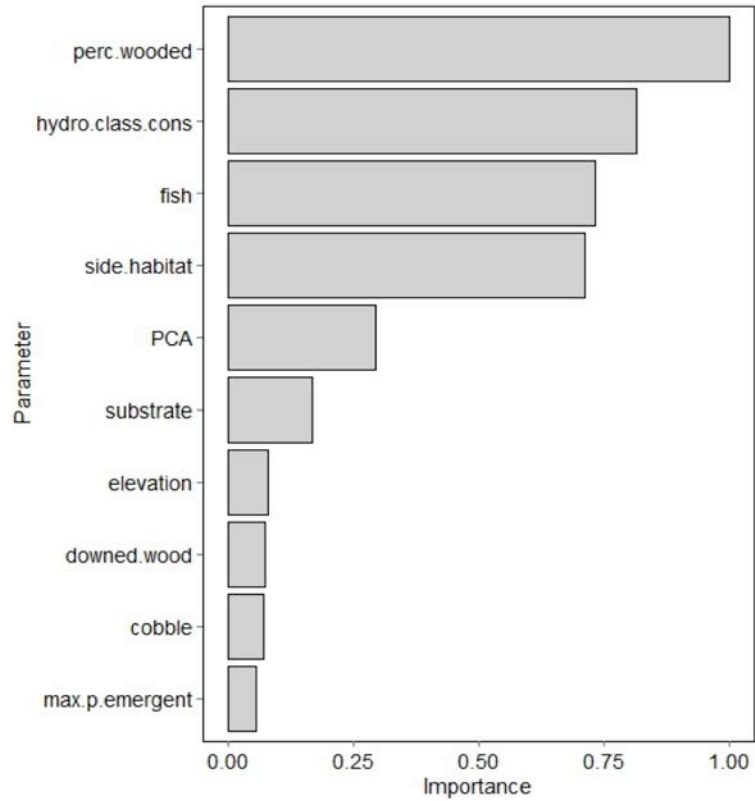


Figure 12. Model-averaged parameter coefficient estimates for variables included in the set of models holding 95% of the weight of support based on Akaike weights for *A. gracile* adults.

Table 12. Model averaged coefficients and 95% confidence intervals for factors predicting injury at the individual and habitat unit levels for *A. gracile* adult presence. Variable codes are defined in Table 1. Stars indicate variables with confidence intervals that do not bound 0.

Variable	Model-Averaged Estimate (95% CI)	Unconditional SE
<i>Intercept</i>	-15.96 (-2715.92, 2683.99)	1377.55
<i>elevation</i>	0.18 (-0.32, 0.70)	0.26
<i>PCA</i>	-0.81 (-1.74, 0.13)	0.48
<i>emergent</i>	0.13 (-0.43, 0.71)	0.29
<i>cobble(yes)</i>	-0.32 (-1.40, 0.75)	0.55
<i>wooded*</i>	0.99 (0.42, 1.55)	0.29
<i>fish(yes)*</i>	-3.31 (-6.64, 0.02)	1.70
<i>hydro(intermediate)</i>	15.89 (-3007.37, 3039.15)	1542.51
<i>hydro(perennial)</i>	17.61 (-3005.65, 3040.87)	1542.51
<i>hydro(permanent)</i>	18.24 (-3005.02, 3041.49)	1542.51
<i>side habitat(yes)*</i>	-1.78 (-3.33, -0.24)	0.79
<i>downed wood(yes)</i>	0.37 (-0.74, 1.48)	0.57
<i>substrate(muck)</i>	2.54 (-0.42, 5.51)	1.51
<i>substrate(mud.clay.silt)</i>	2.21 (-0.51, 4.93)	1.39
<i>substrate(sand.gravel)*</i>	3.39 (0.28, 6.51)	1.59

Synthesis

Assessment of amphibian vulnerability to future climate change

Sensitivity: Both aquatic life stages (eggs and tadpoles) and adult *Rana cascadae* (Cascades frog) were associated with intermediate hydroperiod ponds, which are also the most sensitive class of wetlands to climate change. *Ambystoma macrodactylum* breeding is associated with intermediate and perennial hydroperiod wetlands, both of which are anticipated to experience reduced water levels and in some cases conversion to shorter-hydroperiod ponds. For this species, which requires multiple years at higher elevations for successful metamorphosis, an increase in the frequency of pond drying, as is projected for these kinds of sites, will have a negative effect on recruitment. Adults in contrast are more strongly associated with deeper, less shallow ponds that are less sensitive to climate impacts. *Ambystoma gracile* appear to be the least sensitive of the three species to climate change, since their breeding and adult habitat use are skewed towards longer hydroperiod permanent ponds. While they also use more climate-vulnerable perennial ponds, they do not rely strongly on the most sensitive wetland types, the intermediate and ephemeral ponds, as either breeding or adult habitat.

Exposure: Exposure varies across the three parks with significant variation in the increase in the likelihood of drying (e.g. for intermediate ponds) across different regions of the parks. Parks also differ in the underlying distribution of pond types across the landscape (Northwest Climate Science Center report; and see below). In assessing the distribution of sites, it is important to note that our perennial wetland class does not align well with any of the NWI categories, so perennial ponds are likely under-represented in this assessment. Also, as the results of a mapping study in Mount Rainier National Park show (Northwest Climate Science Center report), small wetlands of any hydrologic type are substantially underrepresented by the NWI in the montane regions we have studied. Therefore it is fair to assume that

we are underestimating the total amount of small wetland habitat, which is of course our primary interest for two species, and also the most at-risk wetland type. We discuss approaches to dealing with this deficiency in our Northwest Climate Science Center report. In the meantime, acknowledging the limitations, Figures 13-15 and Tables 13-15 show the distribution of NWI mapped wetlands overlaid with the VIC model projections for the change in probability of drying for intermediate ponds. Since we cannot parameterize analogous projections for perennial ponds (due to no history of drying in the historical record), we use this projection to also indicate relative risk to perennial ponds assuming that the mechanisms affecting both are the same. Our findings from our focal sites project that 22% of current perennial ponds will become intermediate wetlands by the 2080s, and 3% will become ephemeral. (In contrast, 58% of intermediate wetlands are projected to become ephemeral.) These estimates can be applied to the list below to estimate the shift from perennial to other pond types within each VIC cell (colored grid cell).

In Mount Rainier National Park, our projections suggest that the areas of highest climate impacts that overlap with large numbers of intermediate and perennial ponds are in the central southern part of the Park, with additional areas of significantly elevated climate risk in the east and west (Figure 13). In North Cascades National Park, the regions of greatest change in the probability of pond drying due to climate change occur in a somewhat patchy pattern across the western half of the park, and with stronger

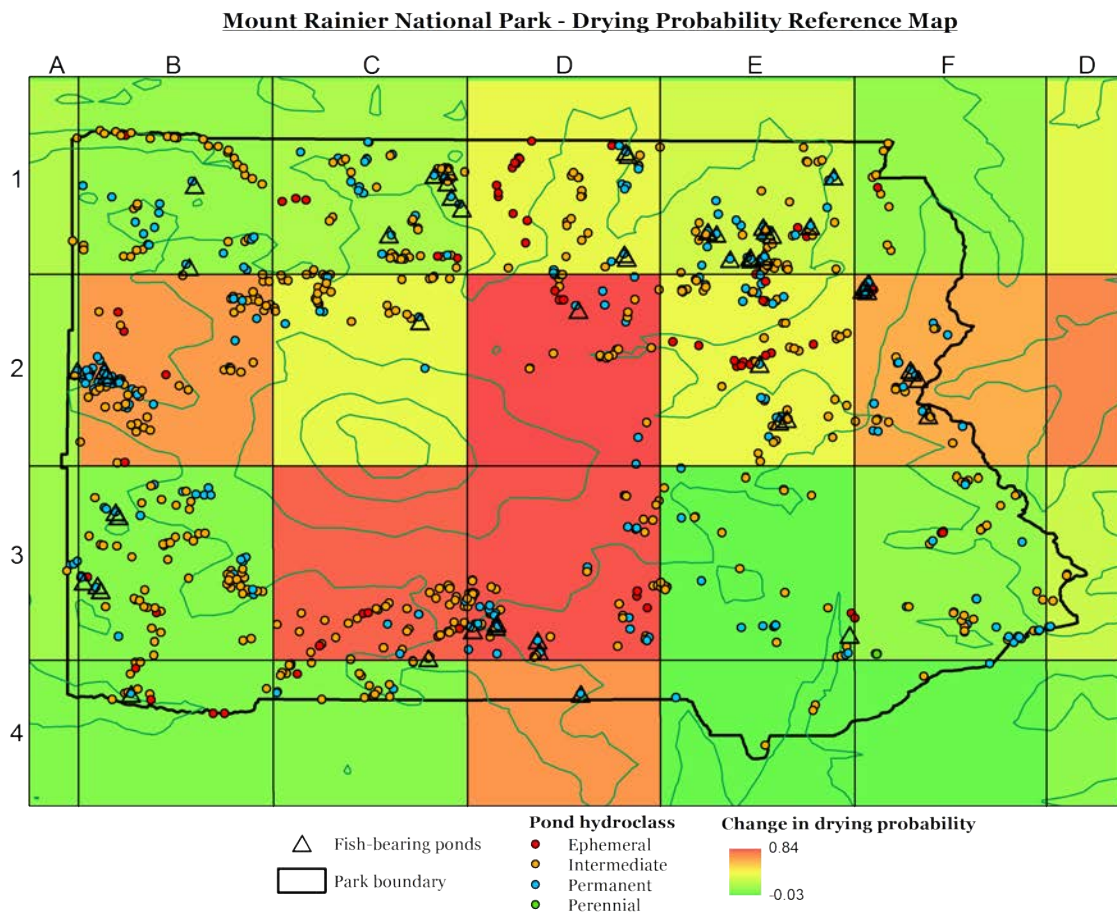


Figure 13. Mount Rainier National Park wetland and amphibian vulnerability reference map. Colored squares represent VIC grid cells and their projected proportion of change in drying probability for intermediate wetlands. The axis letters and numbers are referenced in Table 13.

Table 13. Drying probability reference map for Mount Rainier National Park

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
A1	0.98	0.19	0	2	0	0	2	0
A2	0.99	0.15	0	2	0	2	4	1
A3	0.99	0.12	0	1	0	3	4	0
A4	1	0.05	0	0	0	0	0	0
B1	0.99	0.11	2	32	0	15	49	2
B2	0.99	0.68	4	52	0	35	91	3
B3	1	0.06	4	55	0	20	79	5
B4	1	0.05	4	9	0	1	14	1
C1	0.99	0.17	5	32	0	24	61	7
C2	0.35	0.85	0	30	0	6	36	1
C3	0.82	0.32	5	37	0	5	47	0
C4	1	0.06	2	17	0	5	24	1
D1	1	0.36	13	19	0	10	42	4
D2	0.98	0.85	4	19	0	8	31	1
D3	1	0.83	3	43	0	28	74	5
D4	1	0.7	0	0	0	1	1	1
E1	1	0.28	2	32	0	23	57	14
E2	1	0.37	15	35	0	20	70	3
E3	1	0	1	20	0	11	32	1
E4	1	0	0	3	0	1	4	0
F1	1	0.1	1	8	0	0	9	0
F2	1	0.63	1	14	0	19	34	8
F3	1	0.1	3	28	1	16	48	0
F4	1	0	0	1	0	1	2	0
G3	1	0.25	0	5	0	1	6	0
TOTALS			69	496	1*	255	821	58

* Note that perennial ponds do not map well to NWI wetland classes, so are underrepresented in our assessment.

climate impacts generally projected for the west side of the Cascades crest, with exceptions (Figure 14). North Cascades has relatively few mapped wetlands, many of which are permanent so less subject to the kinds of climate impacts we are considering. Many of the clusters of intermediate ponds in the southern part of the Park appear to be at lower risk of climate impacts, while those in the far northern and western regions of the Park are more at risk. Olympic National Park by contrast has many mapped wetlands, a large proportion of which are intermediate wetlands. The areas of greatest change in wetland drying probability are projected to be the central and western regions of Olympic National Park (Figure 15), which include some areas (such as along river valleys) with large numbers of intermediate wetlands. Some of these regions are at elevations too low to be occupied by Cascades frogs, for example, but are likely to be used by *Ambystoma macrodactylum*, which can be found at elevations extending down to sea level (though the developmental constraints are less severe as they can metamorphose faster in warmer,

North Cascades National Park - Drying Probability Reference Map

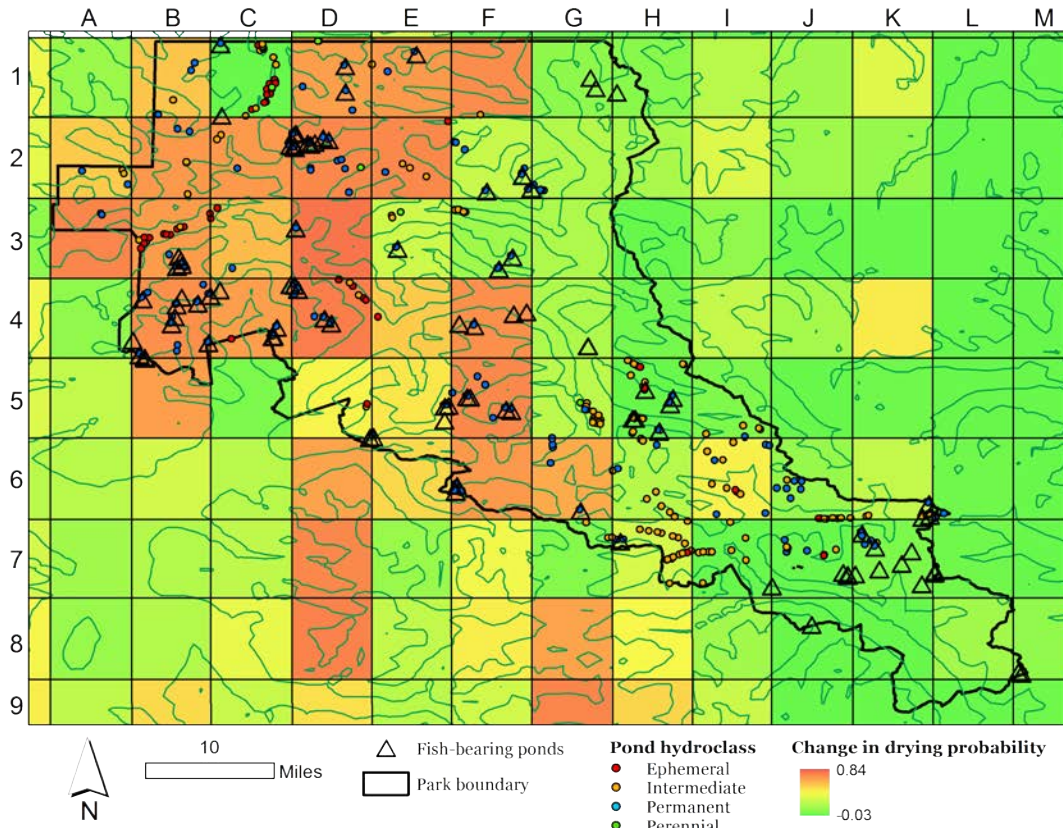


Figure 14. North Cascades National Park wetland and amphibian vulnerability reference map. Colored squares represent VIC grid cells and their projected proportion of change in drying probability for intermediate wetlands. The axis letters and numbers are referenced in Table 3.5.1.

Table 14. Drying probability reference map for North Cascades National Park

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
A2	0.55	0.54	0	2	0	2	4	0
A3	0.82	0.74	0	0	0	2	2	0
A4	1	0.09	0	0	0	0	0	0
B1	0.97	0.55	0	1	0	5	6	0
B2	0.83	0.68	0	2	0	3	5	0
B3	0.85	0.68	11	4	0	4	19	4
B4	0.98	0.71	0	0	0	13	13	11
B5	0.94	0.67	0	0	0	0	0	1
C1	1	0	10	10	2	1	23	2
C2	0.84	0.69	0	2	0	2	4	1
C3	0.93	0.59	1	0	0	1	2	0
C4	0.99	0.67	1	0	0	3	4	5
C5	0.99	0.04	0	0	0	0	0	0

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
D2	0.94	0.73	0	0	1	15	16	10
D3	0.78	0.78	0	0	0	1	1	1
D4	0.98	0.76	4	2	0	6	12	4
D5	1	0.41	1	1	0	0	2	0
D6	0.95	0.65	0	0	0	0	0	1
E1	0.68	0.68	0	1	0	2	3	1
E2	0.73	0.73	1	3	0	1	5	0
E3	0.98	0.27	0	2	1	1	4	1
E4	0.99	0.54	1	0	0	0	1	0
E5	0.98	0.46	0	0	0	3	3	4
E6	0.93	0.49	0	0	0	0	0	0
F1	0.89	0.72	0	1	0	0	1	0
F2	0.99	0.25	0	2	0	7	9	3
F3	0.99	0.12	0	5	1	3	9	2
F4	0.93	0.72	0	0	0	1	1	4
F5	0.87	0.73	0	0	0	8	8	4
F6	0.95	0.69	0	0	0	5	5	2
G1	1	0.08	0	0	0	0	0	2
G2	1	0.19	0	0	0	6	6	0
G3	1	0.17	0	0	0	0	0	0
G4	1	0.17	0	0	0	0	0	2
G5	1	0.22	0	12	1	4	17	0
G6	1	0.63	0	0	0	4	4	1
G7	1	0.05	0	2	0	0	2	0
H1	1	0.07	0	0	0	0	0	1
H2	1	0.15	0	0	0	0	0	0
H3	1	0	0	0	0	0	0	0
H4	1	0.02	0	0	0	0	0	0
H5	1	0	4	17	1	3	25	6
H6	1	0.19	0	8	0	3	11	0
H7	1	0.3	2	17	0	2	21	1
I5	1	0.04	0	4	0	1	5	0
I6	1	0.43	1	7	0	5	13	0
I7	1	0	0	9	0	0	9	1
I8	1	0.13	0	0	0	0	0	0
J5	1	0	0	0	0	0	0	0
J6	1	0.1	1	4	0	6	11	0
J7	1	0	3	6	0	4	13	3
J8	1	0	0	0	0	0	0	1
K6	1	0.23	0	6	0	2	8	3
K7	1	0.03	0	2	0	5	7	7
K8	1	0	0	0	0	0	0	0
K9	1	0	0	0	0	0	0	0
L6	1	0.01	0	1	0	2	3	0
L7	1	0.01	0	0	0	0	0	2
L8	1	0.13	0	0	0	0	0	0
L9	1	0	0	0	0	0	0	0
M8	1	0.09	0	0	0	0	0	2
TOTALS			41	134	8*	140	323	95

* Note that perennial ponds do not map well to NWI wetland classes, so are underrepresented in our assessment.

lower elevation regions). Of greatest concern for our purposes in this study are the mid to high-elevation regions, primarily within the central part of the Park, where large numbers of intermediate wetlands overlap with elevated drying risk. Table 3.5.3 shows the distribution of these sites in tabulated form.

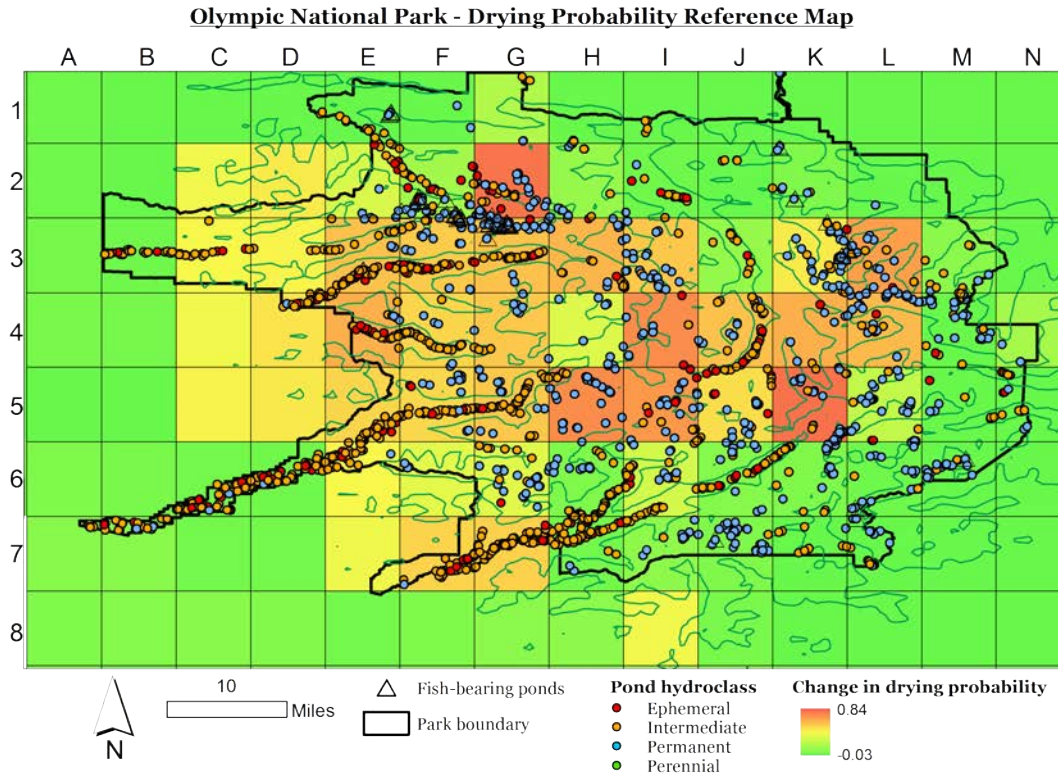


Figure 15. Olympic National Park wetland and amphibian vulnerability reference map. Colored squares represent VIC grid cells and their projected proportion of change in drying probability for intermediate wetlands. The axis letters and numbers are referenced in Table 3.5.1.

Table 15. Drying probability reference map for Olympic National Park

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
A7	1.00	0.06	1	4	0	0	5	0
B2	1.00	0.00	0	0	0	0	0	0
B3	1.00	0.01	4	20	0	0	24	0
B6	1.00	0.03	0	0	0	0	0	0
B7	1.00	0.06	4	15	0	4	23	0
C2	1.00	0.39	0	0	0	0	0	0
C6	1.00	0.02	10	37	1	3	51	0
D1	1.00	0.00	0	2	0	0	2	0
D2	0.93	0.44	0	0	0	0	0	0
D3	0.89	0.42	0	5	0	0	5	0

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
D4	0.86	0.46	4	21	0	1	26	0
D5	0.86	0.45	0	0	0	0	0	0
D6	1.00	0.01	13	27	0	1	41	0
E1	1.00	0.00	0	10	0	2	12	6
E2	1.00	0.35	5	7	0	5	17	0
E3	0.94	0.57	7	44	0	6	57	3
E4	0.87	0.61	8	25	0	3	36	0
E5	0.90	0.51	8	30	0	0	38	0
E6	0.90	0.40	0	0	0	3	3	0
E7	0.92	0.38	6	25	0	7	38	0
E8	1.00	0.02	0	0	0	0	0	0
F1	1.00	0.00	0	0	0	2	2	0
F2	0.99	0.13	9	24	0	17	50	26
F3	0.90	0.53	9	24	0	21	54	2
F4	0.84	0.51	1	29	0	6	36	0
F5	0.88	0.50	4	33	0	13	50	0
F6	0.93	0.34	0	0	0	3	3	0
F7	0.84	0.51	6	25	0	7	38	0
G1	1.00	0.18	0	2	0	2	4	0
G2	0.95	0.78	8	12	0	21	41	1
G3	0.94	0.56	10	14	0	39	63	19
G4	0.83	0.55	0	5	0	7	12	0
G5	0.90	0.54	2	34	0	13	49	0
G6	1.00	0.33	1	8	0	18	27	0
G7	1.00	0.52	9	48	0	0	57	0
H1	1.00	0.00	0	2	0	0	2	0
H2	1.00	0.10	0	3	0	11	14	0
H3	0.98	0.59	2	9	0	9	20	0
H4	0.32	0.32	8	25	0	3	36	0
H5	0.83	0.72	0	8	0	23	31	0
H6	1.00	0.04	3	27	0	20	50	1
H7	1.00	0.02	5	47	0	4	56	0
I1	1.00	0.00	0	4	0	0	4	0
I2	1.00	0.01	5	11	0	2	18	0
I3	1.00	0.55	1	6	0	24	31	0
I4	0.92	0.72	2	1	0	9	12	0
I5	0.88	0.68	5	7	0	16	28	0
I6	1.00	0.23	4	24	0	11	39	0
I7	1.00	0.08	0	5	0	10	15	0
J1	1.00	0.00	0	0	0	0	0	0
J2	1.00	0.00	0	4	0	0	4	0
J3	1.00	0.10	1	9	0	0	10	0
J4	0.95	0.57	8	27	0	9	44	0
J5	0.96	0.48	0	6	0	24	30	0

Cell	2080 Drying Probability	Δ Drying Probability	Ephemeral	Intermediate	Perennial	Permanent	Total	Fish Ponds
J6	1.00	0.05	7	14	0	11	32	0
J7	1.00	0.04	0	8	0	1	9	1
K1	1.00	0.00	0	0	0	1	1	0
K2	1.00	0.04	0	3	0	6	9	3
K3	1.00	0.44	0	9	0	28	37	4
K4	0.98	0.61	1	1	0	5	7	0
K5	0.99	0.78	0	6	0	24	30	0
K6	1.00	0.04	1	12	0	18	31	0
K7	1.00	0.00	0	8	0	1	9	0
L2	1.00	0.00	0	0	0	1	1	0
L3	1.00	0.67	2	6	0	13	21	4
L4	1.00	0.59	1	5	0	28	34	0
L5	1.00	0.78	2	5	0	28	35	0
L6	1.00	0.04	0	2	0	19	21	1
L7	1.00	0.00	1	6	1	11	19	1
M2	1.00	0.00	0	0	0	0	0	0
M3	1.00	0.06	0	7	0	4	11	0
M4	1.00	0.00	3	9	0	22	34	1
M5	1.00	0.00	1	4	0	4	9	0
M6	1.00	0.00	0	3	0	15	18	1
N3	1.00	0.00	0	0	0	0	0	0
N4	1.00	0.14	0	0	0	2	2	0
N5	1.00	0.00	3	9	0	5	17	0
N6	1.00	0.00	0	0	0	0	0	0
TOTALS			198	885	2*	626	1711	74

* Note that perennial ponds do not map well to NWI wetland classes, so are underrepresented in our assessment.

Adaptive capacity: We highlight one primary option for management-based climate adaptation, which is the removal of introduced fish that eat amphibians and tend to limit their distributions to shallower and more at-risk ponds and wetlands (Ryan et al. 2014). Fish removals are within the mandate of the National Parks and are already underway in North Cascades National Park and under consideration elsewhere. As a first step, we report on the location and number of ponds with fish, based on National Park Service records, in relation to the distribution of wetland types and increased drying risk (triangles denoted in Figures 13-15 and listed in Tables 13-15). From a biological standpoint, the degree to which species like *Rana cascadae* and *Ambystoma macrodactylum* may be able to hasten development sufficiently under warmer conditions to compensate for reductions in hydroperiod is an open question, and was not the subject of our research.

All data are available to the public through the Northwest Climate Science Center's data repository and include VIC outputs (for wetland water level and water temperature) and projections; updated wetland maps for Mount Rainier National Park based on LiDAR analysis; NWI-based wetland maps cross-walked to our hydrologic categories for Mount Rainier, North Cascades, and Olympic National Parks; amphibian survey data and R code for the preliminary ecological analyses presented here. We hope that our results

support NPS and USFS's collaborative leadership in climate adaptation planning by providing products that support the aims of the North Cascadia Adaptation Partnership as well as the collaborative climate adaptation efforts of Olympic National Park and Forest.

Discussion

From Northwest Climate Science Center report: Summary of climate impacts on montane wetlands

Our climate-change projections demonstrate that all four of the wetland types on which we focused (ephemeral, intermediate, perennial, and permanent wetlands) are likely to experience hydrologic changes in response to future climate. However, the intensity and duration of climate change effects will differ markedly among the four types. These changes are also likely to lead to transitions along the continuum of wetland types captured in our hydrologic classes. Specifically, some ephemeral wetlands may essentially disappear and more than half of currently ecologically productive intermediate montane wetlands are projected to become ephemeral wetlands by the 2080s, as more rapid recession rate and earlier drawdown causes wetlands to reach their bottom volume earlier, resulting in more frequent and longer dry seasons in summer. For some perennial wetlands in Washington, transitions from perennial to intermediate wetlands or even to ephemeral wetlands are also projected as wetland water levels drop under climate change. Driving these changes is the fact that most montane wetlands are located either in snow-dominated watersheds or mixed-rain-and-snow watersheds where snowmelt is a key water source in late spring and summer. Because a warmer climate is likely to cause less snow accumulation in winter and earlier snowmelt in spring, montane wetlands are particularly susceptible to climate change, especially in combination with projected drier summers (Hamlet *et al.*, 2013; Elsner *et al.*, 2010).

Our models were able to generally reproduce historical patterns of wetland temperature dynamics, so suggest that this approach could be expanded to evaluate water temperature impacts of climate change in more detail. Based on our 18 wetlands, our projections show a general increase (average $\sim 2^{\circ}\text{C}$) in the maximum water temperature for all sites in response to increased air temperatures associated with climate change. For permanent wetlands that are generally deeper than other wetland types, observed water temperature showed much less fluctuation on a daily time step (i.e. less sensitive to air temperature) than that of other wetland types. As a result, permanent wetlands showed lower goodness of fit values than other wetland types. Our small sample set of permanent wetlands suggests that elevation may influence sensitivity of water temperatures in permanent ponds to climate change, as would be anticipated based on other research on elevational gradients in hydrologic impacts of climate change. The extent to which the average $\sim 2^{\circ}\text{C}$ increase in water temperature will affect the biota and function of sites is likely to be species-specific. These kinds of projections could support research on these impacts by providing a range of plausible temperature effects based on climate projections.

Amphibian Assessment

In our study we focused on three pond-breeding amphibians: *Rana cascadae*, *Ambystoma macrodactylum*, and *Ambystoma gracile*. In the preliminary analyses presented here, across all species either hydroperiod or in one case pond shape (i.e. first PCA axis that differentiated deep, large from shallow, smaller wetlands) was among the highest variable importance in each of the six analyses (each species, analyzed by breeding evidence and adult presence). Either hydroperiod or the pond shape PCA axis was a significant parameter coefficient (with the exception of *A. gracile*, where positive observations were so skewed towards deeper pond types that the statistics failed).

The specific relationships of each species to hydroperiod differed, however. *Rana cascadae* breeding was strongly associated with intermediate and perennial wetlands. This likely reflects the beneficial growth conditions found there (warmer ponds allow faster developmental rates), as long as they do not dry. These sites are also unlikely to be occupied by fish, as discussed below. Adult habitat use for *Rana cascadae* was most strongly associated with intermediate ponds, but was also significantly greater in permanent

ponds (compared to ephemeral sites). There was a positive but not significant association with perennial ponds for adults. This split distribution is likely to be capturing multiple uses for these different kinds of sites. For example, early season use of adult *R. cascadae* for breeding, and later summer and fall use of permanent ponds for foraging. *Ambystoma macrodactylum* likewise predominantly used intermediate and perennial ponds for breeding. However adult *A. macrodactylum* were more strongly associated with deeper, less shallow ponds. *Ambystoma gracile* showed a different pattern, with both breeding and adult presence strongly associated with permanent, and to a lesser degree perennial, pond types.

Despite having observations of fish in only 9 out of the 168 sites surveyed, fish also appeared in all of the top models, often with a statistically significant negative effect, and with a negative effect in all cases. These findings – the importance of hydrologically-related features and of fish – support our expectations that these two factors would be of importance for amphibians. Other important factors varied by species and life stage. The common inclusion of the percentage of wooded perimeter in many of the top models may reflect on-the-ground conditions and microclimates in the mountains more accurately than a coarser measure like elevation (which also appeared in several top models). Therefore it was not a surprise that species such as *Rana cascadae* and *Ambystoma gracile* were associated with more heavily forested ponds, while *Ambystoma macrodactylum*, known to be the highest elevation species of the three that may be found above treeline, had a negative relationship with forest cover.

One of the questions that may be answered by future occupancy analysis of this dataset is the degree to which our findings may be improved with estimates of detection. Particularly for species such as *A. macrodactylum* that have larvae that are known to be more active at night, and adults that are relatively small and cryptic compared to the other two species, this assessment will be important. We might expect some associations to strengthen based on incorporation of detection rates. For example, *A. macrodactylum* are associated with higher elevation ponds with cobble in the substrate. The same feature (cobble) also acts as refugia in which larvae can hide to avoid detection. Next steps are to further explore these data with specific hypotheses that incorporate interactions among variables, and to investigate the effect of detection probabilities on estimates of occupancy associated with different habitat types.

Sensitivity: The differences in breeding and adult foraging habitat use and life history requirements among our three focal species translate into different levels of climate-related risk of habitat loss that most clearly could affect breeding and recruitment. *Rana cascadae* is of greatest concern, as a montane obligate species not found at lower elevations, which heavily relies for recruitment on intermediate and perennial ponds that are at highest risk of climate impacts. *Ambystoma macrodactylum* also appear to be at substantial risk of losing breeding habitat due to increased pond drying rates in montane and alpine regions. While in a general sense *A. macrodactylum* is buffered somewhat by its broader range (found from sea level up to alpine regions), the species is doubly at risk of negative climate impacts in montane regions due to 1) its reliance on intermediate and perennial ponds for breeding and 2) its requirement of multiple consecutive years of water for larvae to complete metamorphosis at higher elevations. Therefore while Cascades frogs are likely to be most affected by reduced times to pond drying, risk to long-toed salamanders is amplified by the projected increase in the inter-annual frequency of pond drying. *Ambystoma gracile* appear to be at lowest risk of direct negative impacts on breeding habitat, due to their reliance on longer hydroperiod kinds of ponds that according to our analysis are less sensitive to climate change. However, while *Ambystoma gracile* appear likely to experience less direct habitat loss, shifts in pond conditions or the frequency of drying in perennial sites may have other life history impacts, such as shifting the relative frequency of metamorphosis versus paedomorphosis. This possibility generates additional interesting questions about how survival rates vary in alpine regions among the two adult forms, whether those survival rates may be affected by climate change, and whether shifts in the frequency of adult morphs could have implications for demographic rates and population viability under climate change. Likewise, while our analysis does not address impacts of climate change beyond effects

on breeding habitat, we would anticipate a broader range of demographic effects to either exacerbate or help compensate for recruitment losses.

Exposure: Overall, we found considerable and in some cases severe potential impacts of climate change on montane wetlands in all three National Parks, with the magnitude of impacts varying in space across each landscape with factors such as mountain topography and other key drivers of regional climate variation. Our assessment provides a first step in evaluating the exposure of wetlands and montane species to climate change, with more advances needed. Because the wetland distribution presented here is based on the National Wetland Inventory (NWI), cross-walked to our pond classifications, as noted above we are likely under-representing perennial ponds here since this category does not map well to the NWI classifications. Also, as the results of our related remote-sensing study and mapping of wetlands in Mount Rainier National Park show (see Northwest Climate Science Center report), small wetlands of any hydrologic type are substantially underrepresented by the NWI, at least for the montane regions we have studied. Therefore it is safe to assume that our assessment underestimates the number of small ephemeral, intermediate, and perennial wetlands in our three focal landscapes. The degree to which these errors of omission differ across those landscapes is an open one that we now have the tools to answer were LiDAR coverage, for example, to become available for North Cascades and Olympic National Parks. Nevertheless, the NWI is a good starting point to begin to assess areas of highest risk to amphibians based on the combination of wetland types and the degree of risk associated with climate change. As LiDAR becomes available for North Cascades and Olympic National Parks, the methods used in Mount Rainier may be extended there to develop better estimates.

Adaptive capacity: A promising approach to building resilience in montane wetland ecosystems is the possibility of targeting removal of introduced fish – known to have strong negative effects on a suite on native montane species – to regions where removals could restore habitat that is otherwise not available to amphibians and other native species affected by fish. The dominant effect of fish in our results, despite the very small sample of sites with fish in our surveys (~5%) supports the preponderance of evidence that introduced fish harm native montane ecosystems. Likewise, the demonstrated success of fish removals, and rapid unassisted recolonization by native amphibians and invertebrates shows the real potential of this approach for getting ahead of negative climate impacts (Ryan et al. 2014). The resources provided here can help Park managers and other land managers identify, for example, priority regions where habitat loss and fish presence are projected to most severely interact. Our team is working on further assessment of this as well.

A key area of uncertainty is in the biological capacity of amphibians to respond to climate impacts. Amphibians as a group are highly adapted to variable conditions, but little is known about the plasticity of alpine amphibians in response to climate impacts. For example, for our focal species, there are zero published studies of responses of different life history stages to climate-related impacts. A primary question in terms of breeding success is whether faster tadpole or larval development in ponds with increased water temperatures could compensate for faster pond drying rates. Observations of stranded tadpoles in dried ponds suggest that selective pressure for faster development is there in some years but plasticity is currently insufficient in many cases. For example, 2012 and 2013 were climate analog years in terms of the degree of drying in montane and alpine regions of the Pacific Northwest, and we observed substantial mortality of tadpoles due to pond drying in both years. In wetlands not at risk of drying entirely, ecological effects may also depend on how thermal conditions in ponds change as the climate warms and water levels drop (O'Regan *et al.*, 2013; Ryan *et al.*, 2014; Tarr & Babbitt, 2008). Overall, the potentially complex demographic effects of climate change on different life history stages leaves ample room for research and many uncertainties (Windler and Schindler 2004; Amburgey *et al.*, 2012; Duarte *et al.* 2012; Gerick *et al.* 2014). Therefore proactive management approaches to building resilience (aka adaptive capacity) provide some insurance in the face of those uncertainties.

For amphibians in particular, already known to be in decline in many montane regions, climate impacts are likely to interact with non-climate threats such as disease, pollution, and the presence of introduced fish (Ryan *et al.*, 2014; Adams *et al.*, 2013; Piovia-Scott *et al.*, 2011; Knapp *et al.*, 2007; Davidson, 2004). Amphibians and invertebrates are also important prey for many montane species, so population declines in these assemblages could propagate up food webs, negatively affecting the birds, non-avian reptiles, and mammals that rely on them as prey (Epanchin *et al.*, 2010; Polis & Strong, 1996). Overall, species' exposure will depend on what kinds of wetland habitat they use, the current distribution of wetland types across landscapes, and the degree of change in spatial and temporal hydrologic patterns under future climates (Ryan *et al.*, 2014).

Beyond our three focal species, the broad range of ecological roles played by wetlands means that altered hydrology across whole landscapes will reverberate in many ways, ranging from shifts in wildlife habitat to water storage to patterns of nutrient transfer and transformation. Patterns of soil inundation, for example, determine rates of carbon sequestration and release, nitrogen transformations, and other nutrient cycles. Likewise, changes in temporal pulses of peak water affect local pond metabolism and primary productivity, the structure of plant communities, and patterns of wildlife connectivity (Mitsch & Gosselink, 2007). Montane wetlands serve as critical habitat for a wide variety of species, many of which are adapted and sensitive to particular hydrologic regimes that are projected to shift under future climates. Therefore the hydrologic shifts evident in future projections of wetland dynamics imply widespread changes in the many ecological roles served by wetlands.

Conclusion & Recommendations

Wetlands and Amphibian Monitoring

- On-the-ground observations remain key to linking sophisticated technological modeling approaches to a core understanding of hydrologic mechanisms and their associated influences on species and ecosystems. The primary limiting factors in wetland science overall is lack of data. Field based studies, and to a less extent installation and collection of dataloggers, is time consuming but irreplaceable in the insight it yields and capacity to bridge larger-scale studies to dynamics at local scales. The empirical datasets developed during this project support not only the analyses done here, but also serve as a repository of information to support future studies. A broader approach to wetlands monitoring is sorely needed to be able to develop a more rigorous, scientifically based understanding of wetland dynamics and their ecological implications.

Ecological Analyses

- Ecological models show that three focal species of amphibians that use montane and alpine ponds and wetlands have differential reliance on the four wetland hydrologic classes, and that these differences relate to their vulnerability to climate impacts.
- At highest risk, based on the sensitivity of their core habitat to climate-induced drying, is the Cascades frog, *Rana cascadae*. However, the Cascades frog's capacity to use a range of pond classes in the absence of fish suggests opportunities for resilience if enough fish-free habitat remains or is made available through management actions.
- Long-toed salamanders, *Ambystoma macrodactylum*, are also at elevated risk of negative effects of climate change on breeding habitat due to their reliance on intermediate and perennial ponds, and additionally their life history requirement of multiple years of consecutive pond inundation to successfully metamorphose in higher elevation environments.
- Northwestern salamanders, *Ambystoma gracile*, are associated with generally less at-risk pond types (perennial and permanent ponds). However, their life history vulnerability, i.e. need for ponds that retain water for multiple consecutive years in order for larvae to complete

metamorphosis, means that they will also experience elevated risk in regions with more severe climate impacts and in particular transitions from perennial or permanent to intermediate pond habitats.

Synthesis

- The combination of our four lines of research link observations on the ground that a core assemblage of wetland-reliant species (pond-breeding amphibians) are currently most reliant on the kinds of wetlands that are both a) the most commonly omitted from existing wetland maps and b) the most vulnerable to climate-induced hydrologic changes over the coming decades. Fortunately opportunities do exist to ameliorate impacts through methods with proven success such as fish removals, and these can be applied to existing management plans such as North Cascades High Lakes Fisheries Management Plan.
- Our synthetic approach also suggests that climate-related risk varies considerably across and among the three focal National Parks – Mount Rainier, North Cascades, and Olympic National Parks – thus management efforts and future research and monitoring may be targeted accordingly and applied if desired to an adaptive management framework.

Additional Assessments

- Tying the remote sensing products and climate-hydrologic models developed in our broader project to ecological data representing a variety of taxonomic groups (e.g. waterfowl, amphibians, invertebrates) is a key next step. Additionally, extending the scope of our research to include additional scales of ecological function served by wetlands (such as nutrient fluxes, carbon sequestration, etc) and how these may be affected by climate change would be exciting. Building on our extensive existing datasets, we are also in a position to generate and test hypotheses regarding the metapopulation and metacommunity dynamics of our focal species that would be of interest both to managers concerned with population viability and to questions of basic science.

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Attachment II – Budget

“Hydrologic and Ecological Monitoring of Montane Wetlands in North Cascades, Mount Rainier,
and Olympic National Parks”

Principal Investigator: Dr. Josh Lawler

Category	Description	Units	Amount
SALARIES			
	Field Crew (2 people for 7 weeks, or 280 hrs per person)	\$12.50/hour	7,000.00
		Subtotal	7,000.00
BENEFITS			
	Field crew for 2 people, 7 weeks)	16.5% or \$2.06/hour	1,153.60
		Subtotal	1,153.60
TRAVEL			
	Transportation to field sites	Mileage @ \$0.565/mile	421.72
		Subtotal	421.72
DIRECT COSTS			8,575.32
INDIRECT COSTS (17.5%)			1,500.68
TOTAL PROJECT COSTS			10,076.00