# Assessing the Effectiveness of Amphibian Mitigation on the Sea to Sky Highway:

## Population-level Effects and Best Management Practices for Minimizing Highway Impacts

Final Report

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## Executive Summary

The construction of roads has many well-documented impacts on amphibians and reptiles, including high rates of road mortality over large geographic areas. While underpass systems consisting of modified culverts and drift fencing are increasingly prescribed as mitigation measures, the effectiveness of these structures is still uncertain. In this study, we assess the effectiveness of an underpass system installed along a realignment of the Sea to Sky Highway near Pinecrest Estates, south of Whistler, B.C. The Pinecrest realignment fragmented an existing wetland complex, resulting in the salvage and relocation of 1037 amphibians, including 683 red-legged frogs (*Rana aurora*), a federally-listed Species at Risk.

During fieldwork in 2009 and 2010, we estimated annual red-legged frog road mortality by counting carcasses during roadkill surveys, and adjusting totals with detectability and deterioration probabilities. We estimated population sizes of focal ponds using capture-recapture methods, and extrapolated to the total local population size using a predictive habitat-abundance regression equation. Using road mortality rates from this study, combined with vital rates from the literature, we ran female-only Leslie Matrix models to predict the population growth rate with and without the effects of road mortality. Finally, we used remote cameras and time-lapse photography to assess amphibian use of passageways.

Roadkill surveys estimated 1483 amphibians were killed over two years, including 915 red-legged frogs. Using our 2010 local population size estimate of 1952-2335 individuals, we estimated that 16-28% of the local population suffered road mortality in 2010. Under this scenario, our models predict local population extirpation in 20-40 years. Moreover, at one wetland where "before-after" data was available, we estimated a 73-92 % population reduction from pre- highway construction in 2007 to post-construction in 2010. Remote cameras documented that only 9% of frogs and toads and 4% of salamanders used passageways, and 51% of individuals were observed climbing or jumping over fences.

These results suggest that initial mitigation measures at Pinecrest were not sufficient to mitigate population-level effects of road mortality, nor to facilitate the amphibian use of passageways. However, roadkill rates were at least 50% lower on segments with fencing compared to those without, suggesting that fencing can be effective in reducing mortality when of sufficient length and properly installed. To this end, new re-designed fencing was installed at Pinecrest in 2010. In future projects, passageways can be improved through increasing culvert diameters, installing grates, and increasing moisture input (see Appendix I for Best Management Practices). However, it may not be possible to fully eliminate residual environmental effects, as we estimated that halting the predicted population decline at Pinecrest would require a reduction in road mortality of 69-82%. As such, future highway projects which fragment high-quality amphibian habitat should be avoided. If this is not possible, alternative measures such as wildlife overpasses should be constructed, to increase the effectiveness of mitigation and thereby reduce residual environmental effects.

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# 1 Introduction

The effects of roads on amphibians and reptiles are well-documented and pervasive, and have been implicated as a contributing factor to global amphibian declines (Fahrig et al. 1995, Findlay and Bourdages 2000, Puky 2006, Andrews et al. 2008). Road construction can lead to the loss, fragmentation, and degradation of herpetofauna habitats (Trombulak and Frissell 2000, Andrews et al. 2008), as well as the isolation of populations (Corlatti et al. 2009). However, the most immediate threat to the viability of herpetofauna populations is likely the increased risk of mortality from vehicle collisions (Puky 2006). The life history of pond-breeding amphibians and reptiles cause them to be particularly susceptible to road mortality, as they are frequently required to cross roads when migrating to and from breeding sites in fragmented landscapes (Andrews et al. 2008). Their vulnerability is further increased by their relatively slow movement rates (Hels and Buchwald 2001), and their tendency to become immobile in response to approaching vehicles (Mazerolle et al. 2005).

In this study, we assess population-level effects and mitigation effectiveness of a 2-km realignment of the Sea to Sky Highway just south of Whistler, B.C., near Pinecrest Estates (“Pinecrest”). Realignment of this highway segment resulted in the fragmentation of an existing wetland complex which contained high densities of red-legged frogs (*Rana aurora*) and other amphibians. Red-legged frogs are blue-listed in B.C. (CDC 2010), and listed under the federal Species at Risk Act (of Special Concern), due to concerns regarding the loss of wetland habitat and competition with the invasive bullfrog (COSEWIC 2004).

A growing awareness regarding the potential for population-level impacts has increased the use of mitigation structures designed to minimize the negative effects of roads on herpetofauna (Langton 1989a, Puky 2003, Mata et al. 2008). The most common mitigation approach is the installation of modified culverts that act as wildlife “underpasses”. These are connected to a system of drift fences that help to prevent road mortality and facilitate movement underneath the highway. While these structures are increasingly being prescribed within Environmental Assessments (EAs), post-construction monitoring is rarely undertaken to assess if these mitigation measures are effective. Post-construction monitoring is essential to assess if mitigation is functioning as intended, and to quantify the actual level of residual effects remaining after project completion. In an adaptive management context, this information can be used in future EAs to improve prediction of potential impacts, and increase the effectiveness of associated mitigation measures. The objectives of this study were to: (1) quantify population-level effects on red-legged frogs in the Pinecrest area, (2) assess the effectiveness of culverts and fencing in facilitating amphibian movement, and (3) assess the effectiveness of fencing in excluding animals and reducing roadkill.

# 2 Project Background

The Sea to Sky Highway Improvement Project ("the Project") was established to conduct upgrades of Highway 99 from Horseshoe Bay to Whistler in advance of the Vancouver-Whistler 2010 Winter Olympics. The original EA for the project included a description of wetland habitat occurring within the Pinecrest area, but detailed wildlife surveys were not conducted (STSHIP 2003). The Projects' application for an EA Certificate was approved by the B.C. Environmental Assessment Office (EAO) on June 4, 2004 (Certificate # TO4-01).

As a condition of the Certificate, the Project committed to continue consultations with affected communities along the Sea to Sky Highway. The communities of Black Tusk and Pinecrest Estates, located just west of the study area (Figure 1), expressed concerns regarding potential safety, noise, and water quality impacts associated with the Project (STSHIP 2005). In a community meeting, residents voted by a wide margin to realign the highway 300 m to the east of the existing alignment (STSHIP 2005). The Project proceeded with an application to amend the Certificate in order to facilitate the Pinecrest realignment, which required an additional EA of the area (the "Pinecrest EA"). The Pinecrest EA included one amphibian breeding survey in 2004, and two nocturnal call-playback surveys and a follow-up daytime survey in 2005 (STSHIP 2005). These surveys resulted in observation of 13 red-legged frog egg masses and 1 adult frog, but no red-legged frogs replied to nocturnal call-playback surveys. Potential impacts listed within the assessment included loss of wetland habitat, and impacts to habitat connectivity. Impacts of mortality from vehicle collisions were not discussed within the EA (STSHIP 2005). Proposed mitigation measures included shifting the alignment to minimize encroachment into wetland habitat, installation of five amphibian/wildlife culverts to facilitate habitat connectivity, and installation of drift fencing to facilitate amphibian use of culverts.

As part of the Pinecrest EA process, comments were solicited from the "Working Group", which included the Department of Fisheries and Oceans, Environment Canada, the former Ministry of Environment (MOE), Ministry of Energy, Mines and Petroleum Resources, the Squamish-Lillooet Regional District, and the Squamish and Lil'wat Nations (STSHIP 2005). While present in the Working Group, detailed project review by MOE was limited at this stage. Taking into account the Pinecrest EA and Working Group comments, the EAO approved the Pinecrest amendment on April 24, 2006. The EAO affirmed they were satisfied "the project would not result in significant adverse effects", providing the proposed mitigation measures were followed (EAO 2006a). In their conclusions regarding potential impacts of the Pinecrest realignment, the EAO concluded that wildlife mortality was lower on the Sea to Sky highway relative to other areas of the province (EAO 2006b). This analysis focused on large mammals, however, and did not include data on amphibians or reptiles.

In response to initial salvage efforts which revealed high red-legged frog densities within impact areas (Golder 2006), MOE became actively involved, and expressed concerns that the environmental risks were significantly higher than originally predicted within the Pinecrest EA.

MOE recommended alternative options to improve mitigation, including reverting to the original alignment, or constructing a series of bridges over the entire wetland complex (Kiewit 2007). Citing budgetary constraints, the Project decided to continue with the Pinecrest realignment instead. Negotiations between MOE and the Project resulted in commitments for additional mitigation measures. These included three additional amphibian culverts (8 in total), and construction of compensation wetlands within the old highway alignment. Funds were also provided to MOE to support amphibian inventory, mitigation effectiveness monitoring (this study), wetland restoration, and to protect similar habitat in the area.

Intensive pre-construction salvage operations were initiated on May 7, 2007. Originally expected to take approximately four weeks (STSHIP 2008), salvage was not completed until October 10 (20 weeks later), due to consistently high capture rates of red-legged frogs and other amphibians (Golder 2008). In total, 1037 amphibians of six species were captured and relocated to nearby wetlands, including 683 red-legged frogs (Golder 2008). Construction commenced shortly thereafter, and resulted in a total footprint of 6.77 ha, including loss of 0.416 ha of permanent and ephemeral wetland habitat (Figure 1).

# 3 Methods

## 3.1 Roadkill Surveys

We conducted roadkill surveys along the 1.9 km Pinecrest realignment to assess amphibian carcass distribution and potential impacts to red-legged frog populations. We conducted surveys 1-2 times per week from May 6 to October 20 in 2009 (N = 23), and from May 4 to October 20 in 2010 (N = 37). Surveys were conducted throughout the day on an opportunistic basis, with the majority occurring between 9:00 and 12:00 PM. We also conducted 4 night-time surveys (initiated 1042 PM – 1230 AM) in 2010 to obtain fresh carcasses to improve assessment of species composition and aid in daytime identification of amphibian carcasses. We walked along the shoulder of both sides of the highway transect (northbound and southbound lanes), and recorded all carcasses encountered. The location of each carcass was recorded with a GPS unit, and marked on the road surface with non-toxic spray paint to avoid double-counting.

Roadkill surveys typically only detect a fraction of the total amphibians that are killed on roadways, as carcasses rapidly disappear from traffic deterioration and scavenger removal (Hels and Buchwald 2001). We sought to maximize our ability to detect as many carcasses as possible by examining all potential remains, including those not immediately identifiable as amphibian carcasses (i.e. < 1 cm diameter). These carcass remnants were examined with a hand lens in the field, and compared to reference specimens under a dissecting microscope in the lab if necessary. Small carcass remnants were almost always identifiable to taxa (amphibian, reptile, bird, mammal, or invertebrate). To further increase the accuracy of our roadkill counts, we conducted a "capture-recapture" investigation of amphibian carcasses in order to adjust counts for any bias associated with carcass detectability or deterioration. During a subset of the roadkill surveys in 2010 (August 23-October 12; N = 14 daytime surveys), we marked all fresh carcasses with unique numbers, and documented when they were observed on subsequent surveys. Marks were made with pastel markers on the highway shoulder directly perpendicular to each carcass, in order to avoid leaving marks on the carcasses themselves, which could increase their detectability.

## 3.2 Capture-Recapture: estimating population size

We conducted a capture-recapture study on red-legged frogs in 2010, with the goal of estimating local population size via habitat-abundance relationships. We conducted area-constrained searches over 10 survey days (June 2-16), during which we repeatedly sampled five "focal wetlands" 5-8 times each for adult and juvenile frogs. Focal wetlands included Angus, Brew, Gamble, Moss, and Wetland 9 (Figure 1); all of which were previously confirmed to contain high densities of *R. aurora* adults and juveniles (Golder 2008). We searched the littoral and riparian areas of each wetland with 3-5 people for approximately one hour, or until capture success declined below an acceptable level. Captured individuals were held in an aerated container with water and cover objects, and returned upon the completion of each wetland survey. We weighed and measured each individual, marked adults with PIT tags (>50 mm SVL), and marked juveniles with unique color combinations of VIE elastomer die. Individuals were checked for marks using a passive transponder to identify PIT tag codes, and a UV light to identify elastomer die markings. We followed best practices during searching and handling in order to prevent transmission of pathogens (MOE 2008).

## 3.3 Remote Cameras

We installed remote cameras at passage entrances in 2009 and 2010 to investigate the degree of use by amphibians and other wildlife. Six RECONYX© MC65 cameras were installed at passages 1-6 in 2009 (May 21–October 7; Figure 1), and six MC65 and two PC900 cameras were installed at passages 1-8 in 2010 (May 4 – October 5). We installed cameras approximately 2-5 meters from passage entrances, in order to detect animals that approached but did not pass through passageways. Cameras were attached to steel rods that were driven into the ground, and stabilized with metal "guy-wires" attached to steel spikes. We used a combination of motion sensitive infrared sensors and time-lapse photography to assess passage use. Infrared sensors were activated throughout the 24 hour period to capture avian and mammalian activity, whereas time-lapse photography was activated from dawn to dusk of each day to capture cold-blooded amphibians and reptiles (which did not trigger infrared sensors). We used time-lapse photography (1 minute intervals) in the fall of 2009 (September 1-October 7) and throughout the spring and fall of 2010 (May 4 – October 5).

# 4 Data Analysis

## 4.1 Total annual roadkill

*Estimating total roadkill.* – We used Jolly-Seber models (Pollock et al. 1990) to calculate correction factors in order to estimate annual roadkill totals from raw carcass counts. Jolly-Seber models allow estimation of parameters for open populations, where additions (births and immigrants) and permanent deletions (deaths and emigrants) occur during the sampling period (Pollock et al. 1990). For our purposes, additions to the roadkill "population" occurred when new amphibians were killed on the highway, and deletions occurred when carcasses deteriorated until they were no longer detectable. Using the dataset of marked carcasses, we used "recapture-only" models in Progam MARK (Cooch and White 2010a) to estimate amphibian carcass encounter probability (*p*; probability of detection if present) and persistence probability (*phi*; daily probability of persisting on road and "surviving" until the next sampling period). We assumed *p* and *phi* were constant between species (see below), age classes, and over time (i.e. between surveys). To help meet the time-constant assumption, we removed sampling periods with very low encounter rates. The 95% confidence intervals of *p* and *phi* estimates overlapped broadly when compared between constant and time-varying models, further supporting this approach.

We adjusted each survey total to account for any carcasses additions between sampling occasions, using the following equation:

$$B_i^* = B_\mathrm{i} \frac{\log \phi_i}{\phi_i - 1}$$

where $B_\mathrm{i}^*$ is the maximum likelihood estimator for total carcass "additions" between sampling occasions *i* and *i* + 1, $B_i$ is the observed number of carcasses at sampling occasion *i* + 1, and $\phi_i$ is the cumulative product of *phi* (as estimated above) for the interval between sampling occasions (Schwarz et al. 1993). This equation assumes uniform carcass "recruitment" and persistence between sampling locations, which is appropriate in our case, where persistence rates were relatively high (Schwarz et al. 1993). Finally, we adjusted these total roadkill counts for encounter probability (total/encounter probability).

*Species composition.* – We estimated the proportion of total amphibian roadkill that were red-legged frogs from the subset of fresh carcasses that were identifiable to species (N=94). This approach assumes that carcass encounter and persistence probabilities are equal between species, as differences between species could cause identifiable carcasses to be an inaccurate representation of amphibian roadkill totals. To test this assumption, we ran additional Jolly-Seber models as above, using only the subset of carcasses that were initially identifiable to species. We grouped carcasses into three species groups: red-legged frogs, western toads, and "Other" (including Pacific tree frogs, long-toed salamanders, and rough-skinned newts). The Other category was necessary because none of these species occurred with sufficient frequency to estimate their parameters separately. We compared four models that specified all combinations of *p* and *phi*, including: 1) $p_{(.)}phi_{(.)}$, 2) $p_{(.)}phi_{(g)}$, 3) $p_{(g)}phi_{(.)}$, and 4) $p_{(g)}phi_{(g)}$, where, $x_{(g)}$ = variation by species group, and $x_{(.)}$ = constant by species group. We ranked each of these models using information-theoretic methods, which estimate the formal likelihood of each model in relation to others in the candidate model set (Burnham and Anderson 2002). Models were ranked according to Akaike's Information Criterion (AICc), which balances model fit with the number of parameters used. Models were ranked in MARK, which calculated AICc for each model, ΔAICc, the difference in AICc between the *i*th model and the model with the lowest AICc, and Akaike weights ($\acute{\omega}_i$), defined as the likelihood of each model, given the candidate set of models (Burnham and Anderson 2002).

## 4.2 Local population size

*Capture-recapture*.– We used closed population capture-recapture models (Otis et al. 1978) to estimate population size within each focal wetland. In contrast to Jolly-Seber models, closed population models assume no immigration, emigration, births or deaths between sampling periods. Sampling over a relatively short time period in mid-spring helped to meet this assumption, as movement between wetlands, as well as births or deaths, are likely very rare during this interval. Sampling within this season also eliminated the possibility of "births" from emerging metamorphs, or emigration from juvenile dispersal, both of which occur in the late summer and fall (Calef 1973a).

We ran closed population capture-recapture models in Program MARK to estimate capture probability ($p_c$; probability of capturing a frog if present), and population size (*N*) within each of the 5 focal wetlands, for both adults and juveniles. We compared two models that specified $p_c$ to be either varying or constant between adults and juveniles, and ranked them using information-theoretic methods, as above. To avoid over-parameterization of our models, we assumed constant encounter probability between sampling periods, and equal initial encounter and recapture probabilities (no 'trap effect'). Finally, we compared population estimates from the best model above to $M_h$ models (Chao 1987) which allow individual heterogeneity in capture probability, to test if this potential source of bias was a factor in our study. Individual heterogeneity in capture probability arises when certain individuals are easier to detect and capture relative to others, which violates the standard assumption of equal probability of capture (Pollock et al. 1990, Cooch and White 2010b). Violating this assumption negatively biases estimates of $p_c$, causing *N* to be underestimated.

*Using habitat to predict local population size*.– We combined abundance estimates from above with habitat estimates to predict local population size of Red-legged frogs within the study area. We quantified total area of "high quality" habitat (see below) within a 300 m buffer surrounding the new highway segment. This boundary of the local population is intended to represent the area within which red-legged frog survival may be impacted by road mortality (i.e. includes home ranges of frogs that intersect with the highway). While red-legged frogs are known to occasionally migrate long distances, the majority of individuals remain very close to aquatic sites throughout the year (Bulger et al. 2003). As such, 300 m is likely a conservative estimate of the local population boundary, and effectively encompasses the range of movement distances observed for resident red-legged frogs (Calef 1973b, Bulger et al. 2003, Chan-McLeod and Moy 2003). This boundary also includes the vast majority of wetland habitat in the study area, which is bound by Daisy Lake to the East, and residential development to the West (Figure 1).

To estimate the total area of "high-quality" red-legged frog habitat within the population boundary, we used a combination of GPS tracking in the field, and delineation in ArcGIS 9.3 using high-resolution orthophotos. We defined high quality habitat as portions of wetlands < 1 metre in depth with emergent vegetation, and with surrounding riparian areas that provided protective vegetative cover. Using capture-recapture abundance estimates and habitat amounts for each focal wetland, we ran a univariate linear regression to predict number of frogs from habitat area. We constrained the y-intercept through the origin such that zero habitat area predicted zero frogs. Using this regression equation in JMP, we then predicted the local population size (±95% Individual Confidence Limits), using the total amount of red-legged frog habitat within this boundary.

*Comparison to salvage estimates*.– We compared our 2010 estimate of Wetland 9 population size to capture totals from pre-construction salvage operations in 2007. While this comparison was not part of the original objectives of this study, the availability of systematically-collected population data at this site, both before and after highway construction, provided a rare opportunity to empirically assess population effects of this impact. Capture totals from salvage operations are likely to be accurate population estimates because of the extensive amount of effort involved, and the rigorous exclusion and marking protocols (see section 6.2). Likewise, we implemented capture methods and data analyses to meet model assumptions and account for potential sources of bias, thereby maximizing the accuracy of our capture-recapture population estimates. Consequently, we contend that this trend is an accurate reflection of population changes that occurred on site over this 3-year period.

## 4.3 Population growth rate

We ran deterministic female-only Lesley Matrix population models in Program R to assess the potential effects of roadkill on the local population growth rate. We used published vital rates as estimated from a relatively undisturbed 'control' populations that were unaffected by road mortality (Briggs and Storm 1970, Calef 1973a, Licht 1974, Biek et al. 2001)(Table 1). Using these rates, we compared the dominant eigenvalue of our models (population growth rate; $\lambda$) with and without the negative effect of road mortality on survival ("roadkill survival"). Roadkill survival was defined as the additive reduction in survival probability due to road mortality. We calculated roadkill survival as the inverse of annual roadkill mortality rates, which were calculated from estimates of total roadkill and local population size. We estimated range in roadkill survival using the 95% Individual Confidence Limits from our estimates of local population size, and used this in our models to predict the range in population growth rates. Our Leslie Matrix models were constructed such that roadkill survival was the same for each terrestrial life stage. This assumption was supported by a comparison of the relative proportions of adults and juveniles in our population models (19% and 81% respectively), which were very similar to the proportions of these life stages in the roadkill dataset (17% and 83%). Using the estimate of local population size for 2010 as a starting point, we then projected the population forward over 40 years, using the range in population growth rates as estimated above. We also conducted an exploratory analysis to assess what reduction in annual road mortality rates would be required to increase the population growth rate above the placement levels (i.e. $\lambda > 1$). Finally, we calculated the elasticity of each vital rate, defined as the effect of proportional changes of each parameter on the value of $\lambda$, to assess the influence of roadkill mortality on population growth relative to other vital rates.

## 4.4 Roadkill Distribution

In order to assess factors that determined variation in roadkill rates along the Pinecrest realignment, we divided the surveyed portion of the highway into 33 - 50 m segments, and summed the total number of amphibian roadkill counted within each segment for each year. Using the GPS locations of natural and constructed barriers, we recorded the type of barrier that was present on the wetland-side of the highway for each 50 m segment (west side in all cases). Barrier categories included drift fencing, retaining walls at wetland edges, cliffs (ca. 3 to 5 m high), concrete roadside barriers, no barriers, or partial barriers (extending along only ¾ of the segment or less). Where concrete barriers overlapped with coverage of retaining walls or drift fencing, the latter two barrier types were given precedence in classifying that segment, as these were assumed to be most impervious to amphibian movement. We ran a linear model in JMP 7.0 (SAS 2007), with the pooled dataset over 2009 and 2010, to predict the influence of year, barrier type, distance to nearest wetland, and distance to nearest passageway, on total amphibian roadkill for each segment. Our model also included first-order interaction terms between year and the other predictor variables.

## 4.5 Remote cameras

We examined a total of 791,634 photographs taken by remote cameras in 2009 and 2010, and tabulated all detections of vertebrate species. We defined independent "detections" as a photograph of an individual animal (or family group), or a series of consecutive photographs separated by intervals of 5 minutes or less. For each detection, we recorded species, date, time, whether the animal used the passageway, and any passage avoidance or escapement behaviours. Animals were considered to have used passageways if they entered or exited the culvert without returning in the opposite direction.

# 5 Results

## 5.1 Combining roadkill and local population size to assess population-level effects

*Total annual roadkill*.– Our Jolly-Seber model produce relatively high estimates of carcass encounter probability ($p$ = 0.63±0.08) and carcass persistence probability ($phi$ = 0.85±0.02). This high value of $p$ confirms the robustness of this model to potential bias from individual heterogeneity in encounter probability (Cooch and White 2010b). In our comparisons of species groups, $p_{(.)}phi_{(.)}$ was the highest ranked model (ΔAICc = 0.00, ώ = 0.57), providing strong support for equal carcass encounter and persistence probabilities between species groups. This model had was more than twice the likelihood of the next highest-ranked model ($p_{(.)}phi_{(g)}$; ΔAICc = 2.31, ώ = 0.18).

We observed a total of 221 amphibian carcasses in 2009 (N=21 surveys), and 413 carcasses in 2010 (N=37 surveys). Correcting for the estimates of $p$ and $phi$ above, as well as estimating carcasses additions between sampling occasions as per Schwarz et al. (1993), we estimated total annual amphibian roadkill to be 606 and 877 carcasses for 2009 and 2010, respectively. Assuming 62% of amphibian carcasses were red-legged frogs, total red-legged frog roadkill estimates were 374 and 541 carcasses for 2009 and 2010, respectively.

*Local population size*.– The highest-ranked closed capture model for estimating population size in focal wetlands assumed equal $p_c$ for adults and juveniles ($p_c$ = 0.15±.02; ΔAIC = 0, ώ = 0.75). This model had almost three times the likelihood of the model assuming unequal $p_c$ by age (ΔAIC = 2.04, ώ = 0.266). This model demonstrated that adults and juveniles were distributed unevenly the throughout the focal wetlands (Table 2a). However, because our preliminary results indicated heterogeneity in individual capture probability, we ran $M_h$ models instead (Table 2b), which account for this potential source of bias (Chao 1987, 1989). Our study met the requirements for $M_h$ models, which require relatively low capture probability and at least 5 sampling occasions (Chao 1987). Using estimates produced from these models, we found a statistically significant, positive relationship between habitat area and frog abundance in our focal wetlands ($\beta$=0.05 ± 0.005, $F_{1,3}$=110.47, P= < 0.001, $R^2$= 0.71). Using the total amount of habitat available within the study area (4.99 ha), we estimated the 2010 local red-legged frogs population (±95% Individual Confidence Limits) at 2653 (1952-3355).

*Comparison to salvage estimates*.– Our capture-recapture study estimated 2010 Wetland 9 population size to be 44 (95% CL = 30-96 individuals; Table 2b). Given that 357 red-legged frogs were captured and relocated in 2007 at this location (Golder 2008), this represents a 73 - 92% population reduction over this three-year time period.

*Population growth rate*.- Without the influence of road mortality on survival, our matrix models predicted an expanding population ($\lambda$=1.05). However, our 2010 roadkill totals represent an annual road mortality rate of 16 - 28% of the local population. Adding this "roadkill survival" effect (.72 - .84) to our population models produced population growth rates below replacement levels (0.75 - 0.88). These population growth rate estimates predict that the local population will be extirpated in 20 - 40 years (Figure 2). Further, our exploratory analysis indicated that population growth would not exceed replacement levels until road mortality was reduced to 5% of the local population, which equates to a 69-82% reduction from current levels. Elasticity was highest for roadkill survival (1.00) and adult survival (0.49), and considerably lower for the remaining vital rates (all = 0.17; Figure 3).

## 5.2 Roadkill Distribution

Highway segments with different types of barrier coverage had significantly different mean counts of amphibian roadkill ($F_{65,5} = 3.93$, $P < 0.01$; Figure 4). Mean roadkill (± SE) was at least 50% or lower on segments with fencing coverage (4.06 ± 2.21) compared to segments with no barriers (12.50 ± 1.58) or partial barriers (11.30 ± 1.16; Figure 5). Mean roadkill was also lower on segments with cliffs (4.30 ± 3.51), retaining walls (3.40 ± 2.64), and concrete barriers (6.28 ± 2.17) compared to segments with no or partial barriers. Mean roadkill per segment appeared to be slightly higher in 2010 relative to 2009, although this difference was not statistically significant at $\alpha$=0.05 ($F_{65,1} = 3.35$, $P = 0.07$). Mean roadkill did not vary significantly with respect to distance to nearest wetland or passageway (both $p > 0.70$). Likewise, there were no significant interactions between years and the other model variables (all $p > 0.40$), indicating that roadkill distribution patterns did not differ between years.

## 5.3 Remote Cameras

In 2010, 94% of anurans (frogs and toads) captured on remote cameras were identified as red-legged frogs, 2% were possible Pacific treefrogs (*Pseudacris regilla*), 2% two were not identifiable to species, and 1% was a western toad (*Anaxyrus boreas*)(N=100). The majority of anuran occurrences (84%) were at passages 6-8, with a clear trend of higher occurrence rates at northern passages (Table 3; passages are numbered from south to north). A total of 36 salamanders were captured on remote cameras in 2010 (Table 3). Anurans were only observed moving through passageways in 9% of cases where usage was discernible (N=79; Table 3). Passage six had the highest proportion of use (20%; N=15), followed by passage five (11%; N= 9), passage seven (6%; N= 18), and passage eight (6%; N=32). In contrast, passages 1-4 were never used. Moreover, anurans were observed “escaping” over fences in 51% of occurrences (N=79). Escapement included either climbing over fences, or jumping to the top of fences and then over. Salamanders were also highly concentrated at northern passageways, with 81% occurring at passage 8 (N=36; Table 3). Salamanders used passageways even less frequently than anurans (4%; N=26), with only a single case of use at passage 8 (Table 3).

Patterns of passage use and occurrence were similar in 2009. Frogs were only observed using passageways in 9% of cases (N=20; Malt 2010a), with 70% of occurrences at passage 6, 20% at passage 4, and 10% at passage 3 (note that remote cameras were not installed at passages 6 or 8 in 2009, and time-lapse photography was only conducted in the fall). Frogs were observed escaping the fencing system by either climbing over fences or passing through fencing gaps in 28% of cases (N=20). Remote cameras also captured a wide variety of mammalian and avian species. In general, small and medium-sized mammals were more likely to use passages compared to large mammals, birds, or herpetofauna (see Malt 2010a for more details).

# 6 Discussion

## 6.1 Population-level Effects

Despite the installation of mitigation measures, we documented two years of consistently high road amphibian mortality along the Pinecrest realignment. We estimated that a total of 1503 amphibians were killed on the highway over two years, including 928 red-legged frogs. Based on 2010 road mortality rates, our models suggest that this will cause the local population growth rate to be below replacement levels, and predicts extirpation in 20-40 years. Extirpation in this case means that this population will likely become a ‘sink’ (birth rates < mortality rates), and thus will require immigration from surrounding areas to be sustained over the long term (Pulliam 1988). In addition, the elasticity of roadkill survival was at least twice that of any other vital rate, suggesting that road mortality had a disproportionately strong influence on population growth relative to other vital rates. Taken together, these results suggest that significant population-level impacts occurred at Pinecrest, despite the presence of extensive mitigation measures.

The strong effect of road mortality on population growth stemmed from the fact that roadkill appeared to affect survivorship of all post-metamorphic life stages equally. Our matrix models were thus constructed to reflect our empirical observations, which demonstrated that proportions of life stages within roadkill were virtually identical to that of the overall population (section 4.3). As such, differences in behaviour or life history between adult and juvenile red-legged frogs apparently did not cause any in differences in vulnerability to road mortality at Pinecrest. This pattern of road mortality has the potential to cause impacts of high magnitude on population viability, due to the cumulative effects of reductions in survivorship to multiple life stages simultaneously. For example, reduced juvenile survivorship could reduce recruitment to the reproductive adult stage, and this reduced survivorship in turn could decrease the ability of adults (whose survivorship is also reduced) to contribute to successive generations.

Our population projections are based on 2010 roadkill rates, which occurred after installation of initial mitigation measures as designed during the EA process. As discussed above, our results indicate that these initial mitigation structures were insufficient to prevent significant impacts to red-legged frog populations at Pinecrest. Ministry of Transportation and Infrastructure (MOT) installed new, re-designed fencing in the fall of 2010 to rectify some of these deficiencies (Malt 2010a; see Appendix II for design details). While this likely represents an improvement of mitigation at Pinecrest, the ability of this fencing to reduce road mortality to a level which can halt the decline of the local population is still unknown. Accomplishing this goal may be challenging, given that it appears that annual road mortality will need to be reduced by approximately 70-80% in order for population growth to exceed replacement levels. This highlights the disproportionately high impact road mortality can have on amphibian population viability, and underscores the need to consider effects of this magnitude when conducting EAs of future highway projects.

## 6.2 Evidence of current population reductions

Our predictions regarding the potential for long-term population declines warrant significant concern with respect to conservation of the local red-legged frog population at Pinecrest. However, our results also provide evidence that localized population reductions may have already occurred. At Wetland 9, 357 unique individuals were captured in 2007, whereas our models estimated that only 30-96 occurred there in 2010 (a 73-92% decline). The large magnitude of this apparent decline is concerning, particularly because it is correlated with substantial habitat loss and degradation over this time period.

While populations before and after highway construction were estimated using different methodologies, we contend that the rigorous methods used in both cases provide robust and biologically meaningful results. During the 2007 salvage operations, Wetland 9 was searched repeatedly over a 3.5 month period (472 person-hours), during which all frogs were captured, marked, and relocated until there were no captures on three consecutive days (Golder 2008). The wetland was also surrounded by exclusion fencing to prevent individuals from leaving the area. As such, the salvage population estimate is likely to be closest possible representation of the "true" population size in 2007, and it can be assumed that detectability of any given individual over the entire salvage period was very high. Likewise, population sizes in each pond from our capture-recapture models were adjusted for detectability and individual heterogeneity in capture probability, thereby maximizing the accuracy of these estimates. Finally, our estimated range in population reduction is derived from Confidence Limits produced by our population models, and as such, the estimated range in population decline we present explicitly incorporates any uncertainty inherent within our methods.

The apparent decline in the red-legged frog population at Wetland 9 occurred over a time period where substantial habitat impacts from highway construction were observed at this site. Firstly, the eastern portion was infilled, resulting in a reduction of wetland size of approximately 50% (Figure 1). The hydroperiod of the remaining wetland has also been reduced, likely due to increased drainage caused from the permeable retaining wall built at the road-wetland interface. As a result, vegetative succession has increased drastically, with terrestrial forbs and trees rapidly colonizing previously wetted areas (pers. obs.). Similar impacts occurred at Wetland 5B and other locations along the realignment (Figure 1). In retrospect, these two wetlands were among the highest quality non-breeding (ephemeral) habitat at Pinecrest, which provides key foraging and thermoregulatory habitat for both adult and juvenile red-legged frogs.

Based on the above observations, habitat impacts, combined with road mortality, are the most likely causes of this apparent population reduction at Wetland 9. However, other factors may have also played a role. For instance, it is possible that this wetland has not yet been fully re-colonized since the 2007 salvage, and therefore is still below carrying capacity. Alternatively, population size could have been low in 2010 due simply to natural population fluctuations, as stochastic factors can cause amphibian populations to vary widely in amphibians from year to year (Hallett et al. 1991). In either case, this wetland is unlikely to return to pre-disturbance population levels in the foreseeable future, as its' carrying capacity has likely been significantly decreased from reductions to the amount and quality of habitat it contains. As such, this potential population decline remains a cause for strong concern.

## 6.3 Effectiveness of amphibian passageways

Amphibian movements captured on remote cameras indicated that amphibians were very hesitant to use passageways at Pinecrest, with only 9% of anurans and 4% of salamanders observed passing through culverts. Moreover, anurans were repeatedly observed avoiding passageways, including climbing/jumping over fences and culvert structures in 51% of cases in 2010. These results indicate that the underpass system as originally designed at Pinecrest was not effective in facilitating safe movements of amphibians between habitats on either side of the highway.

Hesitancy of amphibians to enter artificial passageways has been observed previously, and can be largely attributed to differences in environmental conditions between passage interiors and adjacent habitat. For instance, there is evidence that the lack of light within tunnels may deter usage by amphibians (Jackson and Tyning 1989, Krikowski 1989). Consequently, larger tunnels are required to provide sufficient light to facilitate passage use, particularly for longer tunnels which allow less light penetration (Puky 2003). As such, the diameter-to-length ratios of amphibian tunnels at Pinecrest may not have been large enough to provide sufficient light (see Appendix II). Cooler temperatures within tunnels relative to the outside environment may also have deterred amphibian use of passages at Pinecrest (Langton 1989b).

The lack of moisture within passageways at Pinecrest may also have caused amphibians to be hesitant to use these structures. While amphibians are clearly dependent on moist habitats for movement, they may be reluctant to use flooded culverts during terrestrial movements (Jackson and Griffin 2000). As a consequence, it is generally recommended the tunnels be placed above the water table (Puky 2003). However, this has led to the unintended consequence of passageways being placed "high and dry", where they receive little to no moisture input throughout the year. This is particularly evident at Pinecrest, where substrate within culverts was extremely dry throughout periods of amphibian movement (pers. obs.). Innovative solutions are required to find the balance between providing a beneficial amount of moisture input in amphibian passageways, while simultaneously avoiding flooding (Jackson and Griffin 2000).

The location of passageways within the landscape is one of the most important factors influencing passageway use (Podlucky 1989, Jackson and Griffin 2000). Unless passageways are placed within natural migration and dispersal routes of target amphibian species, they are unlikely to be used, even if design features make them otherwise suitable. Heterogeneous use of the landscape was certainly a factor of Pinecrest, where the vast majority of amphibian movements appeared to be concentrated within the northern portion of the study area (Passages 6-8; Table 3). This was likely related to the distribution of habitat and topographical features that influence amphibian distribution and movement patterns. At a smaller scale, passages that were associated with the highest occurrence and usage rates of amphibians all had high amounts of forest cover overhead culvert entrances (pers. obs.), which may positively influence passage use (Jackson and Griffin 2000). Thus, it is important to consider site- and landscape-scale habitat distribution, local topography, and movement behaviour of target species when determining the optimal locations for amphibian passages.

The design and installation of fencing structures will also influence the willingness of amphibians to use underpasses. Consequently, mitigation strategies should consider passages and fences as complementary elements of a larger system; as the success of one depends on that of the other. Successful fencing systems can result in large reductions in mortality, while simultaneously facilitating increases in passageway use (Dodd et al. 2004, Aresco 2005). The angle of fencing is important in this respect, as fencing installed parallel to the highway may effectively exclude animals, but not direct them towards passageways. An ideal design may be “zig-zag” pattern (Jackson and Tyning 1989), which directs animals towards passageways at all points of contact, while simultaneously excluding animals. Moreover, holes or gaps through which amphibians are able to “escape” the system must be eliminated to the greatest extent possible, particularly at culvert-fencing junctions. Escapement at these fencing-culvert junctions was a recurring issue at Pinecrest. At least 28% of individuals were observed escaping through or over the fencing at these locations in 2009, and 51% of anurans were observed climbing or jumping over fencing in 2010. In 2009, fencing was either not securely attached, or in many cases not attach at all to culverts, thereby providing easy routes of escapement (Malt 2010c, b). While attachments were improved in 2010, attaching fencing to the curved surface of culverts often resulted in angled fencing that was easily climbable by amphibians. Consequently, it is clear that the culvert-fencing connection is a key component of underpass systems, and ensuring this juncture is impermeable to target species is critical to their success.

## 6.4 Effectiveness of fencing design

As demonstrated by our results of population-level effects of road mortality, the original fencing design at Pinecrest was insufficient to mitigate negative impacts to red-legged frogs and other amphibians. However, mortality rates were reduced by at least 50% on road segments with sufficient lengths of fencing, suggesting that fencing can be effective when it provides sufficient highway coverage. Without this coverage, amphibians will simply circumvent the fencing to get across the highway (Ryser and Grossenbacher 1989). The distribution of roadkill at Pinecrest suggests that fencing coverage should be at least 200-300 m in length to effectively exclude amphibians (Figure 4). As such, amphibian fencing must be designed and installed to fully exclude animals from highways while simultaneously directing them towards passageways.

While road mortality rates were lower along segments with sufficient fencing coverage, mortality still occurred in these areas of the highway. This indicates that amphibians were able to escape through holes or gaps caused by deficiencies in fencing installation. Indeed, MOE inspections at Pinecrest documented over 70 deficiencies relating to fencing installation and maintenance (Malt 2010c, b). These deficiencies included not properly burying fencing to required depths, and inadequately connecting fencing to culverts, both of which resulted in numerous escapement opportunities for amphibians. Indeed, culvert-fencing connections were the site of virtually all amphibian escapement captured by our remote cameras (see above). The durability of the original fencing design was also a major issue, as the wire mesh was highly susceptible to tearing and collapse, especially as a result of snow accumulation from winter ploughing. The new fencing installed in the fall of 2010 was constructed of more durable materials, and initial observations demonstrate it is more resistant to the impacts of snow removal and the effects of weather in general.

As discussed above, a key function of fencing systems is to exclude animals from the highway. However, if amphibians are hesitant to use passageways, as was the case at Pinecrest, effectively excluding animals may result in the unintended consequence of genetically isolating populations (Reh 1989, Corlatti et al. 2009). Nonetheless, prioritizing exclusion may be the only viable option to sustain the local population, particularly when road mortality rates are high, such as documented by Aresco (2005). This partial isolation may not unduly impact genetic connectivity of local populations, as there is evidence that as little as one migrant per generation may be sufficient to prevent inbreeding depression (Wang 2004). However, all practical efforts should still be made to facilitate population connectivity, including proper design and placement of passages. Maintaining connectivity will be particularly important where small, isolated populations depend upon immigration from adjacent populations to be “rescued” from extinction (Pulliam 1988).

## 6.5 Conclusions

The results of monitoring in 2009 and 2010 indicate that mitigation measures, as originally designed and installed, were not effective in eliminating residual environmental effects at Pinecrest. We estimated that a total of 1503 amphibians were killed on the highway over two years, including 928 red-legged frogs. In 2010, we estimated that 16-28% of the local population was killed on the highway. Our population models predict that this will result in a declining population, with extirpation occurring in 20 to 40 years. In addition to the effects of road mortality, infilling of wetlands and subsequent hydrological changes resulted in significant reductions to ephemeral habitat, which was correlated to a 73 - 92% decline in red-legged frog abundance at one site. Passageway systems also appeared to be ineffective, as only 9% of anurans and 4% of salamanders were observed passing through culverts. Finally, there were numerous issues with the installation and maintenance of fencing structures, which was associated with amphibian escapement and associated road mortality.

Despite these negative findings, there are measures that can be taken to improve mitigation at Pinecrest and other similar projects (see Appendix I). For instance, our results indicate that if fencing provides sufficient coverage, and is properly designed, installed, and maintained, it can significantly reduce road mortality rates. There are also a number of improvements that can be made to passageways that will increase the probability of their use, including maximizing their diameter to length ratios, installing metal grates or light tunnels, and increasing moisture input. However, while the effectiveness of mitigation measures can clearly be increased, our results indicate that it is likely not possible to fully eliminate residual impacts associated with highway projects such as Pinecrest. As such, future highway projects that propose to infill and fragment amphibian habitats should be avoided where possible, particularly in cases like Pinecrest, where viable alternatives exist. Where avoidance is not possible, alternative designs which more fully mitigate environmental impacts should be incorporated into product design, including wildlife overpasses, viaducts and expanded bridges (Jackson and Griffin 2000). These alternative mitigation measures will help to minimize habitat loss and degradation, and provide relatively undisturbed conditions to facilitate safe passage of amphibians and other wildlife.

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# 8 Tables and Figures

**Table 1.** Vital rates for *R. Aurora* used in Lesley Matrix population models, as estimated from proportion of the local population killed annually for Road Survival, and from published literature for all others.

| Vital rate | Description | Value |
|---|---|---|
| Se | Embryo survival | 0.92 |
| Sl | Larval survival | 0.03 |
| Sfw | First overwinter survival | 0.55 |
| S2 | Year 2 survival | 0.36 |
| S3 | Year 3 survival | 0.36 |
| S4 | Year 4+ survival | 0.69 |
| F4 | Year 4+ fecundity | 303 |
| Srd | Road survival[1] | 0.72 - 0.84 |

**Table 2.** Population estimates for focal ponds in 2010, as estimated by: a) a closed-capture model, and b) a $M_h$ model accounting for individual heterogeneity in capture probability.

| **Wetland** | **a) Closed Capture Model** | | | **b) $M_h$ Model** | | |
|---|---|---|---|---|---|---|
| | Adults | Juveniles | Total | Adults | Juveniles | Total |
| **Angus** | 16 (13-26) | 22 (18-24) | 38 (31-50) | 31 (15-114) | 30 (19-77) | 61 (34-191) |
| **Brew** | 2 (2-10) | 29 (24-43) | 31 (26-53) | 2 | 40 (26-90) | 42 (26-90) |
| **Gamble** | 3 (3-3) | 18 (13-31) | 21 (16-34) | | 11 (10-18) | 11 (10-18) |
| **Moss** | 1 (1-7) | 29 (23-44) | 30 (24-51) | 1 | 42 (24-118) | 43 (24-118) |
| **Wetland 9** | 9 (6-19) | 34 (26-52) | 43 (32-71) | 5 (5-7) | 39 (25-89) | 44 (30-96) |
| **Totals** | 31 (19%) | 132 (81%) | 163 | 39 (19%) | 162 (81%) | 201 |

**Table 3.** 2010 amphibian passage use documented by remote cameras using time-lapse photography.

| | **Anurans (Frogs & Toads)** | | | | **Salamanders** | | | |
|---|---|---|---|---|---|---|---|---|
| **Passage** | Use | Usage obs.[1] | %Use | Total occ.[2] | Use | Usage obs. | %Use | Total Occ. |
| **1** | 0 | 1 | 0% | 1 | | 0 | | 0 |
| **2** | 0 | 1 | 0% | 2 | | 0 | | 0 |
| **3** | 0 | 1 | 0% | 1 | | 0 | | 0 |
| **4** | 0 | 2 | 0% | 3 | | 0 | | 1 |
| **5** | 1 | 9 | 11% | 9 | | 2 | | 3 |
| **6** | 3 | 15 | 20% | 20 | | 0 | | 0 |
| **7** | 1 | 18 | 6% | 19 | | 3 | | 3 |
| **8** | 2 | 32 | 6% | 45 | 1 | 21 | 5% | 29 |
| **Total** | 7 | 79 | **9%** | **100** | **1** | **26** | 4% | **36** |

[1]Usage observations are a subset of total occurrences where whether an animal used a passageway was discernible.
[2]Total occurrences include all observations captured by remote cameras.

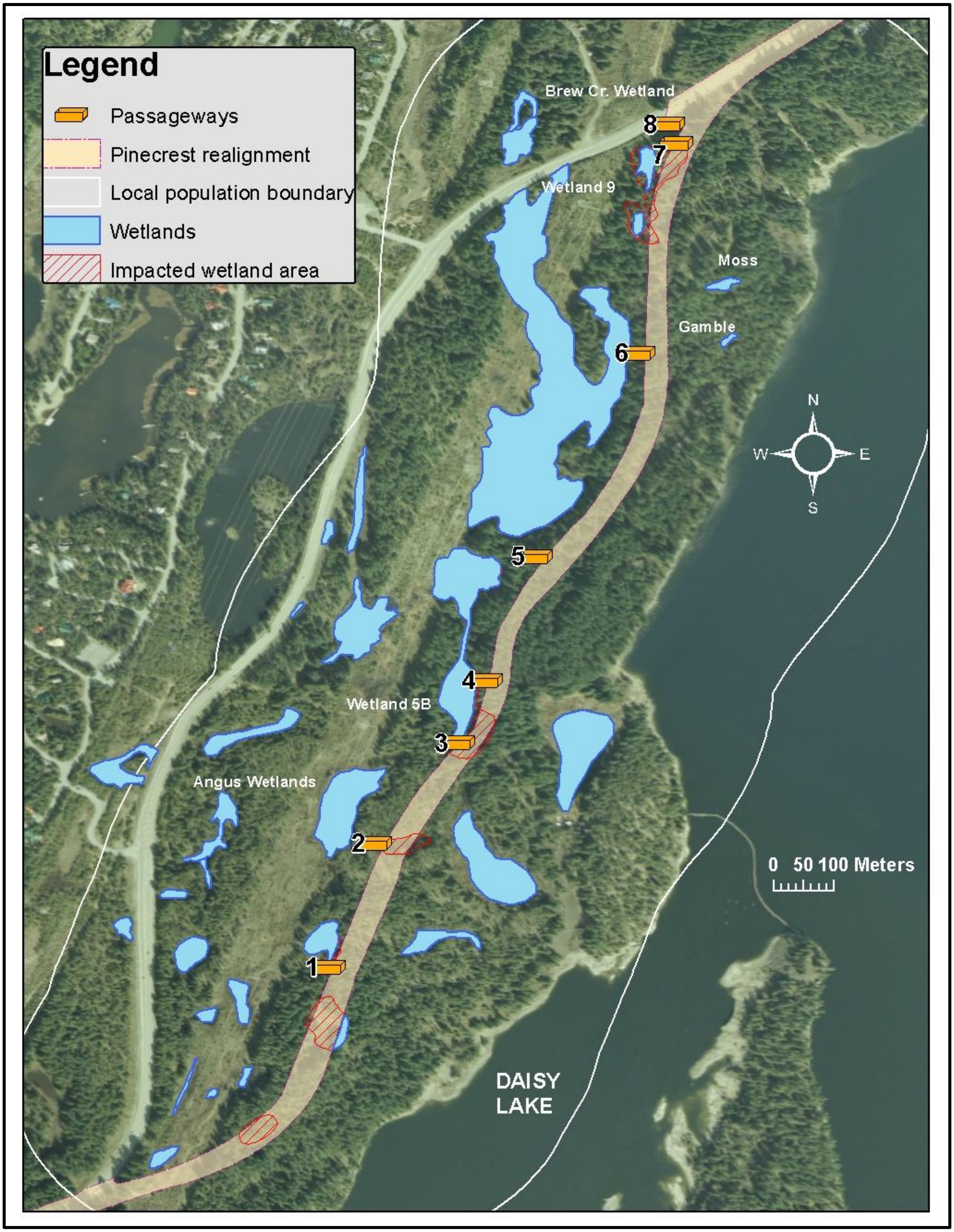


**Figure 1.** Pinecrest study area, including the highway realignment route, passageway locations, pre- and post-impact wetland boundaries, and the red-legged frog local population boundary.

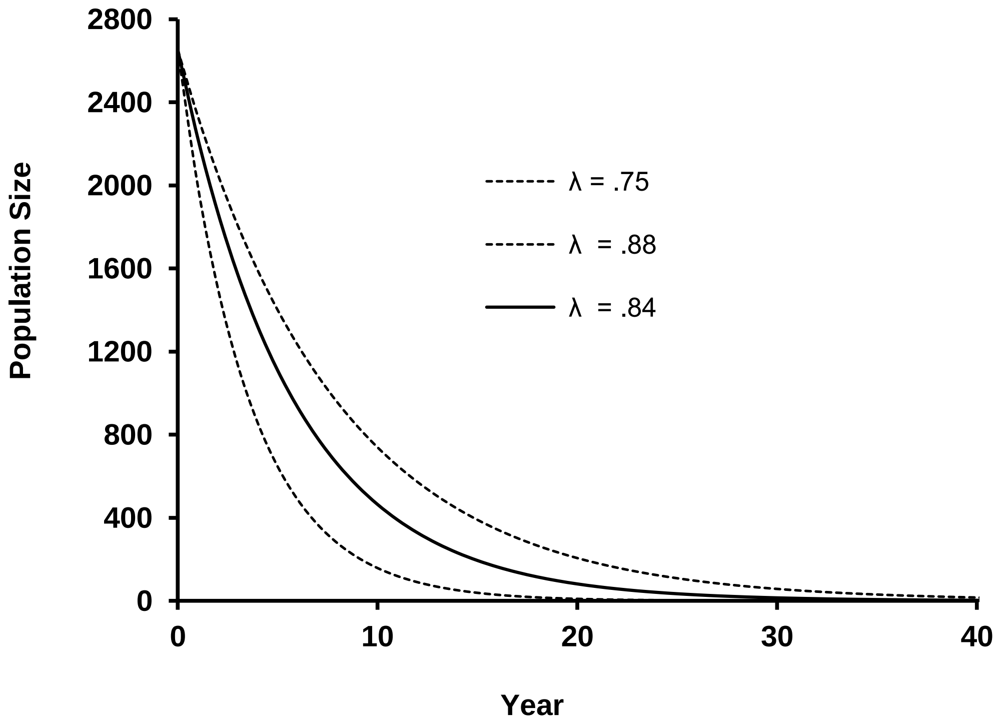


**Figure 2.** Projected local population trajectory for red-legged frogs at Pinecrest, using the estimated population size in 2010 as Year 0, and projecting forward under the range of estimated population growth rates (λ).

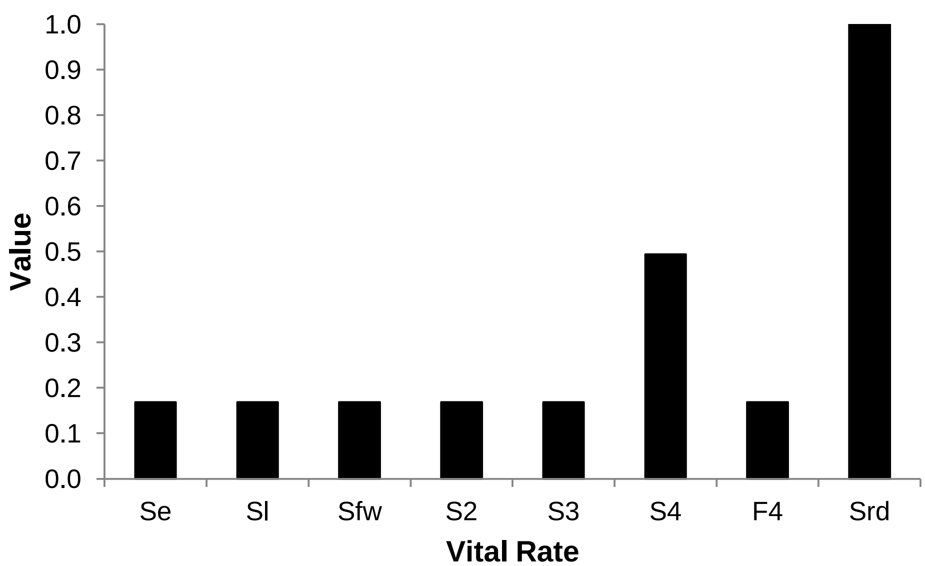


**Figure 3.** Estimated elasticity and sensitivity values of red-legged frog vital rates under Leslie Matrix Models (See Table 1 for descriptions).

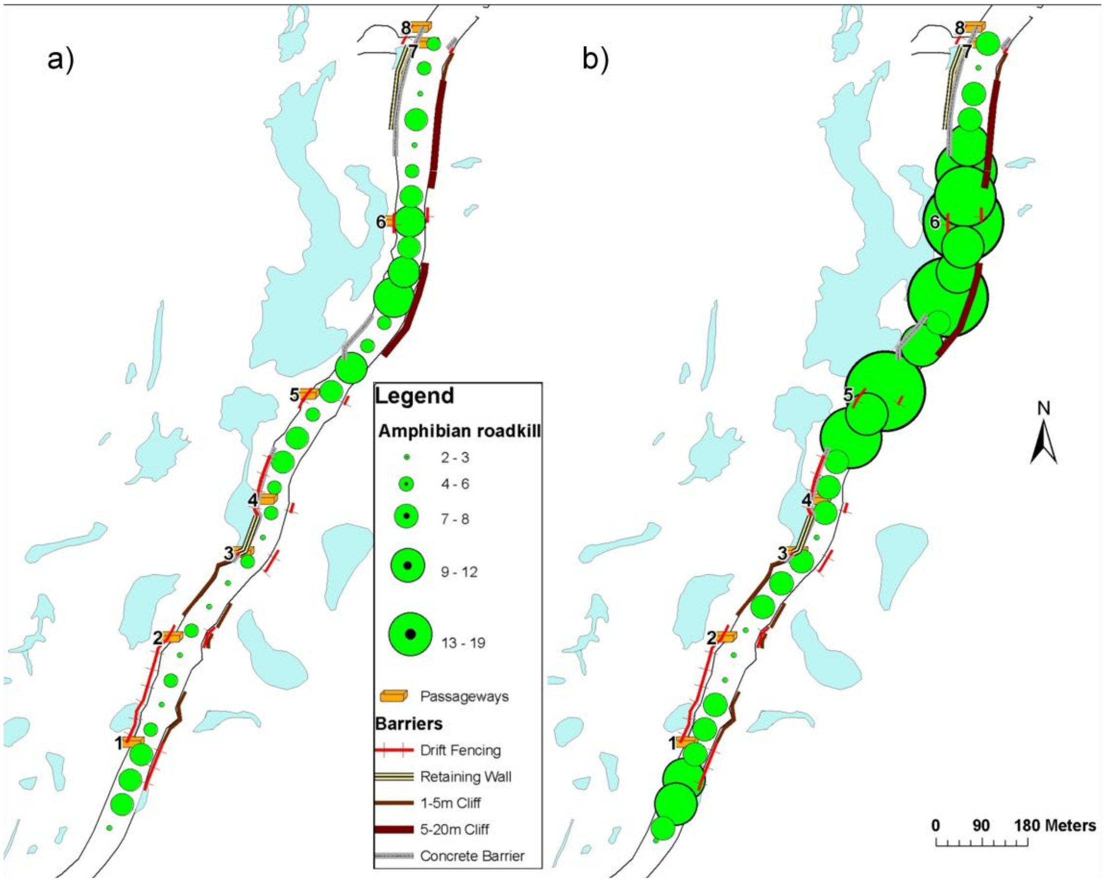


**Figure 4.** Amphibian carcass totals by 50 m highway segments at Pinecrest in a) 2009 and b) 2010. Differences in totals are in part due to more surveys conducted in 2010 (N=37) vs. 2009 (N=21).

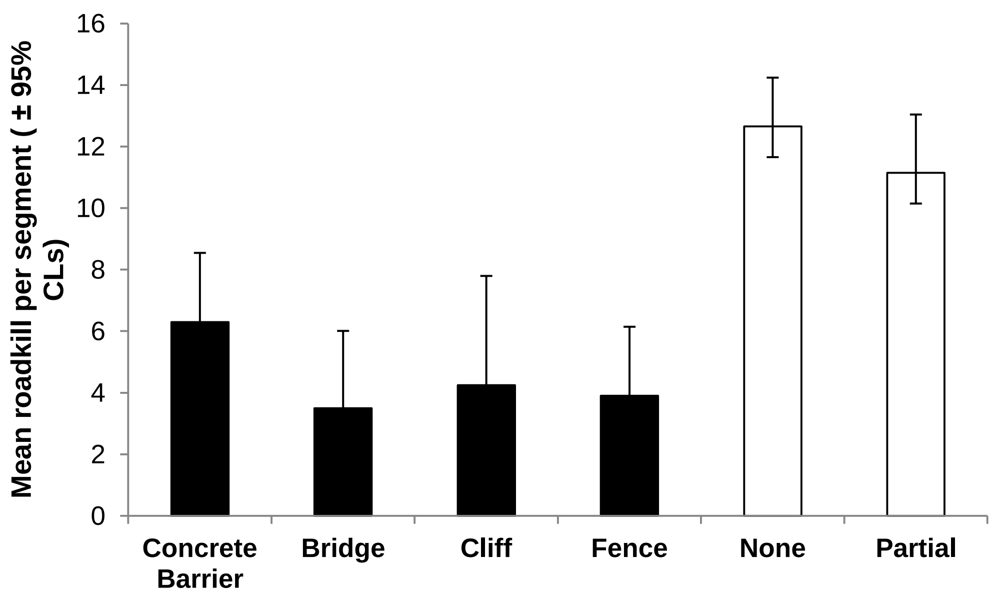


**Figure 5.** Mean number of roadkill per segment (±SE), compared between different types of natural and constructed barriers.

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# Appendix I. Best management practices for minimizing highway impacts on amphibians and reptiles

The following best management practices (BMPs) are based on the findings of this study and associated literature on this topic. These BMPs are intended to supplement existing best practices (Puky 2003, MWLAP 2004).

## Assessment of values

- Utilize sufficient expertise, methods, and survey effort to sufficiently assess all related environmental values that occur within the impacted area (section 6.6).
- Conduct a coarse-scale assessment of the diversity of amphibians that occur within the study area using methods such as egg mass surveys and larval trapping.
- Quantify amounts and distribution of all habitat types that local amphibians use during different portions of their life histories, including breeding and ephemeral/riparian habitats.
- Conduct population-level assessments of "target species" (i.e. species at risk or species particularly sensitive to project impacts). Population-level assessments should utilize sufficient efforts and methods in the appropriate seasons to assess population size within the study area, as well as densities within different habitat types. Potential methods include systematic time-constrained searches, capture-recapture surveys, and habitat suitability modeling.

## Assessment of potential impacts

- Assess project impacts on all habitat types, incorporating factors such as their rarity in determination of significance of impacts. For instance, loss of rare but important habitat may have a disproportionately negative effect, such as the case with ephemeral habitat loss at Pinecrest (section 6.2).
- Assess impacts of project on both habitat quantity and quality. Impacts to habitat quantity include the direct effects of habitat loss, whereas impacts to habitat quality include hydrological changes and other indirect effects.
- Predict the impacts of road mortality on population viability of the target species. Potential methods include prediction of road mortality probabilities based on established models (Hels and Buchwald 2001). The significance of mortality should be assessed according to the size of the local population (see above).

### Mitigation 1: amphibian and reptile salvage

- Do not consider salvage and translocation of amphibians and reptiles as measures to mitigate the effects of habitat loss and mortality risk from construction activities. While salvage may be the only option to avoid direct mortality, the survival of translocated individuals is questionable, particularly when they are moved into occupied habitat (sections 6.2).

### Mitigation 2: passageway locations

- Incorporate information on site- and landscape-scale habitat distribution, local topography, natural and artificial barriers, and movement rates of target species into selection of passageway locations (section 6.3). Maximize the frequency of passageway locations (i.e. every 50 m; MWLAP 2004), to hedge against uncertainty regarding the location of optimal passageway sites.
- In cases where detailed knowledge on the movement behaviour of the target species does not exist (particularly within the study area), conduct supplementary assessments to gather this knowledge. Potential methods include radiotelemetry tracking studies and modeling of potential movement routes.
- Select passage locations, and conduct clearing and grubbing works, such that the approaches to passageways are well vegetated, thereby providing habitat connectivity and protective cover at passage entrances (section 6.3).

### Mitigation 3: passageway design

- Design passageways to minimize differences between interior conditions and adjacent habitats (MWLAP 2004).
- Maximize the diameter-to-length ratio of passageways to maximize light penetration and air flow (section 6.3). Typically the easiest way to accomplish this goal is to widen the diameter of culverts (MWLAP 2004), given that the width of the road and associated culvert length are usually fixed.
- Where possible, install metal grates at the top of passageways to facilitate the input of light, moisture, and air flow (Jackson and Tyning 1989, Meinig 1989). This may be particularly important in cases where large tunnels are not feasible.
- Develop and construct engineering solutions that facilitate input of moisture into passageways, while avoiding flooding (Jackson and Griffin 2000). One option that may warrant further investigation is installation of a passive gravity-fed “trickle line” within culvert substrate, that is fed by a nearby stream or rainwater collection system.

### Mitigation 4: fencing design

- Design fencing systems that *both exclude* animals from the highway to prevent mortality, *and direct animals* towards passageways to facilitate use (section 6.4).
- In order to exclude animals from the highway, design fencing systems that provide coverage of all areas where the road intersects amphibian habitat and potential movement routes. In all circumstances, fencing should be of sufficient length (at least 200 m) to prevent circumvention by migrating individuals.
- Where possible, the ends of fencing should be connected to natural or artificial barriers to prevent amphibians from circumventing fencing at these locations. Where this is not possible, the ends of fencing should be turned back to redirect animals (Puky 2003).
- Design fencing-culvert connections such that attachments are secure and durable, in order to prevent creation of holes or gaps that could act as escapement routes (section 6.3).
- Ensure that fencing is vertically aligned at fencing-culvert connections, as angled fencing at these locations can facilitate climbing by amphibians (section 6.3).
- Design the tops of fences so they curve away from the highway, in order to prevent climbing over fences (Puky 2003). This is particularly important for salamanders and Pacific treefrogs (*Pseudacris regilla*), which can easily climb most vertical surfaces.
- Fences should also be at least 50 cm high (MWLAP 2004), so that individuals are unable to jump on top of, or over fences (sections 6.3 and 6.4).
- In order to direct animals towards passageways to facilitate use, fencing should be angled towards passageways. Where possible, install fencing using a "zig-zag" pattern (Jackson and Tyning 1989), which directs animals towards passageways at all points of contact, while simultaneously excluding animals along its entire length.

### Fencing construction and maintenance

- Construct fencing of durable materials, to limit the occurrence of holes or gaps that could allow amphibian escapement. The original fencing constructed of wire mesh at Pinecrest was highly susceptible to tearing and collapse (Malt 2010c, b), and therefore is not recommended as a viable fencing option. The new fencing at Pinecrest has proven to be more durable, but its' overall effectiveness is still not unknown (See Appendix II for a detailed description of materials and methods).
- Bury fencing at least 6-10 cm (Puky 2003, MWLAP 2004), and backfill with local material to ensure that no gaps exist underneath. Measures should also be taken to limit the erosion of material underneath fences, which can create opportunities for escapement.
- Conduct regular inspections to identify damaged sections of fencing that require repair. This should be done initially in late winter/early spring before the onset of breeding activity, in order to assess damage associated with winter storms and snow removal. After repairs have been completed, weekly or biweekly inspections should be done

throughout the spring and summer, with the appropriate maintenance conducted concurrently to ensure the continued effectiveness of the fencing system.

# Appendix II. Design details of amphibian passageways and fencing installed at Pinecrest

## Description of improved fencing installed in 2010

The following is a description of the new fencing design installed along key areas of the Pinecrest alignment in 2010 (see figure on the following page). The long-term goal of this fencing "trial" is to develop a MOT-endorsed design standard for amphibian fencing. While clearly an improvement, this new design has yet to be fully tested, and therefore the specifications below are included as interim recommendations only. Details were provided by Larry Paradis and Damian Lustic (Miller Capilano Maintenance Corp.).

- Fencing is constructed of aquaculture (oyster farm) netting secured to rebar posts. Netting is black polyethylene, 24" in width, with a ¼" mesh size. Rebar is 15mm in diameter, and 42" in length.
- The top 6" of rebar posts are bent at 60° away from the highway (as per Mitigation 4 above).
- Rebar posts are buried to depth of approx. 18", and with an above-ground height of approx. 18".
- Netting is secured to the outside bend of the rebar with four UV-stabilized black zip ties (40lb sheer strength), buried into the soil approximately 3" in depth, and draped over the rebar bend by approximately 3".
- Netting is attached to roadside concrete barriers with construction adhesive, with an approx. 24" loop to allow for barrier movement.

<u>Note</u>: In 2012, rebar crossbars were welded onto fences in the field, to connect each rebar post to each adjacent post. Crossbars were connected immediately below the bent portion of each post. This modification acted to approve the stability of fencing, and provided additional support for the aquaculture netting to withstand snow loads etc, thereby reducing maintenance costs.

## Design specifications of amphibian/wildlife passages at Pinecrest.

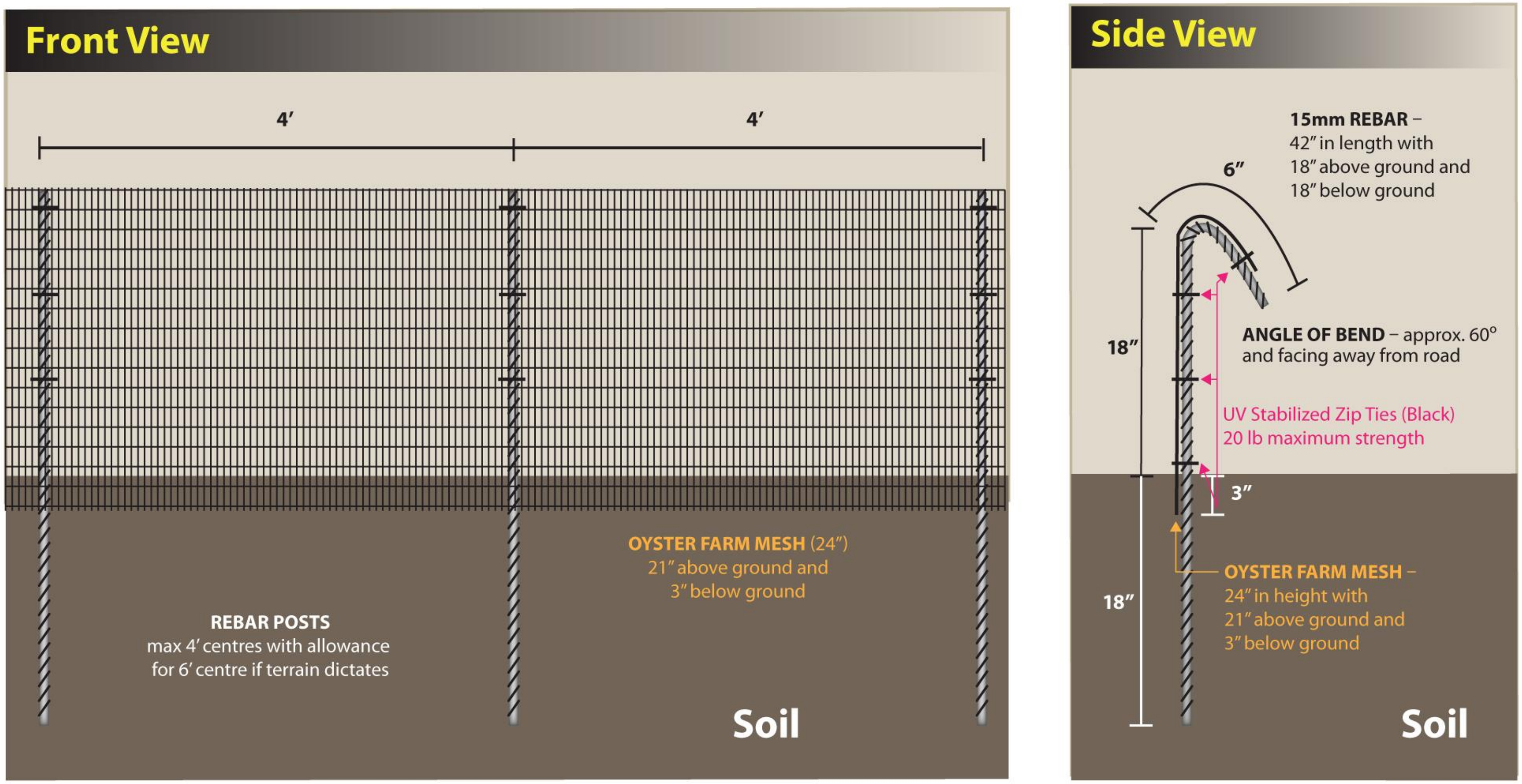
The following table provides designed details of the 8 wildlife culverts installed at Pinecrest.

| Passage | Diameter | Length | Diameter : Length Ratio | Construction Material[1] |
|---|---|---|---|---|
| **1** | 1m | 25m | 0.04 | PVC |
| **2** | 1m | 29m | 0.03 | PVC |
| **3** | 3m | 32m | 0.09 | CSP |
| **4** | 1m | 25m | 0.04 | PVC |
| **5** | 2m | 37m | 0.05 | Concrete |
| **6** | 2m | 34m | 0.06 | Concrete |
| **7** | 1m | 37m | 0.03 | PVC |
| **8** | 2m | 21m | 0.10 | Concrete |

[1] PVC = Polyvinyl chloride, CSP = corrugated steel pipe.

## Amphibian Fencing – Pinecrest

DRAFT ONLY




**Conceptual design for discussion purposes only.**



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