Analyses of the Results of Surveys for Hochstetter's Frogs Undertaken in 2024 to Assess the Impacts of Stream Flow Reductions Associated with the Wharekirauponga Underground Mine

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EXECUTIVE SUMMARY

The results of hydrological modelling indicate that there may be transient reductions of between 2% and 13% in stream flows immediately above, and downstream of, the Wharekirauponga Underground Mine during dry periods caused by dewatering associated with the mine. These stream flow reductions could affect semi-aquatic Hochstetter's frog populations in affected areas of the Wharekirauponga catchment. This report presents analyses of results from preliminary surveys undertaken to develop a Before-After Control-Impact (BACI) programme to monitor potential effects of stream flow reductions on Hochstetter's frog populations.

A variety of modelling methods were used to compare Hochstetter's frog populations along sections of streams in a treatment area likely to be affected by dewatering from the underground mine and a nearby non-treatment area not affected by dewatering. Generalised linear models (GLMs) and generalised linear mixed-effects models (GLMMs) were used to compare frog counts from replicate surveys of transects in the two areas and N-mixture modelling was used to obtain frog abundance estimates for the two areas from the replicate counts. Frog counts were significantly ($p < 0.001$) higher along transects in the non-treatment area: 1.09 vs. 0.23 frogs per transect search. In GLM and GLMM models frog counts in the treatment area increased with stream wet-width ($p < 0.01$) but declined in the non-treatment area ($p < 0.001$). Frog counts in the treatment area were not affected by elevation, whereas frog counts in the non-treatment area were significantly ($p < 0.001$) higher in the mid-elevation class (≥400<500 m a.s.l.)

Abundance estimates from N-mixture models were higher for transects in the non-treatment area: 2.88 vs. 0.91 frogs per transect using classical statistics and 3.2 vs. 1.05 frogs per transect using Bayesian statistics method. Scaling up N-mixture modelling abundance estimates for the 20 m-long transects in the treatment area to the 12.1 km of streams in the area's sampling frame, provides estimates of the total Hochstetter's frog population in the treatment area of 549 (CI95%: 238 – 1,270) and 637 (CI95%: 271 – 1,597) from classical and Bayesian analyses respectively.

During analyses of the survey results problems were apparent in both the survey design and the survey method, which compromise their use as pre-treatment samples in a BACI monitoring programme. Consequently, the surveys described in this report should be considered as pilot surveys used to develop a robust BACI monitoring programme in future. In BACI monitoring programmes, the values of potential covariates for treatment and non-treatment areas should be similar. However, in the surveys described in this report, important characteristics of transects in the two areas are very different. The elevation ranges for treatment transects was lower than for non-treatment transects (120 – 320 m a.sl. vs. 140 – 640 m a.sl.). Treatment transects were in either manuka/kanuka scrub or kauri forest, whereas all non-treatment transects were in lowland podocarp-hardwood forest. The stream substrates in treatment transects included more vegetation (42% vs. 10%) and less boulder and bedrock (26% vs. 54%) than non-treatment transects. This lack of overlap and imbalance in covariates for the treatment and non-treatment areas compromises use of the survey results as pre-treatment samples for a BACI monitoring programme. A robust BACI design should have transects for the non-treatment sample in sections of streams within the Wharekirauponga catchment that will not be affected by potential dewatering but are at similar elevations (100 – 400 m a.s.l.) and with similar NZLRI vegetation types (manuka/kanuka or kauri forest) to

streams in the area likely to be affected by dewatering. The smaller number of transects surveyed in the non-treatment area (12 vs. 40) also compromised comparisons between the treatment and non-treatment areas.

Another problem with the surveys is that the two variables survey duration and the numbers of refuges searched during each search of a transect are the dominant explanatory variables in models of frog counts. However, their role as explanatory variables is problematic, because they are both measures of search effort during a transect search. Their values varied between replicate searches of individual transects, with the number of frogs found during a transect search increasing with increasing search effort. The survey method should be redesigned to make the number of refuges searched on a transect a characteristic of the transect, with the same or similar numbers of refuges searched during each replicate survey of a transect based on the number of refuges available along the transect. This makes it clear that the number of refuges searched is a transect-level variable likely to be associated with frog abundance (i.e. the actual numbers of frogs on a transect), not a survey-level variable measuring search effort and presumably detection probability during individual surveys.

Finally, during each replicate survey of a transect, considerable time was spent collecting information on transect characteristics unlikely to change between surveys. This information has not proved useful in predicting frog abundance, largely because so few frogs found during surveys. Investing less time in collecting transect information would allow more transects to be surveyed, which will provide a better basis for assessing the effects of dewatering.

Experience gained from the surveys described in this report provides the basis for designing a robust BACI programme to monitor potential effects of stream flow reductions on Hochstetter's frog populations in the Wharekirauponga catchment.

INTRODUCTION

The results of hydrological modelling indicate that there may be reductions in stream flows within the Wharekirauponga catchment caused by dewatering associated with the Wharekirauponga Underground Mine (GHD, 2024; van Winkel, 2024). Model results predict that the 7-day mean annual low flow in streams immediately above, and downstream of, the Wharekirauponga Underground Mine could be reduced for short periods during prolonged dry spells by between 2 and 13%. Stream flow reductions and associated effects on instream habitat could affect semi-aquatic Hochstetter's frogs (*Leiopelma hochstetteri*) that live along the edges of forested streams in the Wharekirauponga catchment. This report presents analyses of results from preliminary surveys undertaken to develop a Before-After Control-Impact (BACI) programme to monitor potential effects of stream flow reductions on Hochstetter's frog populations.

Survey Based Studies of Hochstetter's Frog Populations

A variety of methods have been used to survey for Hochstetter's frog distribution and monitor their abundance. Most studies have been restricted to count indices from simple counts along stream transects (Baber, Moulton, Smuts-Kennedy, Gemmell, & Crossland, 2006; Bradfield, 2005; Green & Tessier, 1990; Longson, Brejaart, Baber, & Babbitt, 2017; Musset, 2005; Nájera-Hillman, 2009; Nájera-Hillman, King, Baird, & Breen., 2009; Sadowski, 2016; Whitaker & Alspach, 1999). Count indices with double-observer sampling have also been undertaken (Herbert, Melzer, Gilbert, & Jamieson, 2014). Previously, capture-recapture methods with individual identification by toe clipping were used to estimate abundance (Slaven, 1992; Tessier, Slaven, & Green, 1991) but are no longer used because of ethical concerns. More recently, capture-recapture studies have been attempted with individuals identified by their size and location (Johnson, 2022; Puig, 2009). Replicate counts along stream transect have been used to allow site-occupancy modelling (Crossland, MacKenzie, & Holzapfel, 2005; Nájera-Hillman, 2009; Nájera-Hillman, Alfaro, et al., 2009; Puig, 2009) and N-mixture modelling (Johnson, 2022; Puig, 2009).

Two studies (Johnson, 2022; Puig, 2009) undertook comparisons of the different monitoring techniques, comparing results for capture-recapture methods with individual identification by size and location, N-mixture modelling (A. Royle, Nichols, & Kery, 2005), site-occupancy modelling (MacKenzie et al., 2002) and Poisson regression of single-count data. Puig (2009) concluded that using simple counts is not a good method for long-term monitoring of Hochstetter's frogs. Occupancy modelling from replicate counts is a better method than using single counts, but requires the same field effort as N-mixture, which also provides abundance estimates. Johnson (2022) concluded: *"The probability of occupancy estimates generated in the single-year site occupancy model are good indicators of distribution, but when estimating abundance is the key objective, capture-mark-recapture and N-mixture models provide more reliable estimates than derived estimates of N from occupancy probabilities"*.

In the BACI programme for monitoring potential effects of stream flow reductions on Hochstetter's frog populations in the Wharekirauponga catchment, N-mixture modelling was used to obtain frog abundance estimates from replicate counts of frogs found along transects.

METHODS

Survey Method

Details of the survey method are provided in van Winkel (2024). Briefly, the surveys entailed daytime searches for Hochstetter's frogs in their refuges along 20 m sections of streams. During surveys, the searchers moved slowly upstream along the 20 m long transects searching for frogs beneath rocks, debris, in rock crevices, in leaf litter packs, and debris dams. When a frog was found its age-class was estimated and recorded along with its location and characteristics of its refuge. Environmental variables recorded during transect surveys were: air and water temperatures, relative humidity (RH) and general weather conditions. Stream characteristics (width, depth and substrate) were recorded every 2 m along each transect. Canopy cover and dominant canopy species were also recorded for each transect. Ideally each transect was searched three times, with at least 2 days between replicate searches to ensure independence.

Survey Design

The surveys described in this report are preliminary before-impact surveys undertaken as part of a Before-After Control-Impact (BACI) monitoring programme to assess the effect of stream flow reductions on Hochstetter's frogs (van Winkle, 2023). Separate sampling frames were selected along streams or drainage channels in treatment and non-treatment (i.e. control) areas (Figure 1). Within each of these two sampling frames, 40 transects were chosen randomly to be surveyed for Hochstetter's frogs. The treatment sampling frame (i.e. Edmonds Affected Area) extends along 12.1 km of stream and drainage channels in the lower reaches of Edmonds stream catchment above, and downstream of, the proposed underground mine. The control or non-treatment sampling frame extends along 42.1 km of streams outside of the affected area spread among three adjacent stream catchments: Edmonds, Marototo and Waiharakeke.

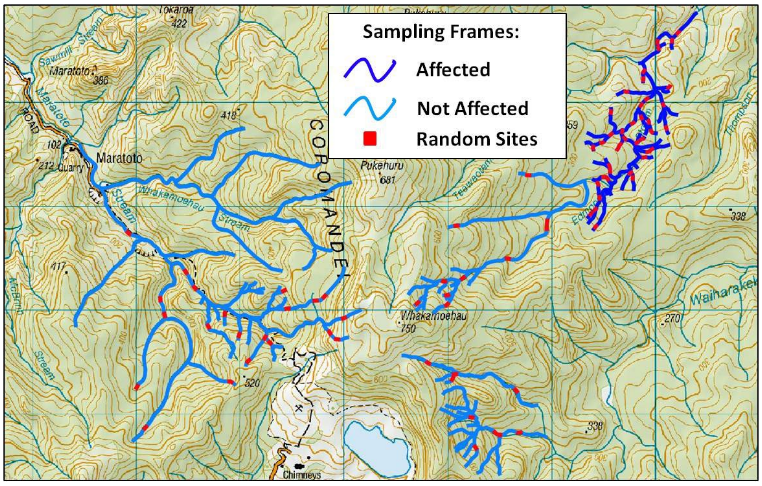

Figure 1. Sampling frames along streams or drainage channels in the treatment (affected) and control (not affected) areas, showing 40 randomly chosen sites for transects in each of the areas.

Numeric Methods

Modelling Methods

Three modelling methods were used to analyse results from the surveys: generalised linear models (GLMs), generalised linear mixed-effects models (GLMMs) and N-mixture modelling (Kery & Royle, 2016; Madsen & Royle, 2023; J. A. Royle, 2004). GLMs were used to investigate the effects of likely transect-level explanatory variables on frog counts during all surveys along transects, with results from surveys in the treatment and non-treatment areas modelled separately. GLMMs with Poisson error distributions were used to investigate the effects of survey-level explanatory variables on frog counts during surveys, with the number of frogs found during each transect survey as the dependent variable and transect identity as the random effect or grouping variable. N-mixture modelling was used to obtain estimates of the numbers of frogs present on transects from counts of frogs found during replicate surveys of transects in the two areas. N-mixture modelling was undertaken both without covariates and with covariates identified in GLM and GLMM analyses.

Fitting Probability Distributions to Frog Counts

Before modelling the data from the surveys, it was necessary to fit probability distributions to the frog count data by selecting the best probability distributions describing the counts. Initially, the goodness-of-fit of distributions was evaluated graphically by overlaying the most likely probability density function curves onto histograms of the counts. This was followed by undertaking goodness-of-fit tests using the *R*-function *goodfit* in *R*-package *vcd*. The variance to mean ratio of count data was used to identify overdispersion in the count data. Where variance to mean ratios are close to one, the Poisson probability distribution provides a good fit to the data. Variance to mean ratios approaching, or greater than, two are evidence of overdispersion and the need to use negative binomial distributions instead of Poisson distributions to model count data.

Box-and-whisker Plots

Box-and-whisker plots used in this report provide graphical descriptions of the locality, spread and skewness of groups of numerical data (Chambers, Cleveland, Kleiner, & Tukey, 1983). The central dark lines in boxes are the median values for each of the groups, while the boxes encompass the middle two quartile values for each of the groups. The whiskers extend beyond the boxes to the most extreme value no more than the interquartile range from the boxes. Values outside the interquartile range from the boxes are plotted separately beyond the whiskers. Notches in the boxes extend over ±1.58 x (Inter Quartile Range/sqrt(n)) around the median and provide approximate 95% confidence intervals around the medians. Where notches in boxes do not overlap, median values in the groups are likely to be significantly different. In this report, box widths in box-and-whisker plots are proportional to the square-roots of the number of observations in the groups.

Explanatory Variables

Elevation

Elevations of transects were obtained using the QGIS (QGIS.org, 2023) geographic information system (GIS) with the QGIS Python Plugin Nearest neighbour join (NNJoin) using nearest neighbour relationships to assign elevations at 20 m intervals in the NZ Contours (Topo. 1:50k) layer[1] to transect locations. For analyses, transect elevations were initially placed into 100 m-wide classes: ≥100<200; ≥200<300; ≥300<400; ≥400<300; ≥500<600; ≥600<700 m a.s.l..

Vegetation Types

The vegetation types along transects were obtained using the geoprocessing intersection tool in QGIS to assign vegetation types from two spatial databases of vegetation types to transect locations. The two spatial databases of vegetation types were: Land Cover Database version 5.0 (LCDB v5.0)[2] and NZ Land Resource Inventory Version 3 (NZLRI)[3]. Information on vegetation types in the NZLRI database was derived from stereo aerial photograph interpretation, with field verification and measurement, undertaken between 1973 and 1983 (Blaschke, Hunter, Eyles, & Van Berkel, 1981; Hunter & Blaschke, 1986). LCDB v5.0 is the most recent spatial database for vegetation types in the Coromandel, last corrected during summer 2018–19. Satellite imagery is the primary data source for classifying vegetation types in LCDB (Thompson, Grüner, & Gapare, 2003). Although information on vegetation types in the NZLRI spatial database was collected 50 years ago, in 1983, it provides more detail than the recent LCDB v5.0 database. In the fifty years since areas were classified using NZLRI (1983), natural regeneration of *Manuka, kanuka scrub* will have converted most of these scrub areas to regenerating forest dominated by early successional species, with the successional process faster at lower altitudes. In the absence of disturbance, vegetation types in areas classified as *Kauri forest* or *Lowland podocarp-hardwood forest* in NZLRI during 1983 are unlikely to have changed significantly.

Canopy Species

Tables of canopy species present on all transects (Appendix1) were compiled from the data. The explanatory value of individual canopy species was assessed using an exact test of success in a Bernoulli experiment (*binom.test* in R) testing the null hypothesis that the probability of frog occurrence in transect with the canopy species was the same as for all transects. Ordination of the canopy species by correspondence analysis (Legendre & Legendre, 2012), implemented with the R-functions *cca* in the R-library *vegan*, was used to identify distinct habitat types on transect characterised by associations of canopy species.

Substrate

To describe stream-bed substrates along transects, substrates were allocated to one of eight substrate types: vegetation, silt, sand, fine gravel, gravel, cobbles, boulders and bedrock. During each survey, substrate types at 5 equidistant points on cross sections of the stream, were recorded every 2 m along the transect. This provided observations of substrate types at 55

[1] https://data.linz.govt.nz/layer/50768-nz-contours-topo-150k/
[2] https://lris.scinfo.org.nz/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand
[3] https://lris.scinfo.org.nz/layer/48055-nzlri-vegetation/

points along a transect during each survey. For transects with three surveys, there were 165 observations of stream bed substrate types. To investigate the influence of substrate type on frog abundance, the proportion of substrate observations belonging to each of the eight substrate types was calculated for each transect.

Stream-margin Width

Because stream wet-width and stream bank-width are collinear, bank-width was replaced by stream-margin width, calculated as the difference between stream wet-width and stream bank-width.

Categoric Variables

For modelling purposes, to avoid the assumption that categoric variables had monotonic relationships with frog counts, categoric variables (elevation, vegetation types, and percentage canopy cover) were converted to unordered factors.

Generalised Linear Models

Generalised linear models (GLMs) were used to investigate the effects of likely transect-level explanatory variables on frog counts during all surveys along transects, with results from surveys of transects in the two areas (Edmonds Affected Area and Outside the Affected) modelled separately. Because of the low numbers of frogs encountered during surveys different age classes and sex were disregarded and only the total numbers of frogs found during surveys of each transect were used as the dependent variable in the models. To adjust for the different numbers of surveys of transects, the logarithm of the number of surveys of each transect was included as an offset term. Potential transect-level explanatory variables used in the models were: elevation, vegetation type, percentage canopy cover, stream wet width, stream margin width (i.e. the difference between stream wet-width and bank-width), proportions of each of the substrate types 1–8, mean numbers of refuges searched during surveys of each transect and mean durations of surveys of each transect.

Separate GLMs were undertaken with Poisson, quasi-Poisson, and negative binomial error distributions. GLMs were implemented in R using R-functions *glm* with a logarithmic link function for Poisson and quasi-Poisson error distributions and *glm.nb* (R-library *MASS*) for the negative binomial error distribution. Dispersion tests, implemented with the R-function *dispersiontest* (R-library *AER*), were used to detect overdispersion or clustering in the Poisson models. Dispersion tests provide estimates of the dispersion parameter and test the null hypothesis of equidispersion in Poisson GLMs against the alternative of overdispersion and, or underdispersion. Overdispersion in Poisson models occurs when variation is higher than expected, usually as a result of important explanatory variables being missing from the model. Initially, separate GLMs were undertaken for each of the likely transect-level explanatory variables. All explanatory variables with significant effects on frog counts in GLMs with only one explanatory variable were than included in a single multi-way model. Starting from this full model with several explanatory variable, stepwise regression with backward elimination (Cameron & Trivedi, 1998) was used to select the best generalised linear model to fit the observed data.

Variance-inflation factors (VIF) (O'Brien, 2007) were estimated for multi-way GLMs using R-function *vif* (R-library *car*) to provide a measure of collinearity, or multicollinearity in the models. Collinearity and multicollinearity occur when explanatory variables in regression models are correlated with each other.

Generalised Linear Mixed Effect Models

Generalised linear mixed-effects models (GLMMs) with Poisson error distributions were used to investigate the effects of likely survey-level explanatory variables on frog counts during surveys, with the total number of frogs found during each survey as the dependent variable and transect as the random effect or grouping variable. Potential search-level explanatory variables are: search-team, time of day, survey duration (minutes), number of refuges searched during each survey, temperature (°C) and relative humidity (RH).

For the GLMMs, all numeric variables were scaled. Scaling entails first centering values by subtracting their means and then dividing the centred values by their standard deviation. Initially, separate GLMMs were undertaken for each of the likely survey-level explanatory variables. Survey-level explanatory variables with significant effects on frog counts in GLMMs with only one explanatory variable were than included in a single multi-way model together with significant transect-level explanatory variables from the GLMs. Starting from this full model containing several explanatory variable, stepwise regression with backward elimination (Cameron & Trivedi, 1998) was used to select the best mixed effects model to fit the observed data. GLMMs with Poisson error distributions were implemented using the R-function *glmer* (R-library *lme4*). The R-function *isSingular* (R-library lme4) was used to evaluate whether fitted mixed model were singular, with parameters on the boundary of the feasible parameter space. Dispersion was estimated using *overdisp_fun* in glmm_funs.R.

N-mixture Modelling

N-mixture modelling (Kery & Royle, 2016; Madsen & Royle, 2023; J. A. Royle, 2004) was used to obtain estimates of the numbers of frogs present on transects from counts of frogs found during replicate surveys of transects in the two areas. N-mixture modelling was undertaken with and without covariates. Covariates used in the models were selected based on results from GLM and GLMM analyses. Two computational methods were used for N-mixture modelling: one based on classical inference and the other on Bayesian inference. N-mixture modelling based on classical inference was undertaken with the R-package *unmarked* (Fiske & Chandler, 2011, 2020), which uses maximum likelihood estimation of marginal likelihoods in hierarchical models such as N-mixture models. N-mixture modelling based on Bayesian inference was undertaken using the R-package *Nimble* (P. de Valpine et al., 2023; P de Valpine et al., 2017). In the Bayesian N-mixture models Poisson and binomial distributions for estimating abundance (λ) and detection probability (p) respectively. Bayesian models were run with 1,00,000 iterations and a 25,000 burn-time. Priors for the models were un-informative, with Gamma distribution shape and scale parameters 0.001 and 0.001 for λ, and random values between 0 and 1 for p.

Results

Transect Locations

Transects were surveyed at 34 of the 40 randomly chosen sites in the affected stream sections. The other 6 chosen sites in the affected area were not surveyed because of safety concerns and were replaced by transects at nearby locations: three within the sampling frame for the affected area and three just outside the sampling frame for the affected area (Figure 2). Outside of the affected area, transects were only undertaken at 12 of the 40 randomly chosen sites, with 10 transects along the Marototo Stream and 2 in the headwaters of Waiharakeke Stream (Figure 3).

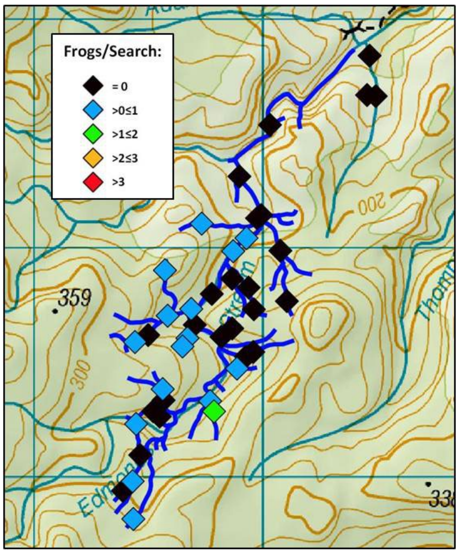

Figure 2. Locations of surveyed transects in Edmonds affected area with the mean numbers of frogs found during surveys of each transect.

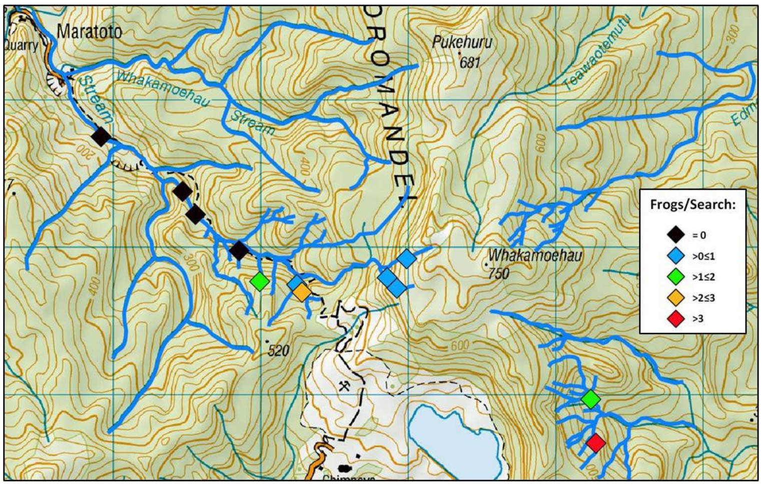

Figure 3. Locations of surveyed transects outside the affected area with the mean numbers of frogs found during surveys of each transect.

Transect Characteristics

Characteristics of surveyed transects in the two areas were very different. Elevations of the 40 transect in Edmond's Affected Area ranged between 120 and 320 m a.s.l., with a mean elevation of 225 (CI95%: 212– 239) m a.s.l. (Table 1 and Figure 4). By comparison elevations of the 12 transects outside of Edmond's Affected Area ranged between 140 and 640 m a.s.l., with a mean of 401 (CI95%: 301–502) m a.s.l.. Results from a GLM showed elevations of transects in the two areas are significantly different ($p < 0.001$).

Table 1. Distribution of transects among 100 m wide elevation classes in the two areas.

Elevation (m a.s.l.)	Edmonds Affected	Outside Affected
<200	8 (20%)	1 (8%)
≥200 <300	29 (73%)	2 (17%)
≥300 <400	3 (8%)	2 (17%)
≥400 <500		4 (33%)
≥500 <600		1 (8%)
≥600 <700		2 (17%)

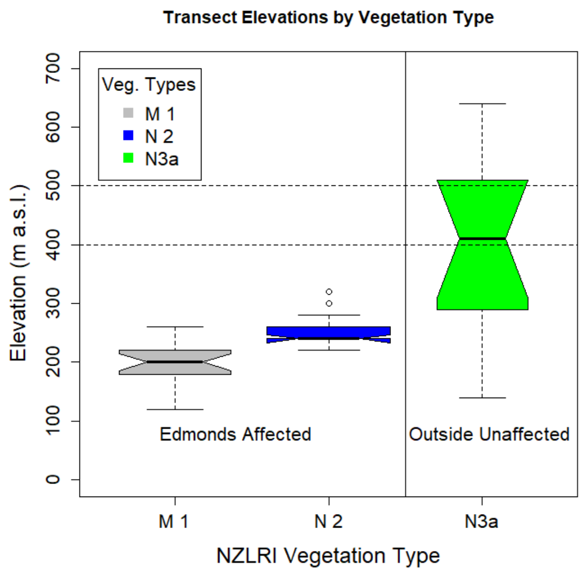

Figure 4. A box-and-whisker plot comparing the elevations and NZLRI vegetation types of transects in the two areas.

All the surveyed transects were located in a single LCDB vegetation type "*Indigenous forest*: *vegetation dominated by indigenous tall forest canopy species*". Consequently, LCDB vegetation type was not used in any analyses. Transects were located in 3 different NZLRI vegetation types (Table 2 and Figure 4):

- *Manuka/kanuka scrub* (M 1)
- *Kauri forest* (N 2)
- *Lowland podocarp-hardwood forest* (N3a)

Table 2. Distribution of transects among NZLRI vegetation types in the two areas.

NZLRI Vegetation Type	Edmonds Affected	Outside Affected
Manuka/kanuka scrub (M 1)	18 (45%	
Kauri forest (N 2)	22 (55%)	
*Lowland podocarp-hardwood (*N3a)		12 (100%
Total	40	12

In the Edmonds Affected area, 18 transects were in manuka/kanuka scrub (M 1) and 22 were in kauri forest. All 12 transects in the unaffected area were in lowland podocarp-hardwood forest (N3a). The proportions of the eight substrate types on stream beds along transects were very different in the two areas (Table 3 and Figure 5), with much higher proportions of vegetation and silt (Substrate Types 1 & 2) found along transects in Edmonds Affected Area than in the unaffected area.

Table 3. Proportions of the eight substrate types on transects in the two areas.

Substrate	Size (mm)	Code	Edmonds Affected	Not Affected
Vegetation		1	42%	10%
Silt (Mud)	<0.06	2	10%	0%
Sand	0.06–2.0	3	3%	1%
Fine gravel	2–8	4	3%	2%
Gravel	8–64	5	4%	7%
Cobble	64–264	6	14%	25%
Boulder	>264	7	23%	41%
Bedrock		8	3%	13%

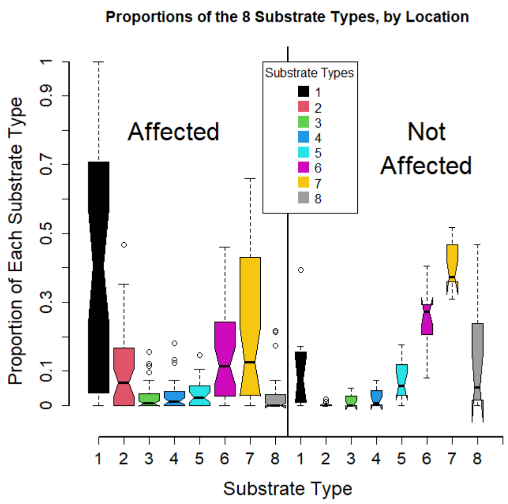

Figure 5. Comparisons of the proportions of the 8 substrate types along transects in the two areas.

Stream widths along transects in the affected area were skewed upwards. Consequently, in a GLM with normal distribution mean stream widths in the two areas were not significantly different, whereas in a Wilcoxon rank sum test the median stream width along transects was significantly (p<0.001) greater outside of the affected area (Figure 6).

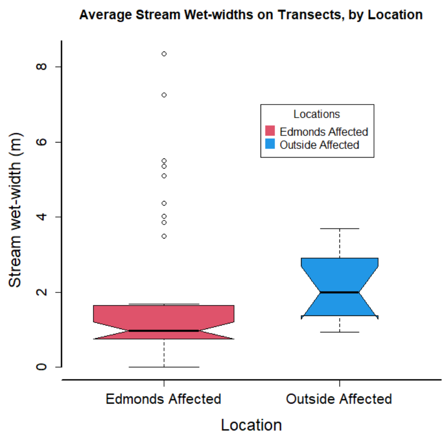

Figure 6. Comparisons of average stream widths along transects in the two areas.

Thirty canopy species were identified on the 52 transects with only 12 canopy species occurring on 6 or more transects (Appendix 1). In an ordination plot (Figure 7) with elevation fitted to the results from ordination of the canopy species using correspondence analysis (Legendre & Legendre, 2012), the three NZLRI vegetation types mapped to clusters in the ordination, supporting the use of NZLRI vegetation types in analyses, instead of clusters from the ordination of canopy species.

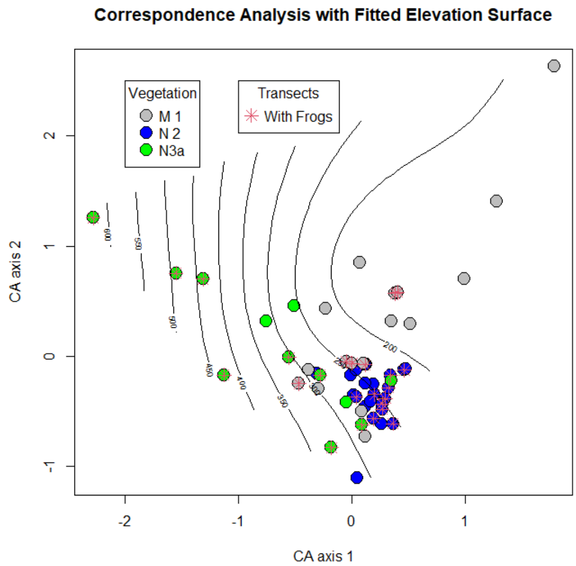

Figure 7. The results of correspondence analysis of canopy species present on transects, mapping transects position on the first two correspondence axes. A smooth surface for elevation was fitted to the ordination diagram using ordisurf in the R-library vegan. Symbol colours identify the three NZLRI vegetation types the transects were in: grey is manuka/kanuka scrub (M 1); blue is kauri forest (N 2); and green is lowland podocarp- hardwood forest (N3a).

Survey Effort

All transect surveys were undertaken during the 59-day period 27 December 2023 – 23 February 2024. In the affected area, 37 of the 40 transects were surveyed 3 times, and 3 transects were surveyed 4 times. Outside of the affected area, 3 of the 12 transects were surveyed twice, 8 were surveyed 3 times and 1 was surveyed 4 times.

Summary of Frog Occurrence and Encounter Rates

Hochstetter's frogs were found along 16 of 40 transects (40%) in the affected area and 8 of 12 transects (67%) outside the affected area. Frog encounter rates were significantly ($p < 0.001$) lower along transects in the affected area than along transects outside the affected area with 0.228 (CI95%: 0.151–0.329) frogs found per transect search in the affected area compared to 1.088 (CI95%: 0.766–1.500) outside the affected area (Table 4).

Table 4. Summary results of the surveys in the two areas, with the total numbers of frogs found and the mean numbers of frogs/search. CI95% are 95% confidence intervals for Poisson means (i.e. λ lambda).

Location	N. of Transects		N.		N. Frogs/Search	
	Total	With Frogs	Searches	Frogs	Mean (λ)	*(CI95%)*
Edmonds Affected	40	16 *(40%)*	123	28	0.228	*(0.151–0.329)*
Outside Affected Area	12	8 *(67%)*	34	37	1.088	*(0.766–1.500)*

The distributions among size classes of frogs found in the two areas (Table 5) were significantly ($p < 0.05$) different, with a lower proportion of juvenile frogs (11% *vs.* 30%) and a higher proportion of mature females (21% *vs.* 0%) found in the affected area.

Table 5. A comparison of the age classes of frogs found during surveys in the two areas.

Location	Juvenile		Sub-adult		Adult		Mature female		Total
Edmonds Affected	3	*(11%)*	7	*(25%)*	12	*(43%)*	6	*(21%)*	28
Outside Affected Area	11	*(30%)*	8	*(22%)*	18	*(49%)*	0	*(0%)*	37

Fitting Probability Distributions to Frog Counts

The Poisson distribution was a good fit for frog counts from surveys on transects in the affected area (Figure 8). In a goodness-of-fit test, the observed distribution of counts was not significantly ($p > 0.1$) different to the expected Poisson distribution with the λ parameter estimated from the observed counts. The variance/mean ratio was 1.07 indicating there was no overdispersion in the count data and the Poisson distribution was suitable for modelling the counts.

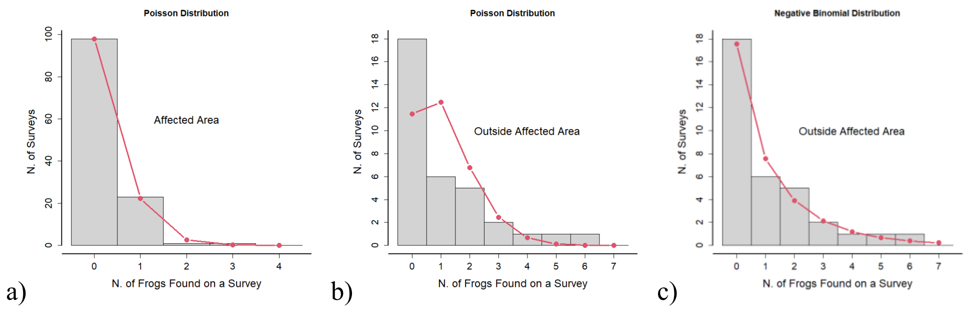

Figure 8. Poisson distributions fitted to frog counts from the two areas (a & b) and the negative binomial distribution fitted to frog counts from outside of the affected area (c).

By contrast the Poisson distribution didn't fit frog counts from surveys on transects outside of the affected area (Figure 8). In a goodness-of-fit test, the observed distribution of counts was significantly ($p < 0.001$) different to the expected distribution for a Poisson distribution. However, in a goodness-of-fit test the observed distribution of counts was not significantly ($p > 0.1$) different to the expected negative binomial distribution with the size and location parameters estimated from the observed counts (Figure 8). A high variance/mean ratio of 2.25 is further evidence that the count data was overdispersed, indicating that a negative binomial distribution is a better fit for the data.

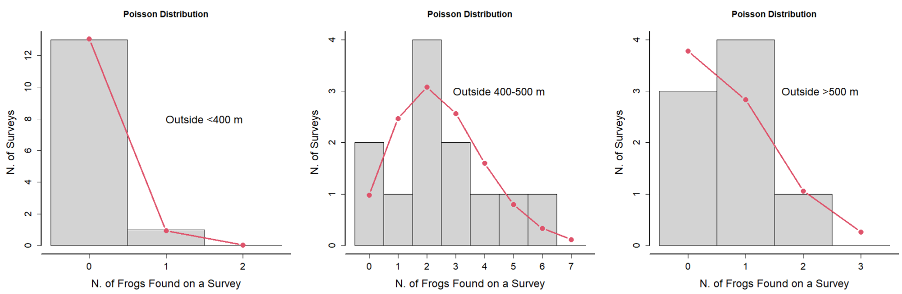

Figure 9. Poisson distributions fitted to frog counts from transects in different elevation classes outside the affected area.

Overdispersion in count data can occur when counts comprise a mixture of counts that fit differently parameterised Poisson distributions. To investigate this, count data from transects in the three elevation classes (<400; ≥400<500; ≥500 m.a.s.l.) were examined separately (Figure 9). Poisson distributions fitted frog counts from all the three elevation classes when they were examined separately. In goodness-of-fit test, the observed distribution of counts from each of the three elevation classes was not significantly ($p > 0.1$) different from expected. The variance/mean ratios were 1, 1.3 and 0.7 for the three elevation classes indicating there was no overdispersion in the count data when counts from the three elevation classes were examined separately.

Frog Location Characteristics

Refuge Types

In a chi-square test of the difference between distributions of frog refuge types where frogs were found along transects in the two areas (Table 6), there was no significant ($p > 0.1$) difference between the areas. However, there were higher proportions of frogs found in leaf pack (29% *vs.* 11%) and under wood or logs (7% *vs.* 0%) and a lower proportion under rocks (57% *vs.* 78%) in the affected area than outside the affected area.

Table 6. A comparison of the refuge types where frogs were found during surveys in the two areas.

Refuge Type	N. & Percentages of Frogs					
	Edmonds		Outside		Combined	
Rock crevice	1	(4%)	2	(5%)	3	(5%)
Leaf pack	8	(29%)	4	(11%)	12	(18%)
In the open	1	(4%)	2	(5%)	3	(5%)
Under rock	16	(57%)	29	(78%)	45	(69%)
Under wood or log	2	(7%)	0	(0%)	2	(3%)
Total	28		37		65	

Stream Wet-width at Frog Capture Sites

Stream wet-width at sites where frogs were found on transects in the affected area (Table 7 and Figure 10) were significantly greater ($p < 0.05$) than at sites where frogs were found on transects outside the affected area (2.6 m *vs.* 1.3 m). Although stream wet-width at sites where frogs were found in the affected area were not significantly ($p > 0.1$) different to mean stream widths along transects in the area, outside of the affected area the streams were significantly narrower ($p < 0.05$) at sites where frogs were found than mean stream widths along transects in the area (1.3 m *vs.* 2.1 m).

Table 7. Comparisons of stream wet-widths at frog capture sites with mean stream wet-widths along transects in the two areas.

Location	Freq	Stream Wet-width (m)		
		Mean	Range	*(CI95%)*
Edmonds Affected Area				
Transects	40	1.85	(0.00–8.36)	*(1.20–2.50)*
Capture Sites	27	2.58	(0.17–12.90)	*(1.34–3.82)*
Outside Affected Area				
Transects	12	2.13	(0.93–3.70)	*(1.55–2.71)*
Capture Sites	37	1.29	(0.16–5.41)	*(0.89–1.69)*

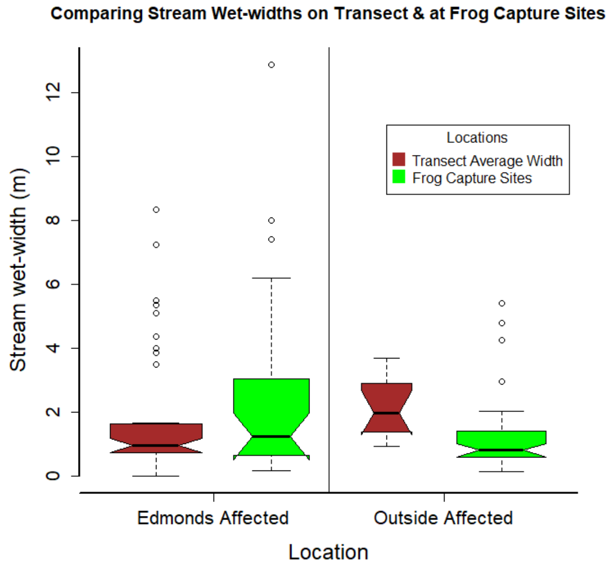

Figure 10. Comparisons of stream widths at frog capture sites with average stream widths along transects in the two areas.

Substrates at Frog Capture Sites

There are only minor differences in proportions of substrates found in cross sections of streams at sites where frogs were found compared to the overall proportions of substrates along transects in the two areas (Table 8). In the affected area, the largest difference is a higher proportion of vegetation (14% *vs*. 25%) at frog capture sites than along transects. Outside the affected area, there was a higher proportion of cobbles (10% *vs*. 19%) and a lower proportion of bedrock (13% *vs*. 5%) at frog capture sites than along transects.

Table 8. Comparisons of the proportions of the 8 substrates at frog capture sites and along transects in the two areas.

Substrate	Size (mm)	Code	Edmonds Affected		Not Affected	
			Transects	Frog sites	Transects	Frog sites
Vegetation		1	42%	*36%*	**10%**	***19%***
Silt (Mud)	<0.06	2	10%	*5%*	0%	*3%*
Sand	0.06–2.0	3	3%	*4%*	1%	*1%*
Fine gravel	2–8	4	3%	*1%*	2%	*3%*
Gravel	8–64	5	4%	*1%*	7%	*6%*
Cobble	64–264	6	**14%**	***25%***	25%	*23%*
Boulder	>264	7	23%	*25%*	41%	*41%*
Bedrock		8	3%	*2%*	**13%**	***5%***

Effect of Transect Characteristics on Frog Occurrence and Encounter Rates

Canopy Species

To test whether the presence of individual canopy species on transects had any effect on frog encounter rates, exact tests were undertaken comparing the probability of frogs being on transects with each of the 12 most commonly encountered canopy species with the overall probability of frogs occurring on a transect (i.e. 24/52; p = 0.46). There was a significant ($p < 0.05$) effect for only one canopy species kanuka/manuka, with no frogs occurring on the 6 transects with kanuka/manuka present compared to the 2.8 expected transects (i.e. 6 x 0.46) for the null hypothesis. Thus, with the exception of kanuka/manuka, the presence of individual canopy species on transects has little explanatory value for frog occurrence.

NZLRI Vegetation Types

In the affected area, encounter rates on transects in the two NZLRI vegetation types were not significantly ($p > 0.1$) different (Table 9). All transects in the unaffected area were in a single NZLRI vegetation type: lowland podocarp-hardwood forest (N3a).

Table 9. Summary of the survey results in the two areas by NZLRI vegetation type, with the total numbers of frogs found and the mean numbers of frogs/search. CI95% are 95% confidence intervals for Poisson means (i.e. λ lambda).

Location	NZLRI	N. of			Frogs/Search	
	Veg. Type	Transects	Searches	Frogs	Mean	*(CI95%)*
Edmonds Affected	M 1	18	55	10	0.182	*(0.087–0.334)*
	N 2	22	68	18	0.265	*(0.010–0.418)*
Outside Affected Area	N3a	12	34	37	1.088	*(0.766–0.500)*

Elevation

All transects in the affected area were in a single elevation category (< 400 m a.s.l.). Outside of the affected area there were significant ($p < 0.001$) differences among encounter rates on transect in the three elevation categories (Table 10). The frog encounter rate was highest at 2.50 (CI95%: 1.687–3.569) frogs/search on transects in the mid-altitude category (≥400<500 m a.s.l.), lowest at 0.07 (CI95%: 0.002–0.398) frogs/search at low altitudes (<400 m a.s.l.), and intermediate at 0.750 (CI95%: 0.275–1.632) frogs/search in the highest altitude category (≥500<700 m a.s.l.).

Table 10. Summary of survey results in the two areas by three elevation categories, with the total numbers of frogs found and the mean numbers of frogs/search. CI95% are 95% confidence intervals for Poisson means (i.e. λ lambda).

Elevation	N. of			Frogs/Search	
Class (m a.s.l.)	Transects	Searches	Frogs	Mean	*(CI95%)*
Edmonds Affected Area					
<400	40	123	28	0.228	*(0.151–0.329)*
Outside Affected Area					
<400	5	14	1	0.071	*(0.002–0.398)*
≥400 <500	4	12	30	2.500	*(1.687–3.569)*
≥500 <700	3	8	6	0.750	*(0.275–1.632)*

General Linear Models for Transects in the Edmonds Affected Area

In GLMs with single explanatory variables for data from transects in the Edmonds Affected Area (Table 11) the transect attributes: vegetation type, elevation within 100 m-wide classes, stream margin-width and substrate proportions did not have significant effects ($p > 0.10$) on total frog counts. The attributes stream wet-width and percentage canopy-cover were both significant ($p < 0.10$) in models with Poisson distributions, but not in models with quasi-Poisson or negative binomial distribution. The two collinear variables mean survey duration and mean numbers of refuges were both significant ($p < 0.001$) in models with Poisson or quasi-Poisson distributions, but models with a negative binomial distribution failed when their iteration limit was reached. The influence of percentage canopy cover was trivial, because of the extremely uneven distribution of transects among canopy cover classes: 3, 1, 4 and 32 transects in cover classes 0–24%, 25–49% 50–74% and 75–100 % respectively.

Table 11. Summary of the results from GLMs of the total numbers of frogs found during all searches of each transect in the two areas.

Summary of GLM Results				
	Affected Area		**Outside Affected Area**	
Single transect-level variable models	*p*-value	Coefficient	*p*-value	Coefficient
Vegetation types	*NS*		*na*	
Elevation (5 categories)	*NS*		$p<0.001$	
Elevation (3 categories)	*na*		$p<0.001$	
Stream wet-width	$p<0.1$	0.136	$p<0.001$	-0.884
Stream margin-width	*NS*		$p<0.001$	-0.428
Canopy cover	$p<0.1$		*NS*	
Substrates (1, 2 & 8)	*NS*		$p<0.001$	4.70
Mean duration	$p<0.001$	0.075	$p<0.001$	0.070
Meant N. of refuges	$p<0.001$	0.003	$p<0.01$	0.002

Best multi-level models	*p*-value	Coefficient
Affected Area:		
Wet-width	$p<0.01$	0.258
Mean duration	$p<0.01$	2.64
Mean N. of refuges	$p<0.01$	2.68
Mean duration: Mean N. of refuges	$p<0.05$	-2.46
Outside Affected Area:		
Substrates (1, 2 & 8)	$p<0.1$	2.20
Elevation (3 classes)	$p<0.001$	
Or just		
Elevation (3 classes)	$p<0.001$	

When all four significant explanatory variables (stream wet-width, percentage canopy-cover, mean survey duration and mean numbers of refuges) were included in a multi-way GLM, models with interactions between all variables failed. Starting from a multi-way model containing the four explanatory variables with interactions between mean survey duration and mean numbers of refuges, stepwise regression with backward elimination was used to select the best model. The best models using either Poisson or quasi-Poisson distributions contained three explanatory variables stream wet-width, mean survey duration and mean numbers of refuges with interaction between mean survey duration and mean number of refuges (Table 11). Multi-way models with interactions using the negative binomial distribution failed when their iteration limit was reached. In a dispersion test of the best model (*stream wet-width + mean survey duration x mean number of refuges*), the estimated dispersion parameter was 0.86, and the null hypothesis of equidispersion was accepted ($p > 0.1$). In the best model, frog counts increased with stream wet-width, mean survey duration and mean number of refuges, but declined as a result of interaction between mean duration and mean number of refuges. In a Poisson model with only mean number of refuges and mean survey duration the two variables are collinear, with variance-inflation factor (VIF) = 8.43 and Pearson correlation coefficient = 0.73.

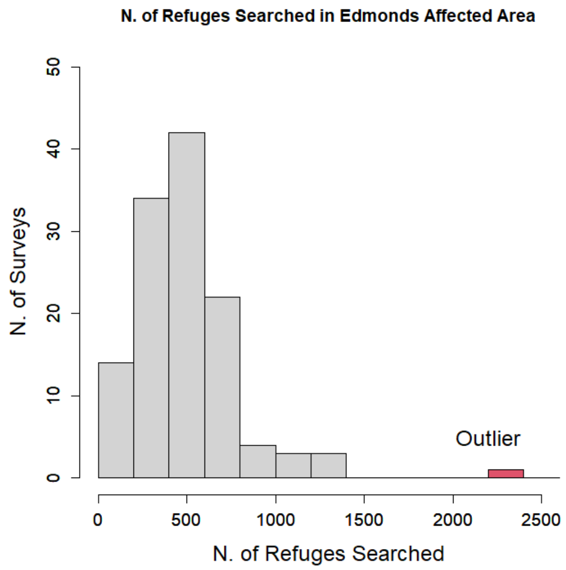

Figure 11. The distribution of values of the number of refuges searched during surveys along transects in the affected area showing the single outlying value.

The result from one survey in Edmonds Affected Area was an outlier. During this survey 2,334 refuges were searched along one transect, well outside the range of between 38 and 1,323 refuges searched during other surveys (Figure 11). The survey's duration was also longer than for other surveys, at 95 minutes compared to between 4 and 71 minutes. Three frogs were found during the survey compared to a maximum of 2 frogs found during all other surveys in the affected area (Figure 12). During two other surveys of the same transect 659 and 132 refuges were searched, with only one frog found during each survey. Trimming the dataset by removing the outlying results from the one survey had little effect on the results of GLM analyses.

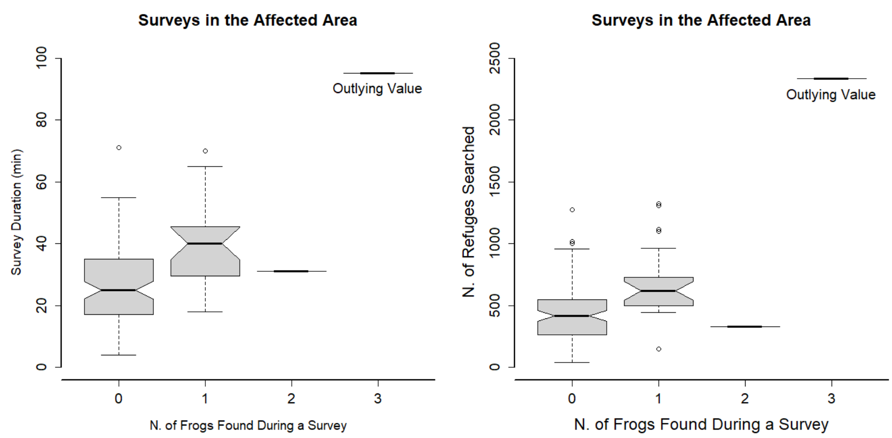

Figure 12. Comparisons of the number of refuges searched (a) and survey duration (b) in the affected area during surveys with different numbers of frogs found.

General Linear Models for Transects Outside the Affected Area

To investigate the influence of elevation on frog counts on transects outside of the affected area 5 elevation categories (<200, ≥200<300, ≥300<400, ≥400<500 and ≥500<700 m a.s.l.) were initially used. During preliminary modelling with GLMs of the data from transects outside of the affected area, frog counts for the three elevation categories <400 m a.s.l. were not significantly different, therefore elevation categories <400 m a.s.l. were collapsed into a single category, creating three elevation categories: <400, ≥400<500 and ≥500<700 m a.s.l. for further analyses.

During preliminary modelling with GLMs of the data from transects outside of the affected area, proportions of the eight substrate types were modelled separately. The proportions of substrate types associated with higher numbers of frogs (substrate types 1, 2 and 8) were combined for further analyses examining the influence of substrates. Vegetation types were not included in models for transects outside of the affected area because all transects were in the same vegetation type, lowland podocarp-hardwood forest (N3a).

In GLMs with single explanatory variables for data from transects outside of the affected area the transect attribute percentage canopy-cover did not have significant effects ($p > 0.10$) on total frog counts. The other attributes (elevation, stream wet-width, stream margin-width substrate proportions and total duration of surveys) were all significant ($p < 0.01$) in models with Poisson or quasi-Poisson distributions. The attribute numbers of refuges was significant ($p < 0.001$) in models with Poisson, but not with quasi-Poisson or negative binomial distributions. Negative binomial distribution models with the elevation variable failed when their iteration limit was reached. Frog counts increased with the increasing total number of refuges searched and total search duration (Table 11) but decreased with increasing stream wet-width and stream margin-width.

Starting from a multi-way model containing the five explanatory variables (elevation, stream wet-width, stream margin-width, substrate, duration and numbers of refuges) with interactions between elevation, stream wet-width, stream margin-width, substrate, stepwise regression with backward elimination was used to select the best model. The best model using either Poisson or quasi-Poisson distributions contained the single explanatory variable elevation with 3 elevation classes ($p < 0.0001$). The dispersion estimate in the quasi-Poisson model was 0.93.

Multi-way models using the negative binomial distribution all failed when their iteration limit was reached.

Generalised Linear Mixed Effect Models for Surveys

In GLMMs of survey results with transect as the random effect and single survey-level explanatory variables, survey duration was significant ($p < 0.001$) for surveys of transects in both areas. The number of refuges searched was also significant ($p < 0.001$) for surveys of transects in the affected area, but not ($p > 0.1$) for transects outside of the affected area. With transect-level explanatory variables added to the models, the best model for surveys of transects in the affected area was a complex model with stream wet-width ($p < 0.05$), survey duration ($p < 0.001$), mean number of refuges and an interaction between mean number of refuges and mean survey duration (i.e. *stream wet-width + number of refuges + mean survey duration:mean number of refuges*). The best model for surveys of transects outside of the affected area with transect-level explanatory variables added to the models was elevation with three classes ($p < 0.05$) and survey duration ($p < 0.01$). Frog counts increased significantly with survey duration in both areas. In the affected area frog counts also increased with stream wet-width (Table 12). On transects outside the affected area frog counts were highest in the mid-elevation class (≥400<500 m a.s.l.), lowest in the low elevation class (<400 m a.s.l) and intermediate in the highest elevation class (≥500 <700 m a.s.l.) (Table 12).

Table 12. Summary of the results from GLMMs of the numbers of frogs found during each survey of a transect in the two areas.

Summary of GLMM Results				
	Affected Area		Outside Affected Area	
Single survey level variable models	*p*-value	Coefficient	*p*-value	Coefficient
Survey number	*NS*		*NS*	
Date	*NS*		*NS*	
Survey leader	*NS*		*NS*	
Start time	*NS*		*NS*	
Temperature	*NS*		*NS*	
RH	*NS*		*NS*	
Survey duration	p<0.001	0.637	p<0.001	0.732
N. of refuges	p<0.001	0.492	*NS*	

Best multi-level models	*p*-value	Coefficient
Affected Area:		
Wet-width	p<0.05	0.379
Survey duration	p<0.001	0.694
Outside Affected Area:		
Elevation (3 categories)	p<0.05	
Survey duration	p<0.01	0.475

N-Mixture Models without Covariates

Estimates of frog abundance from N-mixture models were similar for classical and Bayesian based models (Table 13): 0.91 (CI95%: 0.39 – 2.1) and 1.05 (CI95%: 0.45 – 2.6) frogs per transect in Edmonds Affected Area and 2.9 (CI95%: 1.44 – 5.8) and 3.2 (CI95%: 1.58 – 6.7) outside the affected area. Estimates of frog detection probabilities from the two model types were also similar (Table 13): 0.25 (CI95%: 0.01 – 0.50) and 0.26 (CI95%: 0.01 – 0.45) in Edmonds Affected Area, and 0.36 (CI95%: 0.17 – 0.61) and 0.36 (CI95%: 0.16 – 0.57) outside the affected area.

Table 13. Summary of the results of N-mixture models without covariates for the two areas.

Edmonds Affected Area					
N. Transects	N. Surveys	Total Count	Mean/Survey	SE	*(CI95%)*
40	123	28	0.228	0.043	*(0.151–0.329)*
N-Mixture Estimates:					
	Lambda:		Estimate	SE/SD	*(CI95%)*
		Classical	0.910	0.389	*(0.394–2.10)*
		Bayesian	1.054	0.823	*(0.449–2.64)*
	P-detection:				
		Classical	0.249	0.106	*(0.099–0.502)*
		Bayesian	0.263	0.095	*(0.082–0.451)*

Outside Affected Area					
N. Transects	N. Surveys	Total Count	Mean/Survey	SE	*(CI95%)*
12	34	37	1.088	0.179	*(0.766–1.50)*
N-Mixture Estimates:					
	Lambda:		Estimate	SE/SD	*(CI95%)*
		Classical	2.88	1.02	*(1.44–5.75)*
		Bayesian	3.20	1.43	*(1.58–6.66)*
	P-detection:				
		Classical	0.364	0.119	*(0.173–0.610)*
		Bayesian	0.363	0.107	*(0.161–0.566)*

Abundance estimates for transects outside of the affected area (2.9 and 3.1 frogs/transect) were considerable higher than estimates from the affected area (0.91 and 1.05 frogs/transect). Detection probabilities were also higher outside of the affected area, 0.36 compared to 0.25 and 0.26. Difference in estimates from the two areas stem from the much higher numbers of frogs (11) found during surveys of two high elevation (≥ 400 m) transects. By comparison, all transects in the affected area were < 340 m a.s.l..

The wide 95% confidence intervals (CI95%) around all the estimates are to some extent a feature of N-mixture models, as they reflect uncertainty on the relative contributions of abundance and detection probabilities to generation of the observed counts. However, features of the observed data add to the uncertainties in estimates, with low numbers of frogs (Mean 0.28; Mode 0; Median 0; Range 0–3) found during surveys of transects in the affected area and the small sample size (12 transects) outside of the affected area.

Extrapolating N-mixture abundance estimates for the forty 20 m-long transects in Edmonds Affected Area to the 12.1 km of streams in the area's sampling frame, provides estimates of the total Hochstetter's frog population in the affected area of 549 (CI95%: 238 – 1,270) and 637 (CI95%: 271 – 1,597) from classical and Bayesian analyses respectively. Estimates of the total Hochstetter's frog population in the sampling frame outside of the affected area by extrapolating from transect abundance estimates were not attempted because of the small, unrepresentative sample of transects.

N-Mixture Models with Covariates

Transects in Edmond's Affected Area

The significant explanatory variables identified in GLMs and GLMMs of the count data from surveys of transects in Edmond's Affected Area were used in N-mixture models to estimate frog abundance from frog counts. In N-mixture models, site covariates are used as predictors of the number of frogs present on transects and observation covariates are used as predictors of the detection probability for frogs present on the transect. Two site covariates were used in the N-mixture model: mean number of refuges searched during surveys of transects and mean stream wet-width along transects. A single observation covariate was used: survey duration.

In N-mixture models obtained using classical inference with the R-package *unmarked* both site covariates had significant effect on frog abundance, with mean number refuges $p < 0.01$ and stream wet-width $p < 0.1$. The observation covariate survey duration was highly significant ($p < 0.001$) in models without mean number of refuges as a site covariate, but less so ($p < 0.05$) in models that included the mean number of refuges as a site covariate. In a plot of the relationship between predicted abundance and stream wet-width (Figures 13), the 95% confidence interval around the modelled line for predicted abundance, ranged between 8.1 and 48.6, with the upper confidence limit outside the plot bounds. This indicates that wet-width is a poor predictor of abundance. By comparison the 95% confidence interval for the relationship between predicted abundance and mean number of refuges searched is much narrower (Figure 13), ranging between 1.4 and 6.8.

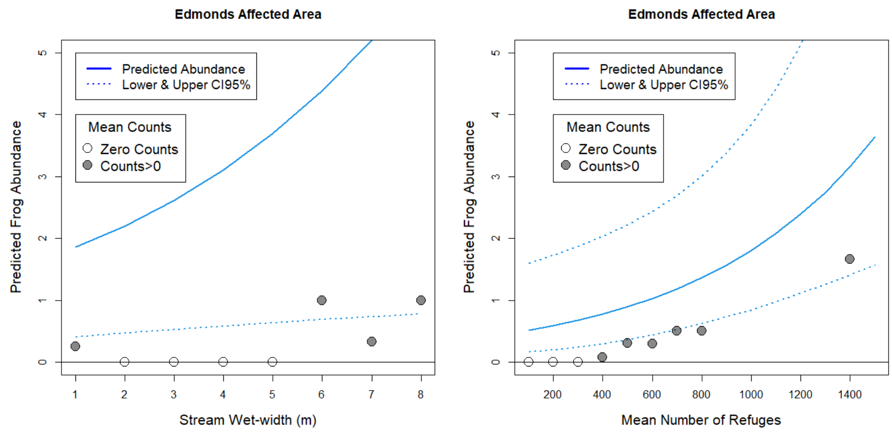

Figure 13. The effect of stream wet-width (a) and mean number of refuges searched (b) on the predicted abundance of frogs on transects in the affected area from N-mixture models with covariates.

In the model with the mean number of refuges as a site covariate, the average abundance estimate is 0.944 frogs/transect and the average detection probability is 0.214. These estimates are close to estimates from N-mixture models without covariates: 0.91 and 1.05 frogs/transect and detection probabilities of 0.25 and 0.26 for classical and Bayesian estimates respectively. By contrast, the model with stream wet-width as a site covariate provide a much higher average abundance estimate of 2.31 frogs per transect and a much lower average detection probability of 0.096.

Transects Outside of the Affected Area

In N-mixture models of count data from surveys of transects outside of the affected area elevation with three classes was used as a site covariate and survey duration as an observation covariate. Results of N-mixture modelling showed strong support ($p < 0.05$) for elevation as an explanatory variable, but little support for survey duration ($p > 0.10$), which had extremely wide confidence interval around the predicted relationship (Table 14 and Figure 14). The average abundance estimate from the model for transects outside of the affected area was 2.21 frogs/transect with an average detection probability of 0.501. These results are different to the results from N-mixture models without covariates for the area, which were 2.9 and 3.2 frogs/transect and a detection probability of 0.36 (Table 13).

Table 14. Summary of the results of N-mixture models for surveys outside affected areas with elevation class as a covariate.

Elevation Class	Frogs/Transect		
(m a.s.l.)	Predicted	SE	*(CI95%)*
<400	0.23	0.23	*(0.03–1.64)*
≥400 <500	5.09	1.41	*(2.96–8.75)*
≥500<700	1.70	0.85	*(0.64–4.54)*

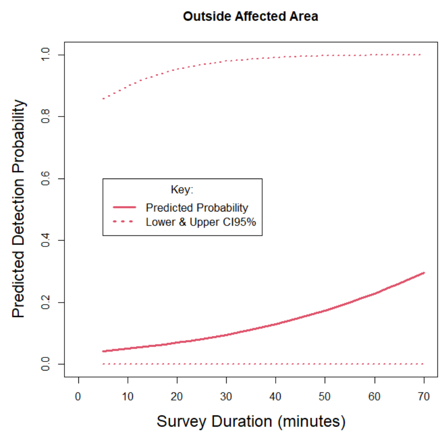

Figure 14. The effect of survey duration on the predicted detection probability for frogs during surveys of transects outside the affected area from N-mixture models with covariates.

Survey Duration and Numbers of Refuges Searched

The two variables survey duration and the numbers of refuges searched during a survey are the dominant explanatory variables in models of frog counts from the affected area (Tables 11 & 12), while survey duration and elevation class are explanatory variables for frog counts from outside the affected area. However, the role of survey duration and the numbers of refuges searched during a survey as explanatory variables in models of frog counts is problematic. Intuitively the two variables should be correlated, with survey duration increasing as more refuges are searched. The two variables are correlated (Figure 15) with the Pearson product-moment correlation coefficient $\rho = 0.61$, which is less than expected. In GLMs of search counts the two variables are multicollinear with Variance Inflation Factors of 17.

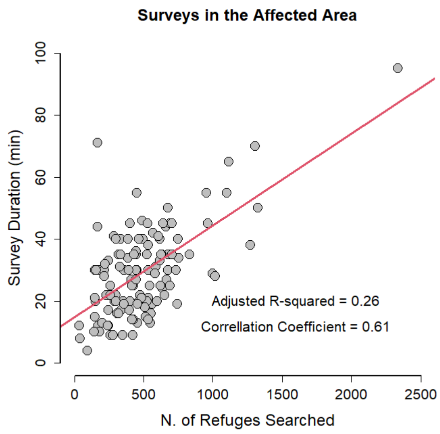

Figure 15. The relationship between number of refuges searched and survey duration for surveys of transects in the affected area.

It seems likely that the two variables provide measures of search effort during individual surveys, consequently detection probabilities should increase with survey duration and the number of refuges searched. However, the number of refuges searched during a survey along a transect could also be a measure of frog habitat quality and be a predictor of frog abundance along the transect. If the number of refuges searched during a survey is a transect-level variable predicting frog abundance along a transect, the numbers of refuges searched during different searches of the same transect should be similar. This isn't the case. Correlation coefficients comparing the numbers of refuges searched during different surveys of the same transect are relatively low, ranging between 0.33 and 0.53 (Table 15). Correlation coefficients comparing survey duration for different searches of the same transect are lower, ranging between -0.17 and 0.24 (Table 15), indicating survey duration is not a function of transect characteristics. It is possible that survey duration increases when frogs are found during a search because of time spent processing the frogs that are found.

Table 15. Correlation between the three searches of transects for values of survey duration and number of refuges searched.

Search Comparisons	Correlation Coefficient	
	Duration	N. Refuges
1st and 2cnd	-0.17	0.45
1st and 3rd	0.24	0.53
2cnd and 3rd	0.11	0.33

DISCUSSION

N-mixture Models

N-mixture models (J. A. Royle, 2004) were developed to model animal population size from point counts with imperfect detection of individuals, where capture-recapture methods are not possible. Because N-mixture models estimate both abundance and detection from replicated counts, model parameters may not be clearly identifiable (Madsen & Royle, 2023), consequently confidence intervals around estimates of abundance and detection probabilities can be wide.

BACI Design

The surveys described in this report are preliminary before-impact surveys undertaken as part of a Before-After Control-Impact (BACI) monitoring programme to assess the effect of possible future stream flow reductions on Hochstetter's frogs. In the BACI monitoring programme, surveys of the 40 transects in Edmonds Affected area are undertaken to measure impact, while surveys of 12 transects outside of Edmonds Affected area provide a control (i.e. non-treatment) sample. However, transect characteristics in the two areas are very different, with imbalances or little, or no, overlap in the values of many of the likely explanatory variables or covariates for transects in the two areas.

Imbalance occurs when the distribution of values of pre-treatment variables differ for treatment and non-treatment samples. Lack of overlap occurs when there are regions in the space of variables that are not present in both treatment and non-treatment samples. Chapter 10 of Gelman and Hill (2007) (Gelman & Hill, 2007) provides a detailed discussion of how imbalance and lack of overlap limits the causal conclusions that can be made from BACI data. In summary, when treatment and control groups:

- *are unbalanced, the simple comparisons of group averages is not a good estimate of the treatment effect. Instead, some analysis must be performed to adjust for pre-treatment differences between the groups.*
- *do not completely overlap, the data are inherently limited in what they can tell us about the treatment effects in the region of nonoverlap. No amount of adjustment can create direct treatment/ control comparisons, and one must either restrict*

inferences to the region of overlap, or rely on a model to extrapolate outside of this region.

Imbalance and, or lack of overlap in values of the variables in samples from transects in the two areas are apparent in several of the potentially confounding covariates. There is no overlap in NZLRI vegetation types (Figure 4 & Tables 2 & 9), limited overlap in elevation values (Figure 4 & Table 1) and substantial imbalance in both stream substrate composition (Figure 5 & Tables 3 & 8) and stream wet-widths (Figure 6). Thus, the 12 transects surveyed outside of Edmonds affected area do not provide a suitable pre-treatment control sample in a BACI monitoring programme for assessing the effects of the de-watering treatment in the affected area and the surveys described in this report should only be considered as pilot surveys for developing a robust monitoring programme.

For a BACI monitoring programme the values of potentially confounding covariates for transects in the non-treatment or control area should be similar to values for transects in the treatment area. This can be achieved by selecting transects for the non-treatment sample in sections of streams within the Wharekirauponga Stream catchment that will not be affected by potential dewatering but are at similar elevations (100 – 400 m a.s.l.) and with similar NZLRI vegetation types (manuka/kanuka or kauri forest) to streams in the affected area. There are suitable stream sections for non-treatment in nearby Teawaotemutu, Adams, Thompson streams and the higher reaches of Edmond's stream, as well as the more distant Lignite Stream to the north (Figure 16).

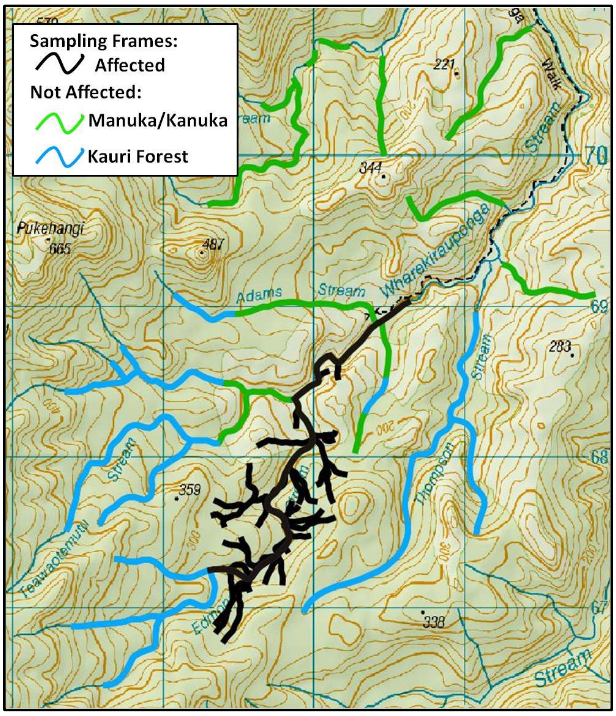

Figure 16. A proposed alternative sampling frame for the non-treatment area outside of Edmonds affected area.

Multicollinearity: Survey Duration and Number of Refuges Searched

Multicollinearity in regression models occurs where some of the explanatory variables in a model are correlated. Stepwise regression to eliminate variables with less predictive power is a common response to multicollinearity in multilevel models and was used in these analyses. However, multicollinearity can be an important natural feature of the phenomena being studied and there are arguments for retaining multicollinear variables in models.

Despite using stepwise regression to select the best multi-level GLM, the two multicollinear variables survey duration and number of refuges searched, as well as an interaction between the two, were retained in the best GLM for frog counts on transects in the affected area (Table 11). On this basis, in N-mixture modelling with covariates for the affected area survey duration was used as a covariate predicting detection probabilities, while the number of refuges searched, along with stream wet-width, were used as covariates to predict frog abundance on the transects.

As discussed in the previous section, the role of survey duration and the numbers of refuges searched during a survey as explanatory variables in models of frog counts is problematic. It would be worthwhile redesigning data collection to clarify their roles. If, as seems likely, the number of refuges available to be searched on a transect is a characteristic of the transect, the same or similar numbers of refuges should be searched during each replicate survey of an individual transect based on the number of refuges available along the transect. This makes it clear that the number of refuges searched is a transect-level variable likely to be associated with frog abundance (i.e. the actual numbers of frogs on a transect), not a survey-level variable measuring search effort and presumably detection probability during individual surveys.

Sample Size

The precision of estimates as well as confidence in the results of modelling generally increase with sample size. Although the sample of 3 or more replicate surveys of 40 transects in the treatment area is a satisfactory sample for estimating frog abundance in the area, sparseness of frogs frustrates any attempt at modelling the effects of covariates on frog abundance in the treatment area. Despite higher frog abundance in the non-treatment area, the small sample of 12 transects surveyed in the area, together with the lack of overlap and imbalance in covariates for treatment and non-treatment areas compromises the use of these data in a BACI monitoring programme investigating the effects of dewatering.

During each replicate survey of a transect considerable time was spent collecting information on transect characteristics unlikely to change between surveys. This information has not proved useful in predicting frog abundance, largely because so few frogs found during surveys. Investing less time in collecting transect information would allow more transects to be surveyed, which will provide a better basis for assessing the effects of dewatering.

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APPENDIX 1: CANOPY SPECIES ON TRANSECTS

Species	Common Name	N. of Transects With Species	Proportion of 52 Transects	N. Transects With Frogs	Expected	Difference In Proportion from 0.46	Proportion With Frogs	Low CI95%	Upper CI95%	P-value
Dicksonia sp. & Cyathea sp.	Tree-fern	46	0.88	20	21.2	-0.025	0.43	0.289	0.589	0.769
Rhopalostylis sapida	Nikau	36	0.69	16	16.6	-0.016	0.44	0.279	0.619	0.869
Coprosma autumnalis	Kanono	31	0.60	17	14.3	0.088	0.55	0.360	0.727	0.370
Melicytus ramiflorus	Mahoe	30	0.58	13	13.8	-0.027	0.43	0.255	0.626	0.855
Beilschmiedia tawa	Tawa	26	0.50	14	12.0	0.078	0.54	0.334	0.734	0.440
Knightia excelsa	Rewarewa	23	0.44	11	10.6	0.018	0.48	0.268	0.694	1.000
Pterophylla racemosa	Kamahi	15	0.29	5	6.9	-0.127	0.33	0.118	0.616	0.439
Laurelia novae-zelandiae	Pukatea	14	0.27	8	6.4	0.111	0.57	0.289	0.823	0.434
Brachyglottis repanda	Rangiora	12	0.23	6	5.5	0.040	0.50	0.211	0.789	1.000
Schefflera digitata	Pate	8	0.15	5	3.7	0.165	0.63	0.245	0.915	0.484
Kunzea ericoides & Leptospermum scoparium	Kanuka/Manuka	6	0.12	0	2.8	-0.460	0.00	0.000	0.459	0.034*
Hedycarya arborea	Pigeonwood	6	0.12	3	2.8	0.040	0.50	0.118	0.882	1.000
Freycinetia banksii	Kiekie	5	0.10	1	2.3	-0.260	0.20			
Pseudopanax crassifolius	Lancewood	5	0.10	1	2.3	-0.260	0.20			
Dacrydium cupressinum	Rimu	5	0.10	2	2.3	-0.060	0.40			
Phyllocladus trichomanoides	Tanekaha	5	0.10	2	2.3	-0.060	0.40			
Ripogonum scandens	Supplejack	4	0.08	2	1.8	0.040	0.50			
Coprosma-robusta	*Coprosma robusta*	3	0.06	1	1.4	-0.127	0.33			

Species	**Common Name**	**N. of Transects With Species**	**Proportion of 52 Transects**	**N. Transects With Frogs**	**Expected**	**Difference In Proportion from 0.46**	**Proportion With Frogs**
Neopanax arboreus	Five-finger	3	0.06	3	1.4	0.540	1.00
Elaeocarpus dentatus	Hangehange	2	0.04	2	0.9	0.540	1.00
Pterophylla sylvicola	Towai	2	0.04	1	0.9	0.040	0.50
Olearia rani	Heketara	1	0.02	1	0.5	0.540	1.00
Hoheria populnea	*Hoheria*	1	0.02	1	0.5	0.540	1.00
Corynocarpus laevigatus	Karaka	1	0.02	0	0.5	-0.460	0.00
Agathis australis	Kauri	1	0.02	0	0.5	-0.460	0.00
Piper excelsum	Kawakawa	1	0.02	1	0.5	0.540	1.00
Didymocheton spectabilis	Kohekohe	1	0.02	1	0.5	0.540	1.00
Carpodetus serratus	Marbleleaf	1	0.02	1	0.5	0.540	1.00
Metrosideros robusta	Northern rata	1	0.02	1	0.5	0.540	1.00
Pinus radiata	*Pinus radiata*	1	0.02	0	0.5	-0.460	0.00