



## Bioacoustic monitoring of lower North Island bird communities before and after aerial application of 1080

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**Abstract:** Modern aerial 1080 (sodium monofluoroacetate) operations are effective in reducing population densities of possums, rats and stoats, thereby reducing predation pressure on birds. Debate regarding 1080 use, however, continues to centre on potential non-target effects, with some groups claiming that birds are killed in large numbers such that forests “fall silent”. We investigated these claims by recording birdsong for 5–8 weeks before and after three separate 1080 operations in the Aorangi and Remutaka Ranges of the lower North Island, New Zealand. A Before-After/Control-Impact (BACI) design was employed for each operation, with simultaneous monitoring carried out in treated and untreated sites using autonomous recording units. The amount of birdsong was quantified, scoring the presence/absence of species’ calls in 30 ten second sub-samples for each of the 695 recordings analysed. The interaction between treatment and time and the “BACI contrast” was tested for at the community- and population-level. After each operation, the amount of silence was the same or lower in sites treated with 1080 relative to paired non-treated sites. Six taxa showed no evidence of an effect of 1080, while two of 26 taxa/treatment tests showed a significant interaction between treatment and time consistent with the silent forest hypothesis: chaffinch and tomtit calling rates declined following the Aorangi 2017 and Aorangi 2014 operations respectively. At  $p < 0.05$ , one or two “significant” results in 26 may be expected by chance. A negative impact on the introduced chaffinch (observed in one of three operations) is plausible considering the species’ granivorous diet. The pathway for a negative impact on insectivorous tomtits is less obvious; as a precautionary approach we recommend further research. Overall, we found little evidence of forests falling silent after aerial 1080 operations and the likelihood of negative population-level impacts via poisoning of native birds was concluded to be very low.

**Keywords:** 1080, autonomous recording units, ARU, avian, BACI, non-target impacts, silent forest, sodium monofluoroacetate

### Introduction

The toxin sodium monofluoroacetate (also known as “1080”) is aerially distributed at a significant scale across New Zealand for the control of introduced mammals that act as predators of native species, vectors of bovine tuberculosis, and significant herbivores of native vegetation (Eason et al. 2010; Nugent & Morriss 2013; Byrom et al. 2016). Control is often carried out in the interest of protecting native avifauna, with the intention of reducing predation pressure and thus increasing nesting success and adult survival to facilitate species’ recoveries (Starling-Windhof et al. 2011; Fea et al. In Press). However, when toxic baits are broadcast aerially there is the risk of poisoning non-target native species (Veltman & Westbrooke 2011). Some species are more susceptible to poisoning than others (Eisler 1995; Lloyd & McQueen 2000), and at a population-level the impact of an aerial 1080 operation depends on a species’ ability to offset any immediate population losses with improved breeding success following control (Eason et al. 2011; Veltman & Westbrooke 2011). If a species benefits

long-term from reduced predator numbers but consistently suffers short-term population losses from poisoning, regimes of frequent treatment may result in a net-negative impact on its populations (Veltman & Westbrooke 2011). It is therefore essential that any non-target impacts are known and well-understood.

Since the beginning of operations in 1956, individuals from 19 native bird species have been found dead following aerial 1080 drops (Spurr 2000). Most of these mortalities occurred pre-1980, when carrot baits were the predominant bait type used to carry the toxin. Subsequent changes in baiting practices including a shift to the use of cereal baits, the use of non-toxic pre-feeds, and a reduction in sowing rates, prompted by studies in the 1970s, have significantly reduced risk to non-target avian species (Eason 2002; Eason et al. 2011; Veltman & Westbrooke 2011; Morriss et al. 2016). Nonetheless, further changes to baiting practices call for continued vigilance with respect to non-target impacts (Veltman & Westbrooke 2011; Veltman et al. 2014).

The use of 1080 remains a controversial issue in

New Zealand. While official complaints regarding its use have declined (Environmental Protection Authority 2012), so too has public support for its use (Eason et al. 2010; Green & Rohan 2012). The incidental poisoning of non-target native avifauna continues to be raised as a concern in ongoing debates regarding aerial operations (Green & Rohan 2012; Morriss et al. 2016), with some groups claiming that forests “fall silent” following such operations, and that this is evidence of significant adverse impacts upon native bird species (Hansford 2016). Despite little evidence for such adverse impacts occurring over the past twenty years, continued monitoring of the responses of bird populations to aerial 1080 operations in a range of habitats and methodological situations is required to ensure that any positive or negative effects are documented and to provide both up-to-date evidence to public debates and information for the continual improvement of management techniques.

Bioacoustic monitoring using autonomous recording units (ARUs) is a rapidly growing monitoring method with good potential to monitor bird populations over large-scale management treatments (Klingbeil & Willig 2015; Mortimer & Greene 2017). Traditionally, *in situ* five-minute bird counts (5MBCs) have been used extensively for monitoring bird species’ population responses to 1080 operations (Spurr & Powlesland 2000). However, the 5MBC is subject to some limitations. It requires trained observers and significant field effort, which can limit the spatial scale, temporal scale, and comparability of data. Furthermore, inter-observer variability in identification skills and observer presence in-field can introduce observer and detectability biases in data respectively (Aide et al. 2013; Klingbeil & Willig 2015). Bioacoustic monitoring with ARUs is not subject to such limitations: once deployed, units can record data simultaneously at multiple locations for long periods of time in the absence of observers, collecting comparable recordings which can then be analysed by a single observer out of field (Steer 2010; Aide et al. 2013; Klingbeil & Willig 2015). This significantly reduces the field effort required for collecting data, allowing an increase in the spatial and temporal scale across which data can be collected, and minimising detectability biases introduced by time and observer presence in in-field counts.

Unless relationships between calling rates and species’ densities are known, the data extracted from most bioacoustic monitoring provide an index of relative abundance (Efford et al. 2009; Marques et al. 2013; Stevenson et al. 2015). Calculated here as an acoustic prevalence index (API) derived from the presence and absence of a species’ call being detected in thirty sub-samples of an acoustic recording (Cook & Hartley 2018), inferences drawn from this index assume that vocal activity is positively related to population size (Royle 2004), a relationship that has been found to hold true for a range of taxa (Nelson & Graves 2004; Thompson et al. 2010; Oppel et al. 2014). This same assumption underpins the claim that, should a forest “fall silent” following an aerial 1080 operation, such a decline in birdsong is evidence for an adverse impact of the operation on bird species’ populations. Accordingly, the acoustic data captured by ARUs provide a means to quantitatively test for this effect.

In this study, we sought to use ARUs to monitor and compare the short-term community- and species-level trends of bird song in treatment and non-treatment areas before and after three different aerial 1080 operations occurring in the Aorangi and Remutaka Ranges of the lower North Island of New Zealand. Specifically, we aimed to determine if, relative to non-treatment areas: (1) the amount of silence in recordings

from treatment areas increased after operations, and (2) the acoustic prevalence of any species showed a significant decline in treatment areas in the weeks following the distribution of 1080.

## Methods

### Study areas

Autonomous recording units were deployed across five study areas within the Wellington region: the Aorangi, Northern Remutaka, Southern Remutaka and Tararua Ranges, and Tora Bush. Within a study area, between one and six study sites were monitored (Fig. 1a). A study site comprised between one and nine ARUs that could be serviced (i.e. have their batteries and SD cards replaced) in one day. Vegetation in the study areas was representative of the southern North Island, consisting of a mix of southern beech, broadleaf, and mixed beech/broadleaf forest, with podocarps scattered throughout. Common broadleaf species included hīnau (*Elaeocarpus dentatus*), rewarewa (*Knightia excelsa*), and kāmahi (*Weinmannia racemosa*) (SH, pers. obs.). Beech (*Fuscospora* spp. and *Lophozonia menziesii*) tend to dominate with increasing elevation, and in areas with dry climates and infertile soils (Dymond & Shepherd 2004). Regenerating mānuka/kānuka (*Leptospermum scoparium/Kunzea ericoides*) scrub also exists in the region where vegetation has been modified by historic burning and pastoralism (Wardle 1967; Dymond & Shepherd 2004).

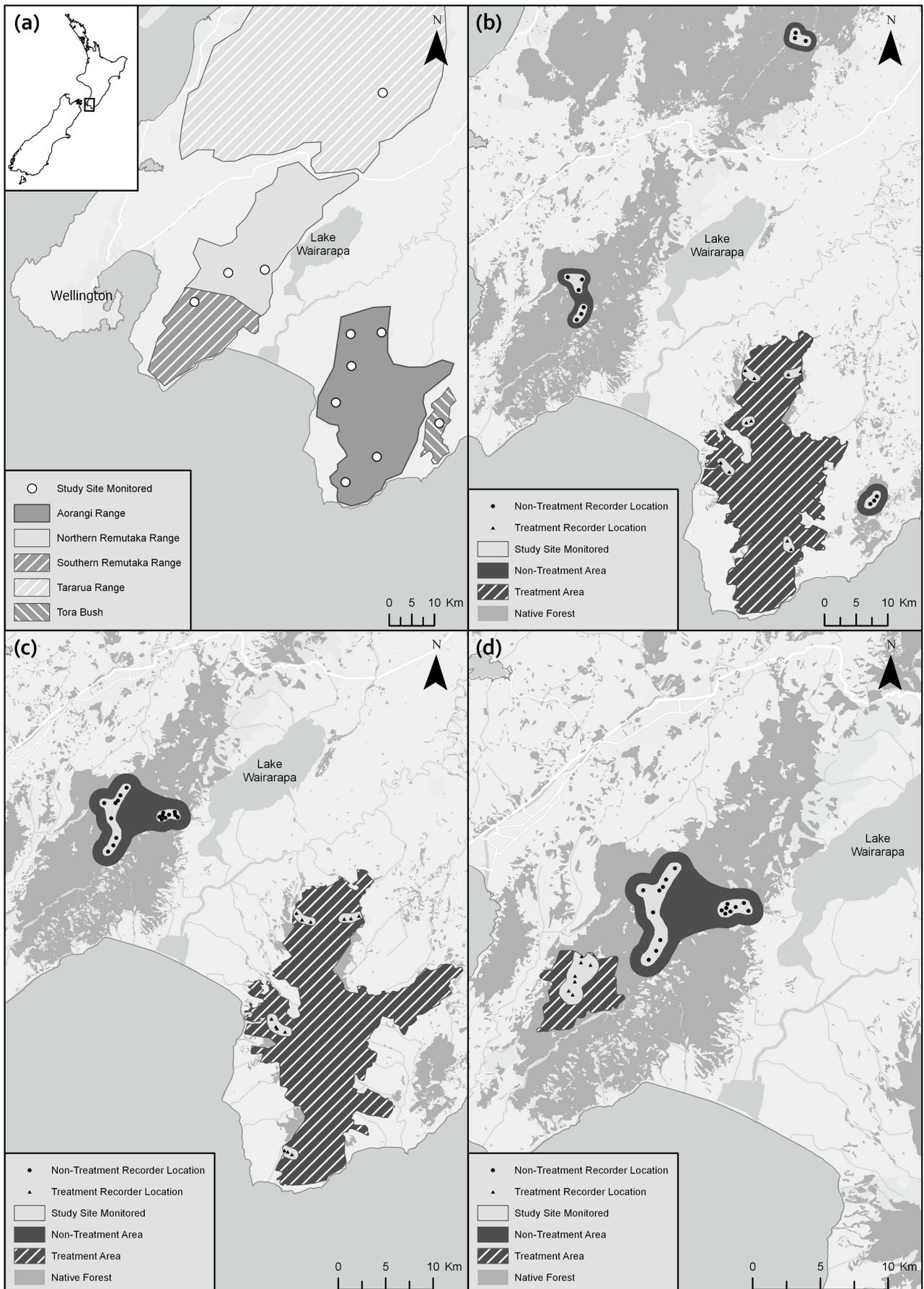
The Wellington region has a mild, wet climate; mean monthly temperatures near sea level range from 8°C (July) to 16°C (January), and annual rainfall varies from 800 mm in the eastern Wairarapa to 2400 mm in the higher elevations of the Remutaka Ranges (Brockie 1992), and up to 7000 mm on the peaks of the Tararua Ranges (Dymond & Shepherd 2004).

### 1080 operations

The Aorangi Range currently receives aerial 1080 treatment for possum control every three or four years. The two most recent operations occurred in August 2014 and June 2017, where approximately 30 000 ha of the range were treated. Cereal baits containing Epro deer repellent were used for both the non-toxic (pre-feed) and toxic bait applications (D. Viner, OSPRI, pers. comm.; Table 1). Prior to 2014, the northern and southern half of the Aorangi Range last received 1080 in July 2009 and September 2006 respectively (D. Viner, OSPRI, pers. comm.).

The Southern Remutaka Range area received its first and most recent aerial 1080 treatment in July 2017, where approximately 10 000 ha of the range was treated. Cereal baits containing Epro deer repellent were used for both the non-toxic (pre-feed) and toxic bait applications (D. Viner, OSPRI, pers. comm.; Table 1).

During the course of this study, Tora Bush, the Tararua and the Northern Remutaka Ranges did not receive 1080 treatment, and were therefore used as experimental control sites for the afore-mentioned 1080 operations. Prior to monitoring commencing in 2014, Tora Bush had not received aerial 1080 treatment since at least 2009 (S. Burles, Department of Conservation, pers. comm.). The Tararua and Northern Remutaka Ranges last received treatment in 2010 and 2012 respectively (TBfree 2010; Uys & Crisp 2018).



**Figure 1.** Maps of (a) the study sites and approximate forested extent of focus study areas monitored; and the study areas, study sites, and recorder locations monitored over (b) the August 2014 Aorangi; (c) June 2017 Aorangi, and; (d) July 2017 Southern Remutaka operations.

**Table 1.** Specifications of non-toxic pre-feed and toxic 1080 bait applications of operations monitored (EDR = Epro deer repellent).

Operation	Date	Bait specifications	Sowing Rate	Distribution method
Aorangi 2014 pre-feed	4–5 August 2014	6 gram cereal bait, 0% toxicity + EDR	0.5 kg ha <sup>-1</sup>	180 m swaths with 40 m intervening strips of non-application
Aorangi 2014 1080	18–27 August 2014	12 gram cereal bait, 0.15% toxicity + EDR	1.0 kg ha <sup>-1</sup>	180 m swaths with 40 m intervening strips of non-application
Aorangi 2017 pre-feed	30–31 May 2017	6 gram cereal bait, 0% toxicity + EDR	1.0 kg ha <sup>-1</sup>	260 m swaths, full broadcast
Aorangi 2017 1080	16–17 June 2017	12 gram cereal bait, 0.15% toxicity + EDR	1.5 kg ha <sup>-1</sup>	260 m swaths, full broadcast
Southern Remutaka 2017 pre-feed	16–17 June 2017	6 gram cereal bait, 0% toxicity + EDR	2.0 kg ha <sup>-1</sup>	180 m swaths, full broadcast
Southern Remutaka 2017 1080	30 July 2017	12 gram cereal bait, 0.15% toxicity + EDR	2.0 kg ha <sup>-1</sup>	180 m swaths, full broadcast

### Experimental design

Bird calls were monitored over three separate aerial 1080 operations: one in 2014 and two in 2017. All studies employed a Before-After/Control-Impact (BACI) experimental design (Underwood 1992), using ARU recordings to compare changes in the acoustic prevalence of native diurnal bird species in treatment and non-treatment areas from before to after respective 1080 operations. Slight differences in equipment and sampling strategy, outlined following, were employed between years.

#### 2014 Aorangi operation

Recordings were made by 10 ARUs distributed across five study sites in one treatment area, the Aorangi Range, and 12 ARUs distributed across four study sites in three non-treatment areas: the Tararua and Northern Remutaka Ranges, and Tora Bush (Fig. 1b). Recorders were spaced at least 800–1000 m apart.

#### 2017 Aorangi operation

Recordings were made by 13 ARUs distributed across four study sites in one treatment area, the Aorangi Range, and 18 ARUs distributed across two study sites in one non-treatment area, the Northern Remutaka Range (Fig. 1c). Six recorders (two in the treatment area and four in the non-treatment area) were spaced 350 m from the next-nearest recorder. All other recorders were spaced at least 400 m apart.

#### 2017 Southern Remutaka operation

Initially, 18 ARUs were distributed across two study sites in one treatment area, the Southern Remutaka Range, based on proposed operational boundaries. However, one site fell outside of treatment boundaries after the operation was carried out. Furthermore, two recorders failed in the site remaining within the final treatment area. Thus, recordings were made by 7 ARUs distributed across one study site in the treatment area, and 17 ARUs distributed across two study sites in one non-treatment area, the Northern Remutaka Range (Fig. 1d). Three non-treatment recorders were spaced 350 m from the next-nearest recorder. All other recorders were spaced at least 400 m apart.

### Audio recording specifications

In 2014, audio recordings were made using Wildlife Acoustics Song Meter™ SM2+ ARUs set to record at 44 100 samples

per second (kHz). In 2017, recordings were made using Department of Conservation ARUs (purchased in 2013 and 2016) set to record at the “high frequency” pre-set of 32 000 kHz. These sample rates allow for playback at 22.05 kHz and 16 kHz respectively (Charif et al. 2010). Both of these settings allow the high frequency calls of riflemen, with their power maximum at 10–11 kHz, to be recorded and visualised on a spectrogram (Krull et al. 2009).

All recordings were made in mono and saved as 16-bit .wav files. Recorders were mounted to tree trunks approximately 1.5 m above the ground and situated away from rivers where possible. If a river was nearby, recorders were mounted on the side of the tree opposite to the river to reduce the amount of river noise in recordings.

Recorders were set to record simultaneously across all locations to minimise the effects of intra-day variability in species' calling. Recordings were collected from 0800–0830 h (NZST) and were made daily from eight weeks before to eight weeks after the 2014 operation's application of toxic baits, and for at least six weeks before and six weeks after each of the 2017 operations.

### Selection of audio recordings

#### 2014 operation

Recordings from five days before and five days after the 2014 operation were analysed. The first four days were selected by date: precisely one week and six weeks both before and after the 1080 drop (7 July, 11 August, 25 August, and 2 October 2014). The next six days were selected at one to three week intervals taking into account weather conditions so as to reduce any influence of background noise caused by wind and rain. Days with < 2 mm total rainfall and < 10 km hr<sup>-1</sup> max wind gust were deemed suitable for analysis, based on climate data collected from three weather stations distributed throughout the study region (NIWA 2015) (see Appendix S1 in Supplementary Materials). Of the 31 days passing these criteria, the days closest to two, three and eight weeks both before and after the 1080 drop were analysed.

Eight audio recordings were missing due to malfunction of the ARUs, thus the total number of recordings analysed was 232 (Appendix S2). Of the four days chosen by date alone, one before (11 August 2014) and one after (2 October 2014) the operation would not have passed the weather criteria used to select the other six days. However, inspection of their

recordings showed that background noise was not excessive and, given that they occurred one each on either side of the operation, any potential impact of weather on calling rates for these days would be unlikely to bias results.

### 2017 operations

Recordings were analysed for a five week period before and after each 1080 operation, with one recording analysed per sample week. For each respective drop, the before period consisted of the five weeks leading up to the date of the toxic drop of interest. After periods spanned weeks two to six after each respective drop (i.e. one week separated before and after sampling). Sample days were selected from days within these periods identified to have recordings suitable for analysis.

Identification of suitable days comprised two phases: primary inspection of weather data, and subsequent inspection of recording spectrograms. For each sampling period, rainfall (cumulative mm, 0800–0900 h NZST) and wind speed ( $\text{km h}^{-1}$  at 2.5–10 m, 0800 h NZST) data were inspected as the average of five rainfall and five wind speed weather stations in the wider sampling area (Greater Wellington Regional Council 2018) (Appendix S1). Days with both a mean wind speed of  $< 15 \text{ km h}^{-1}$  and 0 mm of cumulative rainfall were identified as potentially suitable for sampling. The full spectrograms of recordings from these days were then inspected using Raven Lite 2.0 (Center for Conservation Bioacoustics 2014) and classified as suitable for analysis if neither heavy rain nor consistent strong wind were present. Some recordings were missing where an ARU recording failed; these recordings were classified as unavailable for analysis.

Sample days were selected from suitable days to maximise sample size and recording quality: for each week, the day with the greatest number of suitable recordings across all sites was chosen for analysis. However, if less than four days separated the two preferred sample days of two consecutive weeks, the day with the second highest number of suitable recordings between the two weeks was instead chosen for its targeted week. This was done to increase temporal independence between successive samples.

Using this method, recordings affected by high levels of wind or rain were scarce in the days selected. Thus, all recordings available for the sample days selected were analysed. A total of three recordings were unavailable for the sample days chosen due to ARU failures, reducing final sample sizes to 308 and 239 recordings for the Aorangi and Southern Remutaka operations respectively (Appendices S3 and S4).

### Scoring of audio recordings

Audio recordings were scored employing the “intermittent” subsampling method described by Cook and Hartley (2018). This method scores the presence (1) or absence (0) of species’ calls in the first ten seconds of every minute in each recording. This produces a score between 0 and 30 for each species from 30 ten second sub-samples sampled across 30 minutes. The score produced for each species in each recording analysed is converted to an acoustic prevalence index (API), calculated as the proportion of sub-samples in which a species was detected. For example, a species calling in 15 of 30 sub-samples analysed in a recording would have an API of 0.50.

The sub-samples analysed using this method total a sampling effort of five minutes. When compared to scoring species’ presence and absence in 30 consecutive ten second sub-samples of a five minute recording, the intermittent method has been found to both increase the number of species detected,

and decrease variance around estimates of acoustic prevalence because it reduces the short-range temporal autocorrelation of data (Cook & Hartley 2018).

For all analyses, the file names of recordings were randomly coded such that the treatment-type and recording date were unknown to the observer (i.e. analyses were blind). Scoring of species’ presence/absence was carried out by one observer (Asher Cook) using Raven Pro 1.4 for the 2014 operation (Center for Conservation Bioacoustics 2011), and one observer (Roald Bomans) using Raven Lite 2.0 for the 2017 operations (Center for Conservation Bioacoustics 2014). Both observers used Panasonic RP-HC200 headphones with the noise-cancelling function disabled.

### Silence

The amount of ‘silence’ in each recording was calculated to assess if bird communities in the treatment area “fell silent” after 1080 operations relative to non-treatment areas. If a ten second sub-sample contained no bird calling events, it was classified as silent. The prevalence of silent sub-samples was calculated in the same manner as the API of any other species.

### Species-specific bird song

Recordings were scored by simultaneously listening to recordings and visually inspecting their spectrograms. A call was scored as present if it could be both heard at maximum volume and seen on the spectrogram in a sub-sample. Where possible, calls were identified to one of 19 focal species known to inhabit the areas studied (Appendices S5–7). A call was classed as an ‘unknown’ if it could not be confidently identified to a species after listening to a sub-sample five times. Due to the species’ infrequent calling, kererū (*Hemiphaga novaeseelandiae*) were classified as present if their distinctive wingbeats were present in a sub-sample.

The calls of bellbird (*Anthornis melanura*) and tūī (*Prosthemadera novaeseelandiae*) are similar and may influence the reliability of results gained from acoustic identification alone (Mortimer & Greene 2017). To guard against possible misidentifications, tūī or bellbird calls were grouped for the 2014 operation and given one of three classifications for the 2017 operations: calls that could be confidently classified as bellbird or tūī were classified as their respective species, while calls that could not were assigned to the amalgamated “bellbird/tūī” taxon.

Blackbird (*Turdus merula*) and song thrush (*Turdus philomelos*) calls can also be similar and difficult to distinguish (Miller & Anderson 1992). As this study was primarily focused on native bird species’ responses, calls for these species were grouped for all operations (blackbird/thrush, i.e. *Turdus* spp.).

### Statistical analyses

For each 1080 operation, statistical analyses were carried out on the prevalence of silence and on the APIs of those species found to be both calling in at least 1% of all sub-samples analysed across all recordings, and present in at least half of both the treatment and non-treatment recording locations for the respective operation. This filter removed very rare and/or highly localised bird species from analyses.

A logistic mixed effects model was fitted with prevalence of silence or each species’ API as the dependent variable. Treatment (1080 or no 1080), time period (before or after 1080), and their interaction were included as fixed effects, and study site, recorder location, and recording date were included as random effects to control for potentially influential, yet

unmeasured, environmental factors that vary from place-to-place and day-to-day. Fitting was performed in R (R Core Team 2017) using the `glmer` function of the `lme4` (v1.1-18-11 for the 2017 operations; v1.1-23 for the 2014 operation) package (Bates et al. 2015), with a logit-link and the model formula:

$$\text{Species}_x \text{ API} \sim \text{Treatment} * \text{Time Period} + (1|\text{Study Site}/\text{Recorder Location}) + (1|\text{Date}) \quad (1)$$

where the response variable  $\text{Species}_x \text{ API}$  could also include the prevalence of *Silence*. Recorder location was nested within study site to account for the spatially nested structure of the sampling design (Figs 1b–d). For all models, the sample unit of an API value was weighted by 30, consistent with the number of binary sub-samples from which each sample was calculated.

The primary effect of interest was whether any changes in acoustic prevalence from before to after 1080 operations differed between treatment and non-treatment sites. This effect was tested for by the  $\text{Treatment} \times \text{Time Period}$  interaction term. A type III ANOVA was carried out on the resulting models in R using the `car` package (Fox & Weisberg 2019) to produce a chi-square value for the interaction term. However, given the non-independence of the sub-samples within each recording and the non-independence of successive recordings from the same site, we treated the standard errors and  $p$ -values from the `glmer` with suspicion. Instead, we applied a non-parametric permutation test to approximate the null chi-square distribution and test significance for each chi-square value observed. An advantage of permutation tests is that they make fewer assumptions about the distribution of the data and model residuals, although at the cost of reduced generalisability (Anderson 2001) and the inability to gauge effect sizes.

Permutations of observed data values (the APIs of each week by location) were made across time periods and within recorder locations. Any observations missing due to recording failure (NAs) were held constant in their location within the dataset. Five thousand permutations were made to reliably test significance to the 1% significance level. We also extracted the estimated marginal means (EMMs) and their standard errors from the fitted mixed effects (`glmer`) models for the control.before, control.after, treatment.before and treatment.after  $\text{Treatment} \times \text{Time Period}$  combinations of API and silence, expressed on the logit scale (Schwarz 2015). These values were used to generate interaction plots with 83.4% confidence intervals around each EMM to facilitate visual comparison of means from different treatment groups (Goldstein & Healy 1995).

Specifically, a decline in a species' API or an increase in silence in the treatment area relative to the non-treatment area would support the prediction that forests “fall silent” after an operation at the species or community level. To further visualise this, we calculated a BACI contrast of the silence and species' API data as:

$$\text{BACI.contrast} = (\text{logit}(\overline{\text{API}}_{\text{treatment.after}}) - \text{logit}(\overline{\text{API}}_{\text{treatment.before}})) - (\text{logit}(\overline{\text{API}}_{\text{control.after}}) - \text{logit}(\overline{\text{API}}_{\text{control.before}})) \quad (2)$$

where  $\text{API}$  could be substituted for the prevalence of *Silence*. An approximation of the 95% confidence interval for the BACI contrast was calculated via a bootstrap method (sampling with replacement), drawing from the set of observed values within each of the four  $\text{Treatment} \times \text{Time Period}$  levels, applying the above formula, and determining the 2.5th percentile, median and 97.5th percentile of the distributions of the 10 000 values generated. Negative values of this BACI contrast for species'

APIs indicate support for 1080 treatment reducing a species' calling prevalence, while positive values for silence indicate support for 1080 leading to increased overall silence after treatment in the treated area relative to the untreated area. Any instances of confidence intervals overlapping with zero are indicative of no significant interaction between treatment type and time period on the response variable.

The calculation of the BACI contrast could not account for spatial and temporal non-independence between samples. As such, greater weight was given to the results of permutation tests in the testing for the significance of the  $\text{Treatment} \times \text{Time Period}$  interaction. Nonetheless, the BACI contrast was calculated as a supportive means to gauge and interpret the approximate effect size and direction of any interaction between treatment and time on species' APIs or silence.

## Results

### Silence

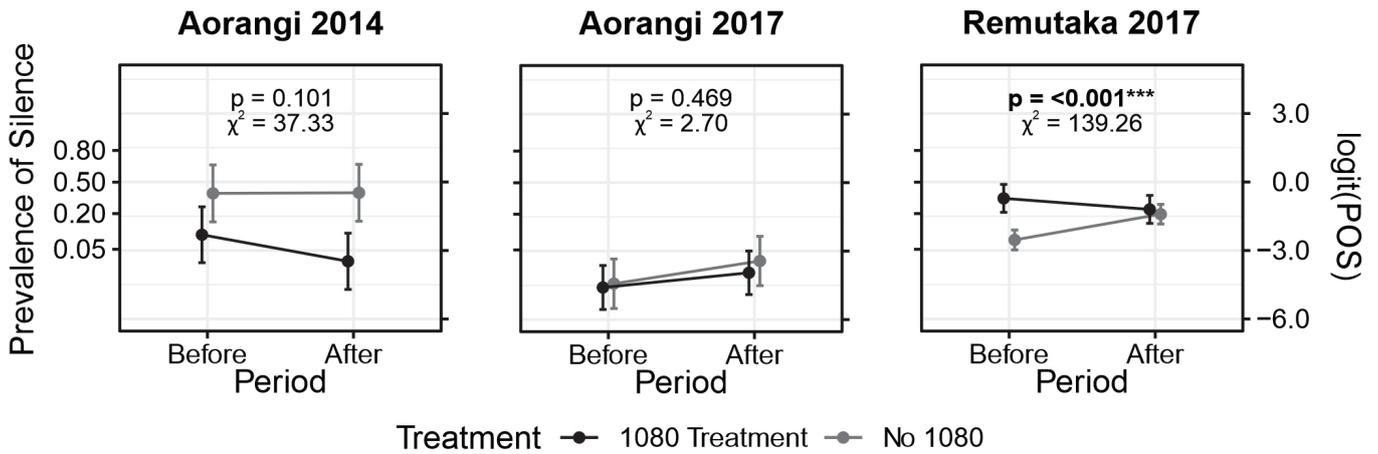
There was no evidence for an interaction between time and treatment on the amount of silence recorded across the 2014 Aorangi operation ( $\chi^2 = 37.33, p = 0.101$ ). The mean prevalence of silence decreased slightly in treatment sites, whereas non-treatment sites showed near identical levels of silence before and after treatment (a non-significant difference, Fig. 2). Across the 2017 Aorangi operation, both treatment and non-treatment sites showed a general increase in silence (Fig. 2), but again there was no significant interaction effect ( $\chi^2 = 2.70, p = 0.469$ ), suggesting that this was not an effect of the 1080 operation.

There was a significant interaction between treatment and time period on the prevalence of silence recorded across the 2017 Southern Remutaka operation ( $\chi^2 = 139.26, p < 0.001$ ). However, relative to pre-treatment levels, silence decreased (i.e. bird song became more prevalent) in the treated sites two to six weeks after the application of 1080. Specifically, the treatment site showed a decrease in the prevalence of silence from 0.328 [0.210, 0.474] (back-transformed EMM of prevalence of silence with an 83.4% confidence interval [lower, upper confidence limit]) before the operation to 0.231 [0.140, 0.357] after the operation, whereas non-treatment sites showed an increase in silence from 0.073 [0.049, 0.109] to 0.196 [0.137, 0.273] (Fig. 2).

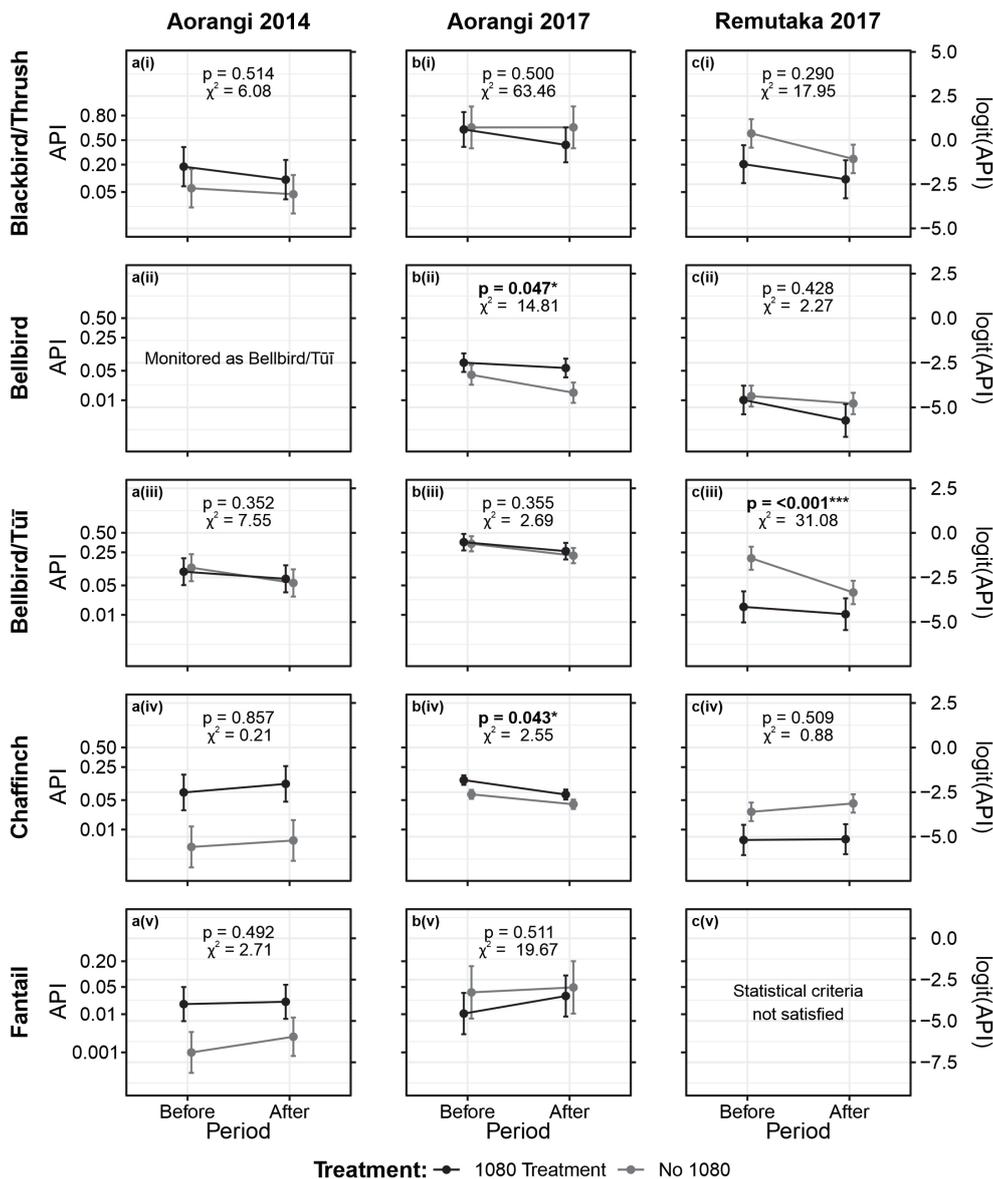
### Species-specific bird song

#### 2014 Aorangi operation

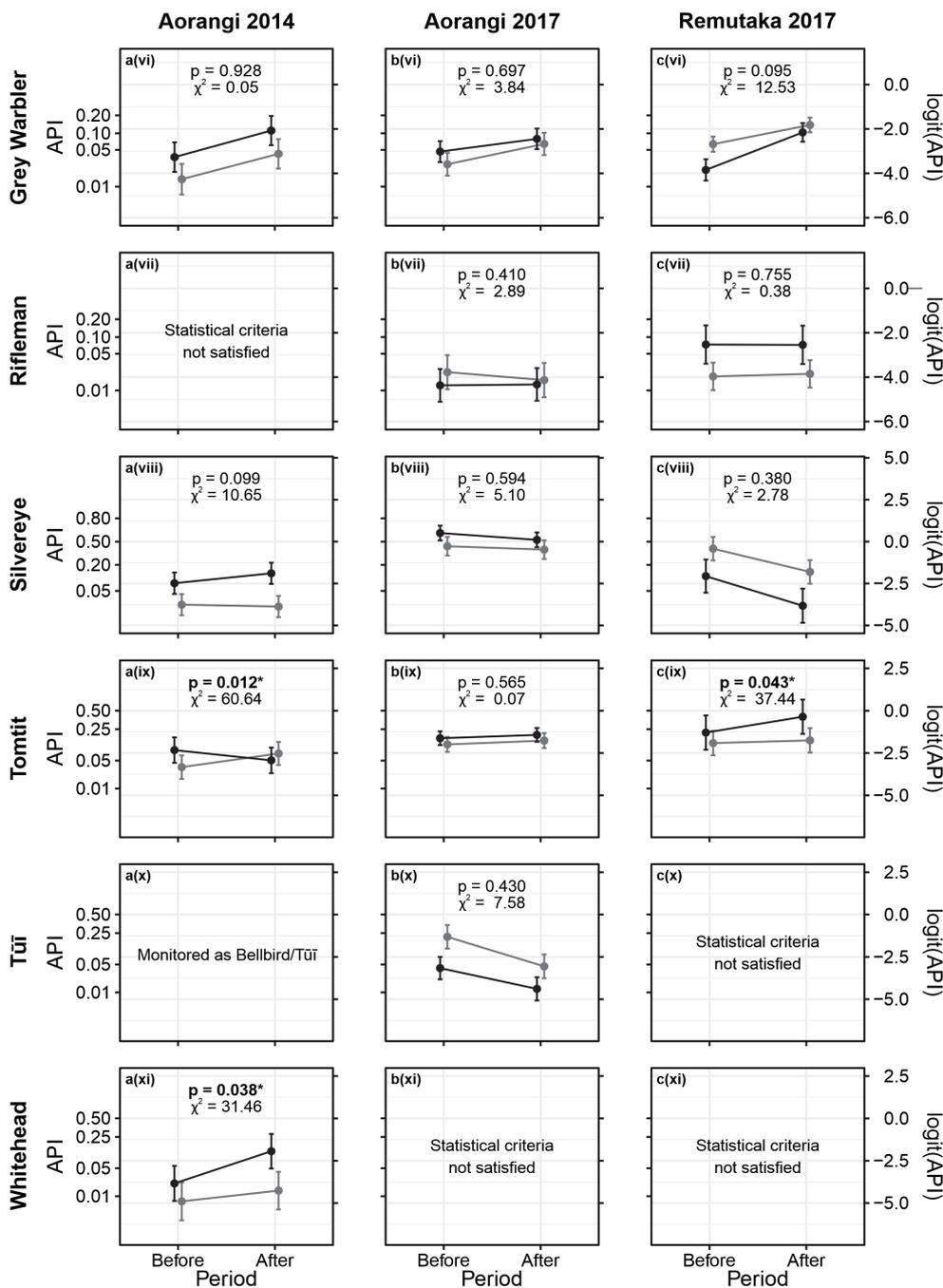
Nine taxa satisfied requirements for statistical analyses. However, only eight are presented here as models for rifleman data permutations suffered a high rate of fitting errors. Of these taxa, six showed no evidence for an interaction between time and treatment, while two species (tomtit and whitehead) did show a significant interaction at the 5% significance level, based on the permutation test. The mean tomtit API declined across time periods at treatment sites from 0.089 [0.044, 0.171] (back-transformed EMM of API with an 83.4% confidence interval [lower, upper confidence limit]) to 0.051 [0.024, 0.102], while the opposite trend was observed in non-treatment sites where mean API was higher after the 1080 drop (0.074 [0.039, 0.137]) than it was before (0.035 [0.018, 0.067]), (interaction effect:  $\chi^2 = 60.64, p = 0.012$ ; Fig. 3b-a(ix); Table 2). In contrast, the mean API of whitehead increased in both treatment and non-treatment sites from before to after the 1080 drop. However, this increase was greater in treatment



**Figure 2.** Interaction plots for the mean prevalence of silence (POS) with 83.4% confidence intervals (CI) in treatment and non-treatment areas, before and after three aerial 1080 operations. Means and CIs presented were calculated from the estimated marginal means of each operation’s fitted binomial model, accounting for the spatially nested and temporally replicated nature of the experimental design. Associated chi-square and *p*-values are derived from permutation tests. Asterisks (\*\*\*) indicate *p* < 0.001, plots share y-axes.



**Figure 3a.** Mean acoustic prevalence index (API) with 83.4% confidence intervals for species (A–F) monitored across three aerial 1080 operations. Means and CIs were calculated from the estimated marginal means of each species’ fitted binomial model, accounting for the spatially nested and temporally replicated nature of the experimental design. Associated *p*-values were derived from permutation tests. Columns and rows of plots share x and y axes respectively. Asterisks (\*, \*\*\*) indicate *p* < 0.05 and < 0.001 respectively.



**Figure 3b.** Mean acoustic prevalence index (API) with 83.4% confidence intervals for species (G–W) monitored across three aerial 1080 operations. Means and CIs were calculated from the estimated marginal means of each species' fitted binomial model, accounting for the spatially nested and temporally replicated nature of the experimental design. Associated  $p$ -values were derived from permutation tests. Columns and rows of plots share x and y axes respectively. Asterisks (\*, \*\*\*) indicate  $p < 0.05$  and  $< 0.001$  respectively.

sites (from 0.021 [0.008, 0.057] to 0.126 [0.049, 0.286]) than in non-treatment sites (from 0.007 [0.002, 0.022] to 0.014 [0.005, 0.041]) (interaction effect:  $\chi^2 = 31.46$ ,  $p = 0.038$ ; Fig. 3b-a(xi); Table 2). Fitting linear mixed effects models to the tomtit model with the original data resulted in a singularity warning, while fitting models to permutations of tomtit and grey warbler data resulted in singularity warnings for 100% and approximately 15% of permutations respectively, hence these results should be accepted with caution.

#### 2017 Aorangi operation

Eleven taxa satisfied prevalence requirements for statistical analyses. However, only ten are presented here as permutation models for kererū suffered a high rate of convergence failures ( $> 3\%$  of runs). Of the results presented, eight taxa showed a non-significant interaction between treatment and time period

at the 5% significance level (Fig. 3a,b-b(i, iii, v–x); Table 2).

A significant interaction between treatment type and time period was found for the mean API of both bellbird ( $\chi^2 = 14.81$ ,  $p = 0.047$ ) and chaffinch ( $\chi^2 = 2.55$ ,  $p = 0.043$ ). Mean API decreased from before to after 1080 treatment in both treatment and non-treatment sites for both of these species (Fig. 3a-b(ii, iv); Table 2). For bellbird, however, this decrease was more substantial in non-treatment sites (more than halving from 0.040 [0.023, 0.068] to 0.015 [0.009, 0.026]) when compared to treatment sites (from 0.076 [0.047, 0.122] to 0.058 [0.035, 0.094]). The opposite interaction held true for chaffinch, which showed a greater decrease in calls detected in treatment sites (API halving from 0.139 [0.110, 0.174] to 0.067 [0.052, 0.087]) compared to non-treatment sites (from 0.068 [0.053, 0.085] to 0.040 [0.031, 0.051]).

**Table 2.** Tabular summary of the *Treatment* × *Time* interactions presented in Figs 2, 3a,b and tested for significance via permutation tests. Where present, = indicates no significant interaction between treatment and time, implying no effect of 1080, contrary to the silent forest hypothesis (n = 22); ↑ indicates a significant interaction where the 1080 treated area increased or showed a lesser decrease relative to the non-treated area (n = 4); ↓ indicates a significant interaction where the 1080 treated area decreased or showed a lesser increase relative to the non-treated area (n = 3); \* indicates an interaction effect consistent with the silent forest hypothesis (n = 2); # indicates an interaction effect contrary to the silent forest hypothesis (n = 5); and NA indicates data available were insufficient to fit the model (n = 7).

Taxon	Aorangi 2014	Aorangi 2017	Southern Remutaka 2017
Silence	=	=	↓#
Blackbird/Thrush	=	=	=
Bellbird	NA	↑#	=
Bellbird/tūī	=	=	↑#
Chaffinch	=	↓*	=
Fantail	=	=	NA
Grey warbler	=	=	=
Rifleman	NA	=	=
Silvereye	=	=	=
Tomtit	↓*	=	↑#
Tūī	NA	=	NA
Whitehead	↑#	NA	NA

*2017 Southern Remutaka operation*

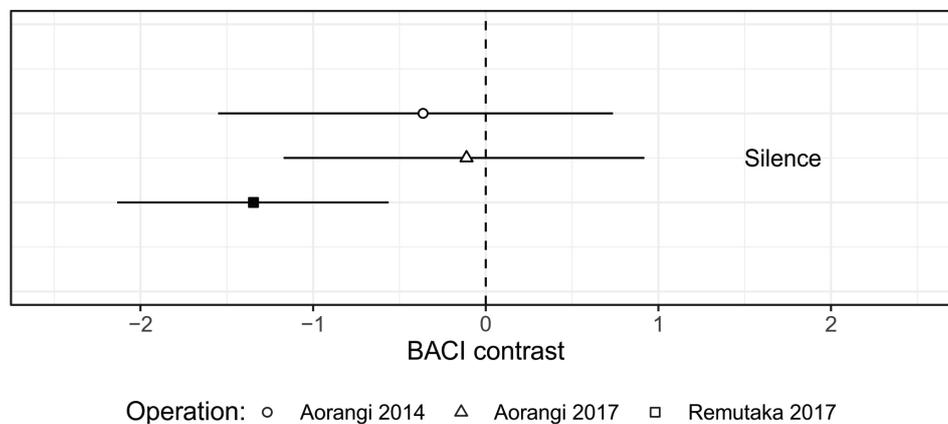
Nine taxa satisfied prevalence requirements for statistical analyses. However, only eight are presented here as permutation models for tūī suffered high rates of convergence failures. Of these eight taxa, six showed no significant interaction between treatment type and time period (Fig. 3a,b-c(i, ii, iv, vi–viii); Table 2).

A significant interaction was found for bellbird/tūī (*A. melanura/P. novaeseelandiae*) and tomtit (*P. macrocephala*). Bellbird/tūī showed a significant decrease in API in non-treatment sites relative to treatment sites ( $\chi^2 = 31.08, p < 0.001$ ), with the mean API in non-treatment sites decreasing from 0.195 [0.112, 0.316] to 0.034 [0.018, 0.064], while mean treatment site APIs decreased from 0.016 [0.007, 0.036] to 0.010 [0.004, 0.025] (Fig. 3a-c(iii); Table 2). In contrast, tomtit showed a significant increase in API in treatment sites relative to non-treatment sites ( $\chi^2 = 37.44, p = 0.043$ ): mean

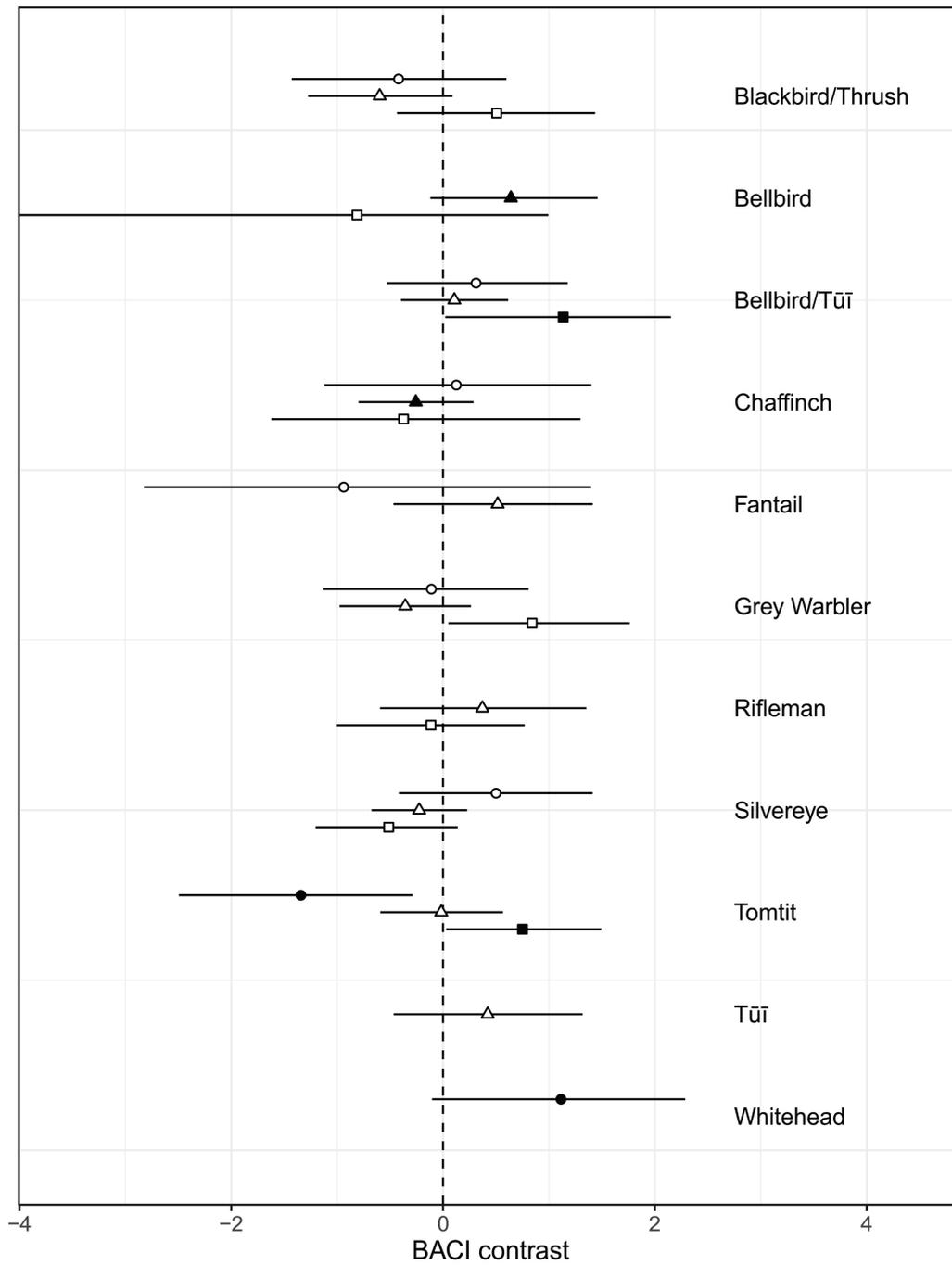
API nearly doubled from 0.215 [0.091, 0.431] to 0.411 [0.202, 0.658] in treatment sites, whereas mean API only increased slightly in non-treatment sites from 0.129 [0.067, 0.233] to 0.148 [0.078, 0.263] (Fig. 3b-c(ix); Table 2).

**BACI contrasts**

BACI contrasts broadly support the results described above, with silence and most species, in most operations, showing differences over time that were similar in both treatment and non-treatment sites, demonstrated by median BACI contrasts close to zero and 95% confidence intervals overlapping zero (Figs 4, 5). The only species to show a significantly negative BACI contrast was tomtit in the 2014 Aorangi operation. Responses of tomtit were highly variable as they also showed an apparent positive effect of 1080 after the 2017 Southern Remutaka operation (Fig. 5, Table 2).



**Figure 4.** Median BACI contrast values with 95% bootstrapped confidence intervals for the prevalence of silence across three aerial 1080 operations. Negative BACI values suggest that treatment areas saw either a decrease or a lesser increase in silence from before to after operations relative to the non-treatment areas (counter to the silent forest hypothesis), while positive values indicate the opposite interaction. Overlaps of confidence intervals with zero indicate a non-significant interaction. BACI contrasts were estimated from logit-transformed proportions and do not account for the spatially nested and temporally replicated nature of the experimental design; they should be interpreted with support of the interaction graphs and permutation tests. Filled symbols indicate interactions found to be significant in permutation tests.



Operation: ○ Aorangi 2014 △ Aorangi 2017 □ Remutaka 2017

**Figure 5.** Median BACI contrast values with 95% bootstrapped confidence intervals for the acoustic prevalence index (API) of species satisfying statistical criteria monitored across three aerial 1080 operations. Negative BACI values suggest that treatment areas saw either a decrease or a lesser increase in the API of respective species from before to after operations relative to the non-treatment areas (in line with the silent forest hypothesis), while positive values indicate the opposite interaction. Overlaps of confidence intervals with zero indicate non-significant interactions. BACI contrasts presented were estimated from logit-transformed proportions and do not account for the spatially nested and temporally replicated nature of the experimental design; they should be interpreted with support of the interaction graphs and permutation tests. Filled symbols indicate interactions found to be significant in permutation tests.

The boot-strapped 95% confidence intervals sometimes differed slightly from the permutation  $p$ -values in their indication of statistical significance. The negative effect of 1080 suggested for the acoustic prevalence of chaffinch in the 2014 Aorangi operation (according to the permutation test of the mixed effect model) was considered slight (relatively close to zero) and not deemed significant according to the calculation of the BACI contrasts. Confidence intervals also overlapped with zero for the apparent positive effect indicated by permutation tests of the mixed effects models for whitehead in the 2014 Aorangi operation and bellbird in the 2017 Aorangi operation. Interestingly, non-overlap of the confidence interval for the BACI contrast indicated a positive effect of treatment on the observed APIs of grey warbler across the 2017 Southern Remutaka operation. While permutation testing deemed the interaction between treatment and time period not to be significant at the 5% significance level for grey warbler across this operation, it was marginal at the 10% level ( $\chi^2 = 12.53$ ,  $p = 0.095$ ; Fig. 3b-c(vi); Fig. 5).

## Discussion

Ignoring differences between species, the results of monitoring silence across three separate aerial 1080 operations showed no support for the claim that forests fall silent following the aerial application of 1080 using cereal baits and non-toxic pre-feeds. Monitoring across the 2014 Aorangi operation showed a slight decrease in silence in treatment sites after the toxin drop that did not differ significantly from the change in non-treated sites, while monitoring across the 2017 Aorangi operation showed an increase in silence post-treatment that did not differ significantly between treated and non-treated sites. Across the 2017 Southern Remutaka operation, total birdsong actually increased in the treatment site following the operation whereas silence increased in non-treated sites, a result contrary to what would be expected in line with the silent forest prediction. These results are in line with mortality and bird count data analysed from 15 operations by Morriss et al. (2016), which collectively suggest that modern aerial 1080 operations pose a negligible direct threat to native forest bird communities.

At the taxon-level there was limited support for species falling silent. Monitored here across three operations, neither tūi nor silvereye, rifleman, grey warbler, or fantail showed API trends that differed significantly between treatment and non-treatment areas based on the permutation tests carried out. These results contrast with what would be expected if these species were suffering widespread poisoning. Rather, they support findings of recent studies suggesting that these native species do not suffer immediate negative impacts from modern aerial 1080 operations (Greene et al. 2013; Elliott & Kemp 2016; Morriss et al. 2016; van Vianen et al. 2018). An insignificant difference in API trends between treatment and non-treatment areas was also observed for the blackbird/song thrush taxon. This result suggests that modern 1080 operations do not have an immediate negative impact on populations of these species, contrary to findings of Morriss et al. (2016). Sourcing data from 15 operations occurring between 2003 and 2014, Morriss et al. (2016) suggest a probable negative effect of modern aerial 1080 operations on blackbird populations, with blackbirds accounting for 55 of 81 carcasses found in searches after the operations, and showing a significant decline in 5MBCs in treatment areas relative to non-treatment areas

from before to directly after two operations in Hauhungaroa. It is important to note that, given the confounding variations that are likely to exist in the diets, behavioural ecology and thus exposure risks and detection probabilities for blackbirds and song thrush, the inferences that can be drawn for these two species may be limited by the grouping of their calls in the current study.

Only two species, chaffinch and tomtit, exhibited a significant decline in API across treatment locations relative to non-treatment locations consistent with the silent forest theory (based on the permutation test). For both species, this decline was observed for only one of the three operations over which the species were monitored. Chaffinch are an introduced, granivorous species that are likely to interact with cereal baits (Miller & Anderson 1992). Morriss et al. (2016) report eight of 81 bird corpses found across 15 operations to have been chaffinches; five of these were analysed for 1080 residues, four of which tested positive. During our own field work in the Aorangi Forest Park in July 2017 we incidentally observed a total of three bird carcasses: two chaffinches and one blackbird, in addition to three dead possums, three hedgehogs, two rats, and one cat. Thus, it is possible that the decline in the acoustic prevalence of chaffinch observed across the 2017 Aorangi operation is a case of a non-target impact resulting from the operation. The fact that this result was not observed consistently across all three treatments may stem from factors affecting the longevity and exposure rate of toxic baits that vary from one operation to another, such as differing sowing methods, differing sowing rates (Westbrooke et al. 2003), and patterns of rainfall received in treatment areas following operations (Booth et al. 1999; Bomans 2018).

The single tomtit response consistent with the silent forest theory was observed across the 2014 Aorangi operation. Tomtit have been known to suffer mortalities over aerial 1080 operations where carrot baits were used (Spurr & Powlesland 1997; Powlesland et al. 1998; Spurr & Powlesland 2000; Morriss et al. 2016). However, more recent studies suggest that modern operations employing cereal baits and lower sowing rates have little, if any, negative impact on tomtit populations (Westbrooke et al. 2003; Westbrooke & Powlesland 2005; Greene et al. 2013; Morriss et al. 2016; van Vianen et al. 2018). The subsequent non-significant difference in calling patterns observed across the 2017 Aorangi operation suggests that, if reliable and reflective of a population loss, the pattern observed across the 2014 operation is not always strong or consistent. With a high reproductive rate, tomtit have been classed as a resilient species with a good potential to recover from any short-term population losses resulting from predator-control operations (Spurr 1979; Spurr & Powlesland 2000). In particular, the species has high reproductive success under low-predator conditions and has been shown to benefit in years following aerial 1080 treatment (Knegtman & Powlesland 1999; Peterson 2014; van Vianen et al. 2018).

The significant increase in tomtit calling observed in treatment areas across the 2017 Southern Remutaka operation occurred over too short of a timeframe to reflect a benefit from recruitment resulting from treatment (Atkinson et al. 1995). Higher calling rates could reflect a behavioural response of increasing boldness when birds perceive a lower risk of predation (Lima 2009). Alternatively, increases in calling rate may result from population losses in territorial species causing remaining individuals to call more in order to establish new pair bonds and/or territories (Spurr & Powlesland 2000). As tomtit are a highly territorial species year-round (Knegtman &

Powlesland 1999; Spurr & Powlesland 2000), it is possible that the observed increase in calling might also indicate a negative effect of aerial 1080 treatment on the species' population. Alternatively, this result may reflect a topographical imbalance in the study design across this operation. Due to final drop boundaries excluding an area identified *a priori* as being within the treatment area, all treatment locations that were monitored were at one site above 500 m elevation, whereas non-treatment recorders were located in one site above 480 m and one site below 150 m elevation. Tomtit breeding behaviour can be affected by elevation, with nesting starting as much as a month later at higher elevations (520–740 m) than at lower elevations (0–200 m) (Knegtmans & Powlesland 1999). Any changes in API associated with the onset of the breeding season are therefore likely to be delayed at higher elevations. Accordingly, over the time-period monitored for this operation (June–July), treatment sites may have been experiencing a delayed seasonal increase in calling prevalence normally seen earlier at lower elevations (Michaux 2009). This result highlights the importance of balancing topography (Spurr & Powlesland 2000) and understanding species' behaviour and ecology when monitoring species' conspicuousness as an index of relative abundance, and the need to interpret the results of all monitoring across this operation with caution given the potential for this elevational influence. In sum, these results collectively suggest little risk of a net-negative long-term effect of aerial 1080 operations on tomtit populations in the Aorangi and Southern Remutaka Ranges. This is in line with a meta-analysis by Fea et al. (In Press), which suggests a net neutral response of the species to forms of low intensity mammal control such as pulsed 1080 operations. Nevertheless, taking a precautionary approach, we suggest that further monitoring of tomtit exposure and responses to 1080 under a variety of conditions may be warranted.

As with tomtit, the increase in API exhibited by whitehead in treatment areas across the 2014 Aorangi operation cannot reflect an increase in the species' abundance resulting from the operation. Given their gregarious and cooperative breeding nature (Gill & McLean 1992; Heather & Robertson 2005), it is also unlikely that this increase is territory-related and reflects a negative effect of the operation on whitehead populations. An increase in acoustic prevalence might occur without any change in population size if birds perceive a decrease in predator presence following treatment and thus a reduced risk in predation from calling behaviour (Lima 2009), or as a result of factors unrelated to pest control treatment. Unfortunately, due to the low prevalence of whitehead in the non-treatment areas monitored across both 2017 operations, the consistency of this pattern across independent aerial 1080 treatments could not be determined in this study. Whitehead was one of the species identified by Fea et al. (In Press) as showing a significant population increase in response to low intensity mammal control and a significant decrease when mammals were not controlled.

Considering that no bellbird deaths have been reported following aerial 1080 operations (Spurr & Powlesland 1997; Morriss et al. 2016), the lesser decline in API observed for bellbird in treatment areas across the 2017 Aorangi operation is unlikely to reflect changes in conspicuousness associated with significant mortality events. We consider it more likely that the trends observed may be associated with population movements following spatio-temporal changes in food availability. Where nectar is available, bellbirds are a primarily nectarivorous species and have been noted to move large

distances to patchy, ephemeral nectar resources (Rasch & Craig 1988; Spurr et al. 2010; Spurr et al. 2011). Such seasonal movements have the potential to disproportionately affect detection probabilities (Pierce & Montgomery 1992), and might lead to differential detection probabilities across study areas where nectar resources are unevenly distributed. Knowledge of the foraging patterns and seasonally related movements of bellbirds is limited and requires further study (Spurr et al. 2010). Such studies would aid the interpretation of results for future BACI studies monitoring bellbird populations around pest control operations.

Nectar-related movements might also explain the lesser decline in API observed for the bellbird/tūi taxon in treatment areas relative to non-treatment areas across the 2017 Southern Remutaka operation. Firstly, it is important to acknowledge that, being the amalgamation of two species, these data should be interpreted with caution. Nonetheless, both bellbird and tūi are primarily nectivorous species known to move large distances to reach nectar resources (Stewart 1980 as cited by Craig et al. 1981; Bergquist 1985; Rasch & Craig 1988; Bergquist 1989; Pierce & Montgomery 1992). Furthermore, only two tūi have been found dead after aerial 1080 treatments: one after a carrot bait operation, and one after a modern cereal bait operation (Spurr & Powlesland 1997; Morriss et al. 2016). Thus, we propose that changes in the detection probabilities of both species are more likely affected by patchily available nectar resources than exposure to 1080. The potential for seasonal movements to affect detection probabilities should be carefully considered in acoustic monitoring for all species that may exhibit such behaviours.

While the current consensus largely suggests a non-negative immediate impact of aerial 1080 operations on native bird species, continued monitoring of species over future operations is recommended to further inform conservation management and to address remaining public concerns as operations continue and change in the future. Veltman and Westbrooke (2011) advocate for long-term monitoring of forest bird species across sites treated with 1080 to facilitate the quantitative assessment of their impacts. BACI designs are often recommended for their ability to tease apart treatment effects from background temporal variability; having multiple treatment and multiple control sites is an even better design (Underwood 1992). While such temporal and spatial replication would be difficult with more labour-intensive traditional monitoring methods, the spatial and temporal replication and comparability of data achievable with bioacoustic monitoring may serve to greatly facilitate such monitoring. Nonetheless, monitoring with ARUs is limited in its capacity to monitor rarer species and species significantly affected by seasonal behaviours; hence more intensive, capture-mark-recapture monitoring methods might be better suited to the monitoring of these species. Employed together, bioacoustic and capture-mark-recapture monitoring methods have the capacity to provide significant integrated insights to the short- and long-term effects of aerial 1080 operations on native bird species.

Analysing data with spatial non-independence and temporal non-independence can be challenging as these sampling designs violate the assumptions of many classical statistical models. Here, we used permutation tests to obtain *p*-values for the interaction effect, and bootstrap resampling to obtain confidence intervals for the BACI contrast (our measure of effect size) as these methods use the empirical data to derive the expected distribution of parameter values. The two methods did not always agree exactly on which effects were significant

and which were not, as the permutation retained more of the spatial structure of the data captured by the random effects, whereas the BACI contrast bootstrap divided the data into four groups (Before-vs-After × Treated-vs-Control) to resample from (see Appendices S2–S4). Nevertheless, we believe it is valuable to present the results from both approaches as where they do agree this provides greater confidence in the result not being dependent on specific decisions of how to model the data, and where they disagree this highlights that further work may be required.

In summary, this study sought to apply bioacoustic monitoring to address the claim that forests fall silent after 1080 operations and found little evidence for overall short-term negative effects on native bird communities. These results corroborate studies of modern aerial 1080 operations, which suggest a negligible threat from operations at the community level, and little threat at the individual level for native forest bird species (Greene et al. 2013; Morriss et al. 2016; van Vianen et al. 2018). Additional research using radio-tracked birds is recommended for species that might be considered vulnerable, especially those too rare to be sufficiently sampled by bioacoustic methods.

## Author contributions

RB, AC and SH co-designed the project. RB and AC conducted the majority of field work and scored the sound files. All authors contributed to analysis, interpretation and writing.

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## Supplementary material

Additional supporting information may be found in the supplementary material file for this article:

**Appendix S1.** Table of information for weather stations used in sample day selection across the Aorangi 2014, Aorangi 2017 and Southern Remutaka 2017 operations monitored.

**Appendix S2.** Dates and recorder locations sampled in treated and untreated areas before and after the Aorangi 2014 operation.

**Appendix S3.** Dates and recorder locations sampled in treated and untreated areas before and after the Aorangi 2017 operation.

**Appendix S4.** Dates and recorder locations sampled in treated and untreated areas before and after the Southern Remutaka 2017 operation.

**Appendix S5.** Summary statistics of raw API data for the species monitored across the Aorangi 2014 operation in treatment and non-treatment areas.

**Appendix S6.** Summary statistics of raw API data for the species monitored across the Aorangi 2017 operation in treatment and non-treatment areas.

**Appendix S7.** Summary statistics of raw API data for the species monitored across the Southern Remutaka 2017 operation in treatment and non-treatment areas.

The New Zealand Journal of Ecology provides supporting information supplied by the authors where this may assist readers. Such materials are peer-reviewed and copy-edited but any issues relating to this information (other than missing files) should be addressed to the authors.