Time-Sensitive Prioritization of Island Disease Eradication Strategies: A One Health and Public Health Intervention Framework

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Data availability

Additional supporting information may be found in the supplementary material of this article. Code and data will be archived in a publically available digital repository.

Abstract

Achieving conservation objectives is time-critical, but the vast number of threats and potential actions means some form of ranking is necessary to aid prioritization. Objective methods for ranking conservation actions based on when they are differentially likely to become feasible, or to succeed, are currently unavailable within existing decision-making frameworks but are critical for making informed management decisions. We demonstrate how statistical tools developed for survival (or time-to-event) analysis can be used to rank conservation actions over time, through the lens of invasive mammal eradications on islands. Here, we forecast the probability of eradicating commensal rat species (*Rattus rattus, R. norvegicus, R. exulans*) from the New Zealand archipelago by the government's stated target of year 2050. Our methods provide temporally ranked eradication trajectories for the entire country, thus facilitating meeting nationwide policy goals. This demonstration highlights the relevance and applicability of such an approach and its utility for prioritizing globally effective conservation actions.

Introduction

Conservation is typically considered as triage (Wilson, McBride, Bode, & Possingham, 2006) and many global actions must be implemented swiftly in order to achieve desired outcomes, such as intervening in species extinction trajectories (Butchart et al., 2010). However, the abundance of urgent actions means that not all can be undertaken simultaneously, if ever (Brooks et al., 2006; Myers, Mittermeier, Mittermeier, Da Fonseca, & Kent, 2000). As a result, conservation actions must be ranked based on how soon they are likely to become feasible, and to succeed, alongside other factors such as their biodiversity outcomes and financial costs. Examples include which undescribed species to focus on taxonomically describing, and which proposed protected areas are more likely to become gazetted. Existing conservation-prioritization frameworks have yet to implement an objective method addressing this temporal issue, representing a significant gap in management planning. Island ecosystems, in particular, are in dire need of temporal prioritization (Jones et al., 2016); endemic island biota are being disproportionately driven to extinction by invasive species at an alarming rate worldwide (Blackburn, Cassey, Duncan, Evans, & Gaston, 2004).

Systematic eradication techniques were developed in the early 1980s to stem the negative impacts of predatory invasive mammals, and to restore island communities (Howald et al., 2007). Projects to eradicate mammals have increased demonstrably in size, scope, and complexity over the past four decades, and are now a principal conservation intervention (Brooke, Hilton, & Martins, 2007; Russell & Broome, 2016). To date, terrestrial vertebrates have been eradicated from >1,000 islands (Holmes, Keitt, et al., 2019), resulting in significant biodiversity conservation benefits (Jones et al., 2016). However, impacts from invasive species continue to accelerate, and resources available for eradication are limited. Therefore, it is imperative to prioritize eradication projects by how soon they can be implemented, or how soon they can succeed, so that achievable conservation goals can be pursued.

The formal process for setting eradication priorities is well documented (Brooke et al., 2007; Harris, Gregory, Bull, & Courchamp, 2012; Helmstedt et al., 2016; Holmes et al., 2015; Joseph, Maloney, & Possingham, 2009; Wilson et al., 2006). Most frameworks evaluate expected conservation benefits, the static likelihood of project success and financial constraints to assist in maximizing eradication benefits under a fixed and limited budget (Howald et al., 2007; Oppel, Beaven, Bolton, Vickery, & Bodey, 2011). However, the number and types of islands that are potential eradication targets is constantly increasing due to ever-advancing eradication technologies and scientific understanding. Therefore, the likelihood of an invaded island becoming an eradication target also has a temporal component; some islands will become feasible targets sooner than others (Holmes, Spatz, et al., 2019) and certain projects can be completed faster than others. Failing to consider time to project success as a component of the ranking and prioritizing framework limits the relevance of proposed decisions, and/or could result in fallacious expectations. To ensure the outcome of a given prioritization exercise, conservation managers must inform the decision-making process with adequate understanding of the time to success. Due to a lack of objective methods, contemporary eradication exercises have yet to include a temporal component within their ranking framework (although see Holmes, Spatz, et al., 2019 for a first attempt using expert elicitation). This gap in the existing prioritizing framework represents a major limitation to scaling-up eradication projects globally to take on challenges of increased complexity, such as invasive species management at large regional and national scales (Kopf et al., 2017).

Here, we rank an important conservation objective by considering the time to eradication success for islands invaded by mammalian predators. Due to the challenges associated with determining when a proposed project will succeed if implemented, we use the record of past eradications to model factors correlating with the time to success. To that end, we use tools developed for survival analysis (Hougaard, 2012; Therneau, 2015) to: (*i*) examine factors hypothesized to have influenced the time to eradication success in the past, (*ii*) use those same factors to predict the time to eradication success in the future, and (*iii*) assess the validity of our prioritizing model using a retrospective analysis. We frame our article using the example of invasive rats in the New Zealand archipelago (Figure 1). Commensal rat species, the ship rat (*Rattus rattus*), Norway rat (*R. norvegicus*), and Pacific rat (*R. exulans*) present in New Zealand, constitute a grave threat to endemic island biota globally. These rat species have colonized 85% of the world's archipelagos (Atkinson, 1985) and have caused the extinction of at least 40 different species (Towns, Atkinson, & Daugherty, 2006). New Zealand is the world leader in undertaking ambitious predator eradication projects (completing 26% of all successful island eradication projects globally) and holds complete data on the distribution and eradication of rats on its islands (Russell & Broome, 2016). Together these factors make for an extremely well suited case study to develop and demonstrate survival methods in a conservation-intervention context.

Methods

i. Case study location and invasion dataset

The New Zealand archipelago (latitude: 29.3° S to 52.5°S, longitude: 166.1°E to 175.9°W; Figure 1) is comprised of many offshore islands that are mostly clustered in four distinct regions, including The Hauraki Gulf, Marlborough Sounds, Fiordland, and Stewart Island (Figure 1a). Multiple outlying archipelagos are part of New Zealand, but rats have invaded only three of these (the Kermadec Islands, Chatham Islands, and Campbell Island Groups; Figure 1b-d). In total, 154 islands (≥ 5 ha area) have been invaded by rats (comprising 77.6 % of all island land-area ≥ 5 ha). 'Mainland' islands (including the North and South Islands, Great Barrier Island, and Stewart Island; Figure 1a) were excluded because they require eradication tools superseding those used in this study (Russell, Taylor, & Aley, 2018).

New Zealand began systematically eradicating rodents in the early 1980s using second-generation anticoagulant toxins (Russell & Broome, 2016). As of 2020, rats have been eradicated from 80 of 154 rat-invaded islands (i.e. just over half). Moreover, New Zealand has set a precedence in resolving to extirpate rats from the entirety of its borders by 2050, as part of its Predator Free 2050 initiative (PF2050; Russell et al. 2015). We thus used this record of successful eradication projects (years 1980 - 2020, n = 80) to inform predictions for remaining invaded islands (n = 74) to facilitate meeting PF2050 objectives. This dataset was compiled as part of an update to the Handbook of New Zealand Mammals (King, 1990) and is an exhaustive account of New Zealand's documented insular rat invasion. See Appendix S1 for complete dataset details.

ii. Drivers of eradication success

The primary mechanisms influencing eradication success on New Zealand's rat-invaded islands include reinvasion probability, and obtaining socio-political support from vested stakeholders (Holmes, Spatz, et al., 2019; Russell et al., 2015). We did not consider project funding as a limiting factor following technical feasibility within our model. Funding availability is paramount to successfully implementing conservation projects globally (Waldron et al., 2013), but this has not been the case historically in New Zealand where technical feasibility has been the limiting factor. New Zealand is widely regarded as an aggressive adopter of island eradications as a form of conservation intervention. As such, the country has moved in lock step with emerging technological, social and scientific advances to undertake eradications of increased size and complexity as they became available (Russell & Broome, 2016; Towns, West, & Broome, 2013). The result is that prospective projects were funded relatively promptly after they were deemed feasible and were identified to be a conservation priority (Broome, 2009). Indeed, the near-exponential increase in eradicated island area from 1980 to mid-2010 is a testament to this willingness and is evidence that funding has not been limiting overall (c.f. Russell and Broome (2016) Figure 1 and Towns et al. (2013) Figure 3). However, we acknowledge the uniqueness of New Zealand's circumstances and the global relevance of project funding herein.

Reinvasion represents the greatest threat to long-term eradication success (Harris et al., 2012). Rodents readily hitchhike to landmasses as seafaring stowaways, and disperse naturally by swimming between islands (Russell & Clout, 2004). Reinvasion risk is successfully explained by measuring insular isolation (Carter, Perry, & Russell, 2020); islands surrounded by larger quantities of seascape are generally better protected from recolonization by terrestrial mammals, and so have a high probability of eradication success relative to other landmasses. We quantified each island's level of geographical isolation to inform our model (Table 1).

Islands with human inhabitants, or with vested interests other than conservation, represent globally limiting factors to increases in eradication size and complexity (Oppel et al., 2011). Given the typical size of a rat's home range (< 1 ha), even a single plot of private property becomes a potential refuge for survivors (Russell, Towns, & Clout, 2008). Therefore, eradication requires full compliance of the local community. However, acquiring compliance is a nontrivial task; the ecological and social benefits of eradication must outweigh incurred costs, and they must be accurately and effectively conveyed to the community (Oppel et al., 2011). To confound this issue, agreeing on common conservation goals becomes increasingly difficult as the number of human inhabitants increases, due to subjectivity in personal values and the potential for disagreement and conflict (Shanahan, Ledington, & Maseyk, 2018). Therefore, the probability of candidate island selection for eradication is inversely correlated with human inhabitation. We captured this aspect of attaining socio-political support for eradication by collating land-tenure metrics for each island, including the type of vested interest(s) (e.g., public, private, mixed tenures), and by quantifying the number of documented stakeholders (Table 1).

We obtained measures of insular isolation (Table 1) following Carter et al. (2020). This approach synthesized many commonly used isolation metrics in to parsimonious factors, including Distance, Insular Network (Area), Stepping Stones, and Landscape Isolation (Buffer). We modelled the effect of human-mediated transport (Landing) on island isolation following Russell and Clout (2004) by considering the presence of landing structures, including wharfs and airfields, as potential reinvasion sources. All such variables were calculated in ArcGIS 10.5.1 (ESRI, 2011). Distance, Area, and Buffer measures were calculated using the standard ArcGIS toolset. Stepping Stones required application of least-cost methodology; we assumed minimal traversal through open-water and sheer cliff faces (Russell et al., 2008). A detailed overview of our least-cost methods can be found in Carter et al. 2020. We collated the location of different landing structures using the NZ Wharf Edges (LINZ Data Service, 2019) and Airport/Airfield Polygon (LINZ Data Service, 2020) layers provided by Land Information New Zealand Data Service (LINZ). GIS landmass polygons were also provided by LINZ (LINZ Data Service, 2018b). Continuous variables were standardized to zero mean and unit variance to conform to statistical assumptions and were transformed ($\log_{10}$ or cubed-root) to remove skew (Table 1).

We collated land-tenure metrics using GIS layers provided by the NZ Primary Parcels Dataset (LINZ Data Service, 2018a), and the Māori Land Spatial Dataset (Maori Land Court, 2017). We designated an islands’ vested interest(s) as including any of Public, Private, and Māori (Table 1). Public islands were those with land owned or administered by New Zealand’s Crown estate, Private islands were those with land held as non-government fee simple properties (including those listed under leasehold, unit title, and cross lease), and Māori islands were those with land identified under *Te Ture Whenua Māori Act* 1993. Islands with mixed land ownership were included in our prioritizing model using interaction terms. We distinguished between Private and Māori land tenures due to the recognized differences in social organization and political composition, which have an effect on wildlife management practices (Russell et al., 2018). We converted the number of stakeholders (Private or Māori owners) per island in to categorical variables. Finally, we included an interaction term to account for correlation between insular land area and the number of private stakeholders ($r_s = .29$, $p < .01$). Doing so allowed us to extricate potential effects of human inhabitation on eradication success at increasing geographic scales, should they be present in our dataset.

iii. Predicting the time to eradication success

We used survival analysis, also referred to as 'time-to-event' analysis, for making temporal predictions and rankings. The rankings obtained can then be interpreted as prioritizations over time. Survival analysis models the time required for an event of interest to occur for an individual, and the factors influencing the occurrence of those events (Hosmer, Lemeshow, & May, 2008). Survival analysis methods resemble the standard regression-modelling paradigm where statistical hypotheses are tested concerning the relationship of predictors to an outcome variable (Hosmer et al., 2008). Here, the outcome dependent variable is 'survival time' - any real and positive value with a continuous distribution (Hougaard, 2012). An 'event' can be any discrete and measurable occurrence capable of happening to an individual, and an 'individual' refers to any entity capable of experiencing the event. We extend the survival framework to investigate the survival time for implementing rat eradication projects on islands. Here, our event of interest is the eradication of rats. An eradication occurs when all rat species present are documented as being purposefully extirpated from a particular island (constituting a successful project). The individual for which the event occurs is the island hosting rats (not the rats themselves), and the summation of these islands is the population of individuals for which we measure survival time. Therefore, survival time in this instance is defined as the time to the occurrence (implementation followed by successful conclusion) of an eradication for a rat-invaded island.

An important concept of survival analysis is censorship; censorship is used to interpret the occurrence of events that have not been directly observed (Hosmer et al., 2008). An individual becomes censored when monitoring stops and the event of interest has yet to occur. Censorship enables inclusion of that individual for the period of time they are known to be present for, even if the event never occurs. We considered islands (i.e., individuals) who have had rats eradicated (i.e., experienced the event of interest) sometime between 1980 and 2020 to be uncensored because the eradication year is known. Islands that have yet to have rats eradicated are censored because the eradication date is known only to exceed the timeframe of the study (known as ‘right censorship’). We used the island’s most-recent eradication status, taken at the time of the study, to designate censorship (however, reinvasion events were considered in our assessment of model validity). Other instances of censorship, including extinction of rat populations, and unknown eradication dates (i.e., interval censorship) did not occur. A single occurrence of left censorship, where the eradication date is known to proceed the start of the study, was removed (Titi Island: 40.95°S, 174.14°E).

From our survival data, we estimated the survival function using the Kaplan-Meier (or product-limit) estimator (Kaplan & Meier, 1958). This function, given as $S(t) = P(T > t)$, estimates the probability P of observing a survival time T that exceeds some specified time t. The Kaplan-Meier estimator includes information from all individuals of the given population (regardless of censorship) by considering survival at any point in time as a series of steps defined at the observed survival times (Hosmer et al., 2008). An estimate of the overall survival function for the population is then provided by multiplying the conditional probability of known survival at each observed survival time. A survival curve represents the survival function graphically and shows cumulative survival over time. We used the survival curve as the fundamental building block of our time-based predictions. In the context of our study, $S(t)$ represents the cumulative probability that a rat-invaded island remains un-eradicated (or rat-invaded) beyond time t. We estimated the survival function using the ‘survival’ package (Therneau, 2015) in R 3.4.1 (R Core Team, 2017). See Appendix S2 for complete details regarding estimation of $S(t)$.

With our survival curve, we were able to predict the time to eradication success. We provide an overview herein; for complete details, see Appendix S2. We first (*i*) used maximum likelihood estimation to fit relevant right-skewed parametric distributions to our survival curve (Groeneboom & Wellner, 1992). A single distribution was then selected by maximizing the likelihood function taking in to account right censorship. We used a parametric bootstrap estimate of standard error, based on 100 bootstrap replications (Efron & Tibshirani, 1993), to generate 95% confidence interval estimates (95% CIE). Next, we (*ii*) used information-theoretic techniques to select an appropriate candidate model for our distribution (Burnham & Anderson, 2002). We used Akaike’s information criterion (AIC_C, small sample version) to select one model from among a suite of candidates; plausible candidates had $\Delta AIC_C \leq 2.0$ relative to the best fitting model. Finally, we (*iii*) predicted the time to eradication success for invaded islands using our selected candidate model. All statistical analyses were performed in R 3.4.1. Probability density functions were provided using the ‘actuar’ (Dutang, Goulet, & Pigeon, 2008), ‘flexsurv’ (Jackson, 2016) and ‘invGauss’ (Gjessing, 2015) packages, or were introduced to the R environment manually. All distributions were fitted using the ‘flexsurv’ package. Candidate model combinations were generated using Python 2.7.16 (Python Software Foundation, 2019) and evaluated in R using a high performance computer (HPC) from the New Zealand eScience Infrastructure.

We frame our temporal predictions by investigating PF2050’s goals and challenges using the survival context. We made three separate predictions in total – two for individual islands (i.e.

the ‘individuals’) and one for the New Zealand archipelago (i.e. the ‘population’ of islands). We make this distinction apparent because prediction methodologies differ slightly between scales. For individuals, survival outcomes are interpreted in the same manner as other regression extensions predicting for dichotomous outcomes – the dependent variable is a continuous probability lying within a spectrum (namely logistic regression; see Bischof et al. (2012) or Conner et al. (2018) for examples). Therefore, we selected an *a priori* probability value with which to deem an eradication project ‘successful’; we assumed rats can be assuredly eradicated from an island when its cumulative eradication probability exceeds 80%. Eighty-percent constitutes an appropriate and conservative benchmark because it is the current success rate of rat eradications on tropical islands (i.e. the projects with the lowest success rate globally) (Holmes et al., 2015). Thus, an island is predicted to be a candidate for successful eradication when its survival probability falls below 20%, although not without what is considered by practitioners an acceptable risk of failure. For population-level predictions, estimates are based upon the cumulative proportion of eradication successes across all islands through time. This prediction simply documents New Zealand’s projected eradication trajectory instead of relying on *a priori* assumptions.

For our first individual-level prediction, we determined the feasibility of extirpating rats from each of the 18 invaded island reserves (Table S2.1) (Parkes, Byrom, & Edge, 2017) by 2025 by calculating eradication probabilities for these islands at this interim deadline (interim PF2050 objective 3.3) (Cabinet New Zealand, 2016). For our second individual-level prediction, we determined the probability of extirpating rats from each invaded island by 2050. We also predicted the expected eradication year and identified the factor(s) that project success is most dependent upon for these islands. To determine these factor(s), we considered each variable of our selected candidate model in turn as irrelevant to eradication success. Holding all other variables constant, we minimized the effect of each single variable increasing survival probability, or maximized the effect of single variables decreasing survival probability. The variable(s) with the largest influence on survival time was then identified for each island. Finally for our population-level prediction, we determined the feasibility of extirpating rats from all islands by 2050 (coinciding with PF2050 objective 1) (Cabinet New Zealand, 2016; Sage, 2020). In short, these predictions determine whether an individual invaded island of interest is likely to be rat-free by 2025 (if applicable) or 2050 and whether the current rate of conservation advances is sufficient for achieving PF2050 as a whole.

iv. Assessing model validity

We conducted two retrospective tests to assess model validity. We first (*i*) investigated the effect that rat reinvasion might have had on PF2050's proposed timeline by comparing the eradication probability of eradicated islands that previously experienced reinvasion following eradication with those that have not experienced reinvasion. Conceptualizing the eradication process as a single-event simplifies the true nature of survival for an invaded island. Indeed, a rat-free island is continually susceptible to reinvasion and exclusion can never be guaranteed to last in perpetuity (Harris et al., 2012). Therefore, this investigation determines if a single-event model is appropriate for characterizing the eradication process through time. We hypothesize that there is no difference between groups, given that concerted efforts have been made to thwart rodent incursions (the precursor of reinvasion) over the past 20 years (Russell & Broome, 2016). We used the Cox proportional hazards model (the Cox model) with stratum (Hosmer et al., 2008) to compare the survival (i.e. probability an island remains un-eradicated) of eradicated islands that previously experienced reinvasion with those that have not experienced reinvasion. We then used the log-rank (Mantel-Haenszel) test to determine whether the effect of reinvasion on observed survival was statistically significant (Peto & Peto, 1972). In total, 23 different eradicated islands have been reinvaded and 57 have not

(Appendix S1). See appendix S2 for an in-depth overview of the Cox model in relation to our study.

For our second test (*ii*), we investigated the predictive power of our selected candidate model. Here, we replaced parameter estimates from our selected model with estimates from New Zealand's eradication history as at 2010 (years 1980-2010). We considered the event status of islands at this time and predicted the probability of eradicating rats from invaded islands within the 2010-20 decade. We hypothesize that islands actually cleared of rats within the 2010-20 decade (n = 12) should have a high probability of project success (≥ 80% eradication probability) by year 2020, seeing as eradication actually occurred within this timeframe.

Results

i. Describing New Zealand's rat-eradication history on islands

Log likelihoods [log($\mathcal{L}$)] for the fit of different distributions indicated highly skewed right-tailed curves appropriately characterize New Zealand's rat eradication history. The inverse gamma, inverse Gaussian, and Burr distributions were the best fitting options (log($\mathcal{L}$) of -390.31, -390.34, and -390.35, respectively) with nearly identical survival estimates (Figure S3.1). We ultimately selected the inverse Gaussian distribution to describe our data (Figure 2) due to similarities in our characterization with stochastic population-persistence models previously used to investigate mammal persistence on islands, e.g., Duncan and Forsyth (2006). These models demonstrate how island populations, affected solely by fluctuations in environmental and demographic stochasticity, have persistence times approaching an inverse Gaussian distribution (Dennis, Munholland, & Scott, 1991; Lande & Orzack, 1988). The survival trajectory of our dataset should be similar because rats have inhabited all considered islands for an extended length of time (≥ c.a. 25 years) (Duncan & Forsyth, 2006) and eradications are analogous to stochastic events that result in population extinction (Dennis et al., 1991).

Modelling the observed survival data with an inverse Gaussian distribution adequately described New Zealand's 40-year eradication history (1980-2020, Figure 2). After an initial lag-period of *ca* 10 years (1980-1990), New Zealand's rate of eradication successes were relatively constant until 2005 (*ca* 15 years), upon which the frequency became more erratic. From *ca* 2010 and beyond, incremental advances in eradication technology (e.g., implementation of eradication units and multi-species eradication projects (Griffiths et al., 2015)) resulted in a significant number of simultaneous eradications, followed by periods of eradication inactivity. Our selected distribution over-estimated survival during these periods but appropriately characterized survival when activity re-occurred.

ii. Factors influencing the time to eradication success

Of the different candidate models (representing all possible combinations of explanatory variables and their interactions), eight had substantial support with $\Delta AIC_C \leq 2.0$ relative to the best performing model (Table 2). All eight models included the parameters island area, land buffer (i.e. proportion of land surrounding the focal island), the number of private owners, and the number of Māori owners. Seven of the eight models also included the number of stepping stones, and the presence of a landing structure. However, the four parameters of distance offshore, public, private and Māori interest were present in only zero to two candidate models. These parameters all increased the relative expected information (Kullback-Leibler) distance and decreased overall model parsimony, suggesting a relatively poor approximation of the survival data (Burnham & Anderson, 2002). Therefore, these four parameters were considered unnecessary for predicting the time to eradication success. We thus used our highest ranked model, which contained all other explanatory parameters (Model 1, Table 2), to make eradication predictions for each rat-invaded island.

iii. Eradication predictions

Our highest ranked model predicts only two of 18 rat-invaded island reserves are likely to be rat-free (≥ 80% eradication probability) by 2025 (Table S3.1), and 14 of 74 islands are likely to be rat-free by year 2050 (Table S3.2). Moreover, our projected survival curve (Figure 2) has a survival estimate of 0.28 (0.14, 0.41 95% CIE) at 2050 (i.e. a cumulative eradication probability of 72% of islands for our dataset). The survival curve did not drop to 0.01 within the foreseeable future (projected to 2080). Eradication probability generally increased in the presence of stepping stone islands and landing structures but decreased with increasing island area, and buffer proportions. Eradication probability also decreased with the number of owners (both private and Māori), though this relationship was not necessarily true for islands with a ‘high’ number of owners (Table 2). This particular result is likely a reflection of the small sample size within specific categorical bins selected for the number of private and Māori owners (e.g. the number of owners having a ‘high’ factor level: private n = 7, Māori n = 14), as opposed to some exposed truism of rat-eradication on inhabited islands. Overall, the predicted outcomes (i.e. eradication predictions for each island to 2025 and 2050, and the total proportion of rat-free islands at 2050) cast doubt as to whether the desired conservation policy will be achieved within the designated timeline under current technology and projections. Our predictions suggest PF2050’s objectives are not feasible under current rates of eradication advancement.

We further demonstrate the effect of the highest ranked model terms on survival probability with their influence on the fitted distribution’s mean parameter and dispersion value, through year 2050 (Figure 3). Except for the number of private and Māori owners (Figure 3e-f), all survival curves were fitted using an "average" island and were manipulated using respective parameter factor levels, discrete values, or selected thresholds for continuous variables. Threshold values for insular area were selected to highlight New Zealand's conservation strategy of targeting medium-sized islands for eradication. An "average" island had a mean value of each model parameter (16.54 % surrounding land-area, one stepping stone along the dispersal pathway, smaller than 316 ha, absent of landing structures, and no private or Māori owners; Table 1). Private and Māori ownership curves required slightly different survival descriptions due to variable correlation (based upon one-way analyses of variance with $p <$ .05, effect of area on: private ownership levels $F_{3,\,150} = 17.93$, Māori ownership levels $F_{3,\,150} = 3.49$, and the presence of landing structures $F_{1,\,152} = 34.88$; see Appendix S2 for complete details).

iv. Retrospective analysis

Supporting our first hypothesis that reinvasion has not drastically affected PF2050's timeline, the re-eradication of formerly eradicated islands post-reinvasion was not statistically different from islands never having experienced reinvasion (log-rank test of the Cox proportional hazards model with stratum (Hosmer et al., 2008): *test statistic* = 0.32, $df = 1$, $p = .57$). Contrary to our second hypothesis, only four of 12 islands eradicated within the 2010-20 decade were predicted to have a high probability of eradication success by 2020 (≥ 80%; Table S3.3). However, nine of 12 islands had an eradication probability ≥ 70%, and all islands actually eradicated within the decade had > 50% eradication probability (Table S3.3). We purposefully used a conservative 80% probability threshold to provide a stringent test of our dataset. Appropriate success thresholds may in fact be context dependent and, evidently, rats can be successfully eradicated from some New Zealand islands under lower success thresholds (e.g. when a regional commitment to biosecurity is higher).

Discussion

i. Eradication predictions

Invasive rats have caused multiple extinctions of island species, and their eradication is critical to saving and restoring native biota. Our survival model conservatively predicted only a fraction of the remaining rat-invaded islands in New Zealand will be eradicated of rats by the designated 2050 deadline (Table S3.1 and S3.2). Given these predictions are extrapolated based only upon the incremental improvements of past eradication practice, further advances are needed to achieve greater magnitudes of eradication step-change (c.f. Figure 2). Therefore, we suggest it is imperative for New Zealand to invest in, and develop, novel technical and social tools for eradication to increase the current rate of eradication successes. Such technologies must be more effective than those of the past and must expand on current capacities to enable projects of increased complexity. In the survival context, these technologies must increase the instantaneous probability of eradication success at any given point in time (describing the 'hazard function' of our survival model, Figure S3.2), thereby steepening the descent of the survival curve that has begun to flatten.

A significant body of work seeks to overcome the limitations of the existing rodent eradication toolbox. In particular, a suite of transformative technologies is being developed to increase the humaneness and specificity of existing tools, but also to overcome current levels of socio-political opposition and high fixed-costs (Campbell et al., 2015). Prominent examples being considered in New Zealand include genetic systems (e.g., the 'Trojan female' approach; Gemmell, Jalilzadeh, Didham, Soboleva, and Tompkins (2013)), species-specific toxins (e.g., the Rattus-specific toxicant *Norbormide*; Rennison et al. (2012)) and novel social processes (e.g., conservation conflict transformation Campbell et al. (2015), Madden and McQuinn (2014)) among others. Based upon the findings of our study, the scale of PF2050 and the heterogeneity of islands under consideration, a combination of transformative tools is likely required to achieve timeline goals. Moreover, substantial funding will be vital to making any such tool a reality. The New Zealand government and supporting bodies have expressed a strong desire to overcome the existing lag in eradication advances (over $23million NZD was allocated to Predator Free 2050 Limited 2016-2020 and $76million NZD has been committed from 2020-2024; Predator Free 2050 LTD (2020)) by recognizing that economic benefits and improved ecosystem services will outweigh incurred costs (Russell et al., 2015), and by virtue of the stated conservation goals (Sage, 2020). We expect in the future, once transformative tools are available, survival methods would substantiate how important this new funding will be to accelerating PF2050's timeline, but the relative novelty of this budgetary allocation currently forbids investigation due to a current lack of survival outcomes from the use of any new technologies yet to be implemented.

As is the case with other transformative technologies throughout history, many of these new approaches have been subject to initial public skepticism and concern. Similarly to entrenched attitudes which limit existing technologies (e.g., aerially applied poison baits), programs of public engagement acknowledging and responding to underlying attitudes, social norms and behaviors will be critical to implementing transformative tools at scale (MacDonald et al., 2020). Indeed, not only will novel tools be essential to the success of PF2050, so too will be effective communication about the tools. However, as the New Zealand government championing of PF2050 is a recent event (from 2016), the galvanizing effect of this unifying program has yet to be reflected in eradication rates, which may steepen the curve's descent again.

Factors most influencing the time to eradication success were those present in our highest-ranking candidate model. We briefly interpret some of their impacts on survival (for complete details see Appendix S3). Expectedly, eradication probability declines with increasing island area, although advances in eradication technology mean the slope continues to flatten. We found that stepping stones increase eradication probability. Such a result seemingly contradicts biogeographical theory, given that stepping stones are noted to increase biotic exchange and hence reinvasion probability between two locations (MacArthur & Wilson, 1967). However, we posit this is due to the homogenously inhospitable seascape surrounding each island, and that distance offshore is correlated to stepping stone quantity in this instance (Carter et al., 2020). Our model affirms that accessibility to other landmasses (i.e. landscape buffer metric) directly influences the time to eradication success. Locations surrounded by land that can be accessed with ease are more difficult to defend against reinvasion (Carter et al., 2020), thereby increasing the time to eradication implementation and success. We also found that the accessibility of islands (presence of a landing structure) decreases the time to eradication success. Rodent presence is documented as being associated with landing structures on islands (Russell & Clout, 2004) and so along with simplifying project logistics, we suspect this results in preferential candidate selection. Finally, we found the largest impact on the time to eradication success is associated with an island's number of owners (regardless of their vested interests). In general, the probability of eradication success through time decreases as the number of island owners increases. However, we note this relationship is not as well defined for the number of Māori owners and some additional interpretation is required. The survival curve drops dramatically below the other variable curves for a 'high' number of Māori owners (> 500 individuals) at *ca.* 30 years, suggesting a non-linear relationship. We posit this outcome is due to the variable's selected bin size (see Appendix S3 for further details), but could also be an emergent reflection of the cultural commitment *iwi* have in caring for the environment (a concept termed *kaitiakitanga*) (Roberts, Norman, Minhinnick, Wihongi, & Kirkwood, 1995). This *kaitiakitanga* may be driving the commonality of values to facilitate mobilizing conservation action for a large number of people; e.g. the 2006 rat eradication on the culturally important *Titi*/Muttonbird Islands

(McClelland et al., 2011). While the mechanisms underlying this concept have yet to be disentangled within a scientific context, we suggest doing so may assist in leveraging conservation action at greater scales for islands with many different private landowners in New Zealand and globally. Overall, we find the most transformative eradication technologies of the future are likely to be those focused on attaining community buy-in from highly populated islands or islands with many different owners (Holmes, Spatz, et al., 2019; Parkes et al., 2017).

ii. Retrospective model validation

The log-rank test could not detect a statistical difference between the survival probability of rat-eradicated islands with reinvasion and without reinvasion. Survival curves for these groups converged from the mid-2000s onward, *ca.* 25 years after the systematic eradiation of rats began (Figure 4). Before this time, reinvaded islands had exclusively lower eradication probabilities (i.e. higher survival probabilities). These findings demonstrate the increased efficacy of New Zealand’s biosecurity measures on preventing reinvasion through time (Russell & Broome, 2016), and that the possibility of reinvasion has not been a significant deterrent to eradication success since *ca.* 2005 (Figure 4). This substantiates our choice to characterize survival with a single-event model instead of one with recurrent events (Hosmer et al., 2008). However, we note reinvasion will remain an important consideration in the future as islands of increasing accessibility and lying within complex land and seascapes are targeted for eradication (Carter et al., 2020).

Retrospective analyses are a powerful and under-utilized model validation tool in conservation decision-making. Upon retrospectively investigating the short-term probability of project success for eradications occurring between 2010 and 2020, we found that only four of 12 islands were predicted for successful eradication based on an 80% threshold probability

(Table S3.3). This underestimate of the true outcomes suggests that this threshold value is too conservative for considering when a project is likely to succeed in this context. Indeed, by relaxing this assumption to 50% survival, our highest ranked model predicted all 12 islands would have had eradication success by 2020. For half of the islands, eradication success between 2010 and 2020 hinged upon an above-ordinary local commitment to maintaining rat-absence in the presence of high reinvasion rates, which would otherwise be prohibitive to maintaining an island rat-free under current circumstances. New Zealand's conservation trajectory is much more optimistic if we apply this same 50% threshold long-term to 2050, with 67 of 74 invaded islands then projected to be rat-free (totaling 85 % eradicated island land-area, excluding mainland landmasses). Although still falling short of achieving complete success, PF2050 could then be regarded as an attainable conservation initiative. This particular outcome highlights the importance of selecting an appropriate probability threshold. We advise future studies reserve values ≥ 50% for feasible projects, and ≥ 80% for projects guaranteed to succeed within the desired timeframe. Such values are likely applicable to other rat eradication projects conducted in temperate or sub-tropical regions globally (Holmes et al., 2015). However, we implore careful examination of the probability of project success prior to setting eradication priorities with a temporal dimension. The effectiveness of a recommendation depends on the accuracy of the selected threshold; otherwise, the given intervention may be too liberal or too conservative, as was the case here. Methods for determining the probability of project success are discussed elsewhere but are often based upon expert consensus, e.g., (Holmes, Spatz, et al., 2019; Joseph et al., 2009). Therefore, our estimates should be viewed as an examination of PF2050's potential outcome if transformative eradication advances are not made, rather than to model its definitive outcome.

iii. Temporal ranking

We ranked each rat-invaded island by when eradication is predicted to succeed and also identified the factor(s) that project success is most dependent upon (Table S3.2). Existing methods for prioritizing island eradications have yet to incorporate the time to project implementation and subsequent success. Invasive species are expected to be complicit in causing the extinction of approximately 1,000 species of avifauna per million species per year (E/MSY) as the 21st century progresses (Pimm, Raven, Peterson, Şekercioğlu, & Ehrlich, 2006). This novel extension of the existing decision-making framework will help combat extinctions by incorporating how soon intervention measures for a particular island are likely to become biogeographically and socially feasible. Moreover, our approach provides an objective tool for determining whether eradication, rather than sustained control, is the most appropriate conservation intervention on an island (Duron, Shiels, & Vidal, 2017). Eradication requires that all target individuals be extirpated within a fixed timeline, else the project becomes a *de facto* control operation (Bomford & O'Brien, 1995). Survival methods inform such timelines via intuitively understood success probabilities (Hosmer et al., 2008); projects likely to exceed a specified timeframe can be recommended for alternate sustained control approaches. Although a back-catalogue of completed projects is required to adequately inform survival methods, information already exists for terrestrial vertebrate eradications from >1000 islands (Holmes, Keitt, et al., 2019), and their efficacy as a conservation tool will increase with time as the number of eradications continues to increase globally.

Critically, our approach should be considered as a key extension to the existing conservation decision-making framework, rather than a tool to prescribe a specific eradication program. Setting eradication priorities requires conservation managers to evaluate multiple different factors. For example, West Chicken Island *Mauitaha* (174.696°E, 35.894°S: Hauraki Gulf; Figure 1) has a cumulative survival estimate of 0.03 at year 2050 (i.e., an eradication probability of 97 %; Table S3.1), and so has a high priority on a purely temporal ranking. However, Pacific rats are protected from eradication on this island under an agreement between local Māori *iwi* (i.e., tribe) and the New Zealand government (Tahana, 2010). In another example, Waiheke Island (36.80°S, 175.10°E: Hauraki Gulf; Figure 1) is an important conservation priority of PF2050 due to its large land-area and posed biosecurity risk (Bassett, Cook, Buchanan, & Russell, 2016). However, although it is currently the target of a PF2050 stoat (*Mustela erminea*) eradication, Waiheke is unlikely to be rat-free by 2050 unless major eradication advances are made (Table S3.2). Our survival model indicates that the very large number of inhabitants here is the single-greatest influencer of survival probability and will, therefore, be the factor that most determines project feasibility (Table S3.2, Figure S3.3). Therefore, in order to achieve the desired outcomes by 2050 for this island, we suggest investing in community engagement to foster eradication support (Russell et al., 2018). Indeed, limiting the availability of management options reduces the efficacy of any optimization scenario (Helmstedt et al., 2016). Instead of writing a project off as ‘improbable’ or ‘impossible,’ our approach identifies alternative roadmaps to success. Prioritizing eradication candidates solely on the time to project success does not produce a fully informed recommendation; other desired outcomes must be identified consequently to using survival models in this context. We thus envision our approach being only the first-pass filter for triaging a suite of eradication priorities.

While survival methods have been applied to answer ecological and conservation-related questions in the past (Bischof et al., 2012; Duncan & Forsyth, 2006), our study bridges an existing gap in conservation decision-making by providing managers with an objective tool to forecast project success timelines. With our case study, we developed an understanding of the mechanisms driving time to project success and forecasted the probability of achieving a legislated conservation target. Such information can inform the prioritization process as part of an existing decision-making framework, and can be used to maximize the probability of achieving desired outcomes for identified conservation priorities within a designated timeline. Moreover, we demonstrated that survival methods can be validated retrospectively to test the robustness of temporal forecasting. Given that urgent conservation initiatives with differing probabilities of success over time are globally abundant (Butchart et al., 2010), survival methods should be of immediate and important benefit to decision-makers. Island regions protecting threatened biota under a fixed and limited budget, such as the United Kingdom, French overseas territories, Polynesia and Indonesia (Genovesi & Carnevali, 2011; Myers et al., 2000), will be particularly amenable to this form of temporal prioritization.

In-text figure captions

Figure 1: The New Zealand archipelago, (**a**) major offshore island groups are framed and mainland islands are labelled in italics, (**b-d**) rat-invaded outlying island groups with major islands labelled in italics.

Figure 2: Non-parametric (Kaplan-Meier) survival curve (step-wise solid line) of all rat-eradication projects occurring throughout New Zealand's modern conservation history (1980-2020) and fitted inverse Gaussian distribution (dotted line) extending through year 2080. Blue shading represents the proportion of rat-invaded islands restored through current, red shading represents the proportion of islands projected to be rat-invaded through year 2050 and grey shading represents 95% confidence interval estimates through year 2080.

Figure 3: Effect of highest ranked model explanatory variables (Table 2) on survival probability through time, as influenced by the fitted distribution's mean parameter and dispersion value. Survival curves for panels (a-d) were fitted using an average island representative, and panels (e-f) were fitted using a range of values. Threshold values for insular area (panel a) were categorised as being "small" (< 25 ha), "medium" (25-200 ha) or "large" (> 100 ha) in size; Land buffer proportions (panel b) were normally distributed and so were categorised as being greater than or less than the mean for "large" and "small" proportions, respectively. Because curves were not all fitted in the same manner, effects on survival should only be compared within terms.

Figure 4: Cox proportional hazards model comparing survival probabilities of eradicated islands that have been reinvaded with those that have not been reinvaded.

In-text tables

Table 1 Explanatory variables and their description in the context of this study.

Isolation Metrics			
Variable	Description	Range	Mean [*n*]
Distance[a,e]	Shortest Euclidean distance from mainland to focal island (km)	0.01 – 875	16.57
Area[a,e]	Land area of focal island (ha)	5 – 80,459	1,133
Stepping Stones[b]	Number of intermediate stepping stone islands (≥ 1ha) between mainland and focal island	0 - 7	0.90
Buffer[a,f]	Proportion of land encapsulated in a 3km buffer projected outward from the focal island's perimeter (%)	0.00 – 73.74	16.57
Landing[c]	Presence of a landing structure on focal island (airfield and / or wharf)	0/1	[34]

Land-Tenure Metrics			
Variable	Description	Range	Mean [*n*]
Public[c]	Publicly owned focal island	0/1	[94]
Private[c]	Privately owned focal island	0/1	[40]
Māori[c]	Māori owned focal island identified under Māori Land Act 1993	0/1	[44]
Private Owners[d]	The number of different private landowners on focal island Categories: none, low (1-10 owners), medium (11-100 owners), high (> 100 owners)		[113, 28, 6, 7][g]
Māori Owners[d]	The number of different landowners identified under Māori Land Act 1993 Categories: none, low (1-100 owners), medium (101-500 owners), high (> 500 owners)		[110, 15, 15, 14][g]

[a]Continuous, [b]Discrete, [c]Binary, [d]Categorical.

[e]$\log_{10}$ transformed, [f]cubed-root transformed.

[g]Corresponding to factor levels none-low-medium-high.

Table 2: Goodness-of-fit, as measured by AIC_C used to explain variation in the eradication time of invasive rats on New Zealand islands (2,400 candidate models in total). Models shown are those with substantial support ($\Delta\text{AIC}_\text{C} \leq 2.0$). Shaded symbols are for included model parameters. Parameter estimates are given for the highest-ranking model. The number of parameters estimated k, log likelihood Log($\mathcal{L}$), AIC_C, $\Delta\text{AIC}_\text{C}$, and model probability w_i are also shown for each model.

	Model Parameter														
Model	Distance	Area	Stepping stones	Buffer	Landing[a]	Public[a]	Private[a]	Māori	Private Owners[b]	Māori Owners[b]	k	Log($\mathcal{L}$)	AIC_C	$\Delta\text{AIC}_\text{C}$	w_i
1	E	7.54 (2.64)	–1.57 (0.82)	14.29 (2.02)	-11.39 (3.28)	E	E	E	14.09 (6.55) 76.69 (61.71) 51.10 (55.15)	58.69 (52.01) 65.50 (56.22) -3.38 (4.16)	12	-354.66	735.54	0.00	0.23
2	–	+	–	+	–	–	E	E	+ + +	+ + –	14	-352.68	736.37	0.84	0.15
3	E	+	–	+	–	–	E	E	+ + +	+ + –	13	-353.94	736.48	0.94	0.15
4	E	+	–	+	E	E	E	E	+ + +	+ + –	11	-356.72	737.29	1.76	0.10
5	E	+	E	+	–	E	E	E	+ + +	+ + +	11	-356.72	737.30	1.77	0.10
6	E	+[c]	–	+	–	E	E	E	– + +[c]	+ + –	15	-351.95	737.37	1.83	0.09
7	E	+	–	+	–	E	+	E	– + +	+ + +	13	-354.40	737.41	1.87	0.09
8	–	+	–	+	–	E	E	E	+ + +	+ + –	13	-354.43	737.46	1.93	0.09

Prefixes: '+' positive effect on survival, '–' negative effect survival, 'E' excluded from model (parameter increases expected Kullback-Liebler distance).

[a] Binary variable compared against '0'.

[b] Factor ordered 'low-medium-high' and compared against 'none'.

Parenthesis values for parameter stand errors.

[c] Parameter interaction (positive relationship).

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