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Monitoring, imperfect detection, and risk optimization of a Tasmanian devil insurance population

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Abstract

Most species are imperfectly detected during biological surveys, creating uncertainty around their abundance or presence at a given location. Decision-makers managing threatened or pest species are regularly faced with this uncertainty, and there are a growing number of examples of managers dealing with imperfect detection. Wildlife diseases have the potential to drive species to extinction, and as such managing species with disease is an important part of conservation. Devil Facial Tumour Disease (DFTD) is one such disease that led to the listing of the Tasmanian devil (*Sarcophilus harrisii*) as endangered. Here we report on the successful use of a state-of-the-art removal modelling approach undertaken in collaboration with practitioners to inform decision-making and facilitate a successful management outcome. We used a Bayesian catch-effort model to estimate population size during removal and monitoring of a diseased Tasmanian devil population. We found it was likely that the population had been successfully removed, even when accounting for a possible introduction of a devil to the site. We then analysed the costs and benefits of declaring the area disease-free prior to reintroduction and establishment of a healthy insurance population. The actions of management, in carrying out additional monitoring prior to this reintroduction, were conservative but prudent given uncertainty and the costs of mistakenly declaring the area disease-free.

Introduction

Confirming the absence of a species is an important and pervasive conservation problem, which is made complex when species are not perfectly detected during biological surveys. Developments may be approved when ecological surveys fail to detect threatened species (Garrard et al. 2015), protection or management can be cut for threatened species incorrectly presumed to be extinct (Collar 1998), and campaigns to eradicate introduced pests are halted when individuals are no longer detected (Regan et al. 2006; Solow et al. 2008). However, imperfect detection analyses allow managers to overcome this problem when making conservation decisions based on data that may contain false zeros (Anderson et al. 2016; Chen et al. 2013; Garrard et al. 2008; Moore et al. 2011;

Royle et al. 2005). Instead of ad hoc rules, such as waiting a set number of years with no detections before declaring a species absent (Hoffmann 2010; Rejmanek & Pitcairn 2002), such decisions can be supported by quantitative modelling that allows decision makers to understand and explicitly incorporate the risks of falsely assuming absence (Boakes et al. 2015; Regan et al. 2006; Solow et al. 2008). There are a small number of examples in which analyses of survey effort requirements due to imperfect detectability have directly informed pest animal eradication programs (Ramsey et al. 2009; Ramsey et al. 2011; Rout et al. 2014). There are few published examples in which state-of-the-art imperfect detection modelling approaches have been used to underpin the management of a cryptic wildlife disease (Anderson et al. 2013, 2015).

Devil Facial Tumour Disease (DFTD) threatens the survival of wild populations of the island endemic carnivorous marsupial, the Tasmanian devil (*Sarcophilus harrisii*). DFTD has two genetically distinct forms, which are two of only four known naturally occurring transmissible cancers (Pye et al. 2016). The disease is assumed to be transmitted through biting and other facial contact, and causes tumours on the face or inside the mouth (Lachish et al. 2007; Pearse & Swift 2006). Once tumours develop, death typically occurs within months (Lachish et al. 2010). The disease was first detected in northeastern Tasmania in 1996 and has since spread across most of the devil's habitat, resulting in an 80% decline in wild populations (Huxtable et al. 2015). In the region where the disease was first reported, mean spotlighting sightings declined by 95% from 1993–2013 (Huxtable et al. 2015). The Tasmanian devil was listed as endangered by the International Union for the Conservation of Nature in 2008 (Hawkins et al. 2008).

In the face of this disease threat, establishment of both captive and wild-living insurance populations of devils is a key management priority (Conservation Breeding Specialist Group 2008). Maintaining wild populations on Tasmanian islands (either offshore or landscape-scale fenced areas) will allow the devil to retain its ecological functionality, wild behaviour and adaptations, and intrinsic, social, economic and political value (Conservation Breeding Specialist Group 2008). Forestier Peninsula, off

Tasmania's east coast, was identified as a potential site for a wild-living insurance population. The peninsula is connected to mainland Tasmania in the north-west via a road-bridge spanning the Denison Canal, and to Tasman Peninsula by a narrow isthmus in the south. This potential for isolation makes the peninsula a good site for maintaining a disease-free devil population. However, DFTD was detected within the resident population, making it essential to remove this population before introducing devils from a captive disease-free population.

We aimed to support decisions about the monitoring effort and duration required to confirm devil absence at a sufficiently high level of confidence to commence reintroduction of healthy devils. We developed a Bayesian catch-effort model of the devil population on Forestier peninsula, implemented in an add-on to Excel, which provided a simple and accessible interface through which managers could interactively assess the effect of different levels of monitoring on uncertainty about whether the devil population had been completely removed. This assisted managers in deciding whether to conduct an additional second camera trapping session in 2015, before devil reintroduction. We formally assessed the cost-effectiveness of additional monitoring given its cost, and the risk and cost of introducing healthy devils when potentially diseased devils may persist on the peninsula. This analysis supported management decisions that ultimately led to the introduction of 39 healthy captive-bred devils in November 2015.

Methods

Data collection

Trapping and removal of devils occurred from 10 May to 5 June 2012. Devils were trapped using PVC pipe traps and standard methods (Hawkins et al. 2006). All trapped devils were removed from the site and assessed by a veterinarian. Individuals confirmed as DFTD positive were euthanized except females with pouch young, the latter received close monitoring and veterinary care in captive facilities and were euthanized once their young were able to survive independently or their welfare was severely compromised. Devils with no symptoms of DFTD were placed into captive quarantine and absorbed into the Tasmanian devil insurance metapopulation.

One infrared camera survey was carried out prior to trapping and removal in 2012, and six surveys conducted afterwards in 2012, 2014 and 2015 (Table 1). Surveys used either Pixcontroller DigitalEye 12.1 megapixel Trailmaster or Reconyx HC500 HyperFire™ remote cameras programmed to take one image on detecting movement, with a 30 second delay before subsequent triggers/images. Cameras were mounted at between 0.5 and 1.5m height targeting a ground bait (Bennett's wallaby or brushtail possum shot locally under crop protection permits) 1.5 to 3m away with a lure (consisting of oats, sardines in oil, dried liver treats, fish oil and mutton-bird oil) aerially suspended above.

Modelling

We developed a Bayesian catch-effort model (Chee & Wintle 2010; Ramsey et al. 2009; Ramsey et al. 2011; Rout et al. 2014) to estimate population size during and after removal and monitoring, and to allow us to plan future monitoring intensity (see Supporting Information for model schematic).

The number of devils present on day *t*+1 was calculated as:

$$N_{t+1} = N_t - n_t,$$

where n_t is the number of devils removed each day by pipe trapping. We modelled this as a binomial process:

$$n_t \sim Bin(p_t, N_t),$$

where p_t is the probability of catching and removing a devil on day *t*. This is a function of the density of pipe traps set that day:

$p_t = 1 - e^{-ag_t}$ (1)

where a is the effectiveness of pipe trapping, and g_t is the number of active traps per km^2 on day t. This exponential function is derived from search theory (Frost & Stone 2001), assuming encounters between devils and traps are random and independent with all individuals having the same probability of capture.

Devils have not been individually identified in camera footage, therefore camera data cannot tell us the number of individuals seen each day. It instead tells us the number of cameras each day that sighted at least one devil (m_t), which we modelled as a binomial process:

$m_t \sim Bin(r_t, c_t),$

where c_t is the number of active cameras on day t, and r_t is the probability of an active camera detecting at least one devil:

$r_t = 1 - e^{\frac{-bN_t}{A}}.$ (2)

where b is the daily effectiveness of camera monitoring and A is the area across which the cameras are distributed (see Supporting Information for derivation).

The model inputs were daily data on the number of devils trapped and removed (n_t), the density of active pipe traps (g_t), the number of cameras with one or more detections (m_t), and the number of active cameras each day (c_t). We generated posterior distributions for daily population size (N_t), the effectiveness of pipe trapping (a) and the effectiveness of camera trapping (b) using Markov chain Monte Carlo (MCMC) sampling. We developed a program in Python that used the package pymc (Anand et al. 2010) for the Monte Carlo sampling and we used py2exe package to compile the program to an executable. Having the program as an executable allowed it to be used on any computer running Microsoft Windows, meaning managers could run simulations to assist in the planning process. This compiled executable is available at [removed as contains identifying information]. It does not require a python installation to run.

At the start of 2012 we used vague prior distributions for the effectiveness of pipe trapping ($a \sim U(0, 1000)$), the effectiveness of camera monitoring ($b \sim U(0, 1000)$), and the initial population size of devils on Forestier Peninsula ($N_0 \sim U(0, 500)$). A maximum of 500 devils on the 190km^2 peninsula implies a density of 2.63 devils/km^2, which is very high given known densities of devils in suitable unmodified habitat are between 0.3-0.7/km^2 (Jones unpublished data, cited in Jones et al. 2004). For each MCMC simulation we ran 1 chain for 10,000 iterations with a thinning rate of 10 and a burn-in of 1,000. We checked for convergence by visually inspecting chains on test data. We were unable to include trace plots in the final executable due to conflicts between python packages. However, repeatedly running the analysis on the same data gave consistent results, indicating that the chains were converging.

Immigration, reproduction, and a possible introduction

Forestier Peninsula is connected to Tasman Peninsula in the south by a 125m wide isthmus, and to the Tasmanian mainland (and DFTD-infected devil populations) in the north by a two-lane bridge spanning a canal that runs through an 800m wide isthmus. These entry points allow some movement by devils onto the peninsula, however barriers to devil movement have been installed to isolate Forestier Peninsula from diseased populations to the north.

This potential immigration is hard to estimate, so we examined a range of possibilities by creating two probability distributions for the rate of immigration (Supporting Information): one that devil managers considered realistic ($B\sim(n = 30, p = 0.05)$) and one they considered high ($B\sim(n = 50, p = 0.1)$). The expected number of entries per year under realistic immigration is $np = 1.5$, while the expected number under high immigration is $np = 5$.

Tasmanian devils are synchronous annual breeders, with recruitment into the population effectively occurring around the beginning of each year when juveniles become independent (Pemberton 1990). In our model we assumed reproduction after immigration each year. We used an annual growth rate of $\lambda = 1.29$, the estimated growth rate of an island population of devils below carrying capacity (Conservation Breeding Specialist Group 2008). We expressed this as a binomial distribution where one devil has probability p of becoming two devils ($B \sim (n = N, p = 0.29)$, where N is the population size before reproduction).

We multiplied these two binomial distributions to obtain distributions for the possible devil population size after one year and two years of immigration and reproduction, given a starting population of zero devils (Supporting Information). To use these distributions as priors for population size in our model, we converted them from discrete binomial distributions to continous gamma distributions. We did this conversion using the 'fit' function in scipy, which uses maximum likelihood.

In May 2015 managers received reports that a nearby landholder had translocated a devil from a DFTD-infected population to the Forestier Peninsula. The accuracy of the report was uncertain, with managers estimating a probability of 0.4 the translocation had occurred. To incorporate a scenario where one year of immigration and reproduction precedes this uncertain introduction, we took the distributions after one year of immigration and reproduction and calculated the expected distribution under each outcome given a probability of $p = 0.4$ that the introduction occurred and $p = 0.6$ that it did not (Supporting Information). Managers undertook 36 days of camera trapping in May-June 2015 immediately following news of this potential introduction (Table 1).

Informing decision-making

Following the session of monitoring in May-June 2015, we assessed whether additional monitoring was needed to confirm absence. We calculated the net expected cost of additional monitoring, which incorporates the risk and consequences of ceasing monitoring prematurely and mistakenly assuming devils have been successfully removed. The net expected cost of conducting *d* days of additional monitoring is (Regan et al. 2006):

$$NEC(d) = (d-1)C_m + [1 - \Pr(E|d)]C_p, \quad (3)$$

where C_m is the cost of a day of monitoring, [1 - Pr($E|d$)] is the probability devils are still present despite *d* days of monitoring without detection, and C_p is the cost of declaring absence prematurely and introducing healthy devils to the peninsula when diseased devils are still present. The probability that no devils remain (Pr($E|d$)) was taken from the posterior distribution for the estimated number of devils remaining on day t,(N_t), given past daily detection and removal data plus the proposed additional monitoring assuming no further detections. We assessed additional monitoring in blocks of 14 days, as this was the standard duration of deployment without refreshing lures and servicing cameras. We tested two monitoring intensities: a low intensity strategy deploying 28 cameras costs AU$6,000 for the first 14 day block and AU$3000 for each subsequent block, while a high intensity strategy of deploying 58 cameras costs AU$8,800 for the first 14 day block and AU$4,400 for each subsequent block.

Results

At the densities of traps used and given the conditions at the time, pipe trapping and camera monitoring were both highly effective (Table 2, also Supporting Information). Estimates of effectiveness remained relatively constant from 2012 to 2015, regardless of the assumed immigration rate (Table 2).

A total of 35 devils were removed from the peninsula in 2012, with all removals occurring within the first 12 of 27 trapping days (Fig. 1a). The probability no devils remained reached 1 (accurate to 2 decimal places) before the end of the trapping period (Fig. 1b). The probability that no devils remained returned to 1 by the end of 2014 monitoring, and by the end of the first 2015 monitoring session, regardless of the assumed immigration rate (Table 2).

We estimated the cost of declaring absence prematurely (C_p) as the cost of having to re-do the depopulation program and translocation on Forestier peninsula (Supporting Information), but then explored the optimal monitoring decisions for a wide range of C_p. Given the baseline estimate of C_p = AU$676,470, it was not cost-effective to carry out an additional 14-day block of monitoring in 2015, regardless of the assumed immigration rate or monitoring intensity (Fig. 2). However, it could be optimal if the cost of declaring absence prematurely were much higher, > AU$1.80 million under a realistic immigration rate or > AU$1.35 million under a high immigration rate (Fig. 3).

Discussion

Managers used the results from this analysis to plan the best time to reintroduce disease-free devils to Forestier Peninsula. These results were assessed as part of an overarching, qualitative risk assessment that considered the assumptions inherent in the model, the limitations of methods used to generate the field data, the risk of reinfection, and the broader context of Tasmanian Devil management,

Our population model has several assumptions that could affect population estimates. Use of a binomial process for camera trap detections could lead to an overestimation of the number of devils remaining (see Supporting Information), which would be conservative. However, the model also assumed that all devils are detectable (given enough effort) at all population densities, an assumption that could lead to underestimation of the number remaining. The use of a single monitoring method (infrared cameras) since June 2012 rather than multiple independent detection methods, makes this assumption particularly relevant. We also assumed immigration was a one-off event occurring at the beginning of the year, discounting the possibility that immigration may have occurred during or after the first monitoring session in 2015.

Several logistical considerations influenced the decision of when to conduct the reintroduction, including the fact that funding and personnel were available for the reintroduction in the near future but could not be guaranteed beyond that. Timing was also relevant in terms of the supply of healthy devils to reintroduce, as immediate translocation of some devils from their captive breeding facilities was required to maintain ongoing breeding capacity for the Tasmanian devil insurance meta-population. Prolonging the project for further monitoring could also divert resources from other devil conservation management activities.

Considering all these factors, managers chose to carry out a second monitoring session in October-November 2015, after which the probability that no devils remained was exactly 1 under both a realistic and high immigration scenario (Supporting Information). A 'devil-proof' fence was then erected across the 800m wide isthmus connecting Forestier Peninsula to mainland Tasmania to reduce the risk of immigration of diseased devils onto the peninsula. On 18 November 2015, 39 adult, captive-bred, DFTD-free Tasmanian devils were reintroduced to the Forestier Peninsula, and on 25 February 2016 an additional ten juvenile devils were translocated. These devils are being managed as part of the Tasmanian devil insurance meta-population by the Zoo and Aquarium Association and the Save the Tasmanian Devil Program, with a sustainable population target of 150 devils on the Forestier Peninsula.

Overall, the removal modelling and risk optimization analysis provided valuable support to managers of the 'Save the Tasmanian Devil' program by insuring that risks of program failure due to the persistence of diseased devils in the insurance population location were sufficiently low, given the monitoring and removal effort that had taken place. Ceasing monitoring prematurely would have wide-ranging consequences that are much more difficult to express as a monetary cost and so were not included in our formal cost-effectiveness analysis. Exposing healthy captive-bred devils to a fatal disease would be a tragic outcome for all involved. The failure of the depopulation project would come at a high reputational and political cost to the Program, which could lead to a loss of financial support. Managers chose to implement a second monitoring session in 2015, despite the fact that our analyses indicated a second monitoring session was not cost-effective given the estimated cost of ceasing monitoring prematurely (C_p = AU$676,470). Nonetheless, the extra round of monitoring was cost-effective if the costs of premature release of healthy devils were assumed to be approximately twice the baseline amount. Taking into account uncertainty in the model predictions, and the intangible costs mentioned above, the extra season of monitoring was prudent and defensible and could accommodate the logistical limitations for the reintroduction. Ultimately, our modelling and analysis informed an adaptive decision making framework for the Save the Tasmanian Devil Program that also incorporates social, political, economic and operational opportunities.

Imperfect detection is pervasive in species management, and if ignored can lead to erroneous assumptions and adverse outcomes (Kéry et al. 2006; Solow et al. 2008). Population estimation and modelling can better inform decision-making, while risk optimisation ensures that the costs and benefits of management decisions are considered. While built expressly for the purposes of this program, our Excel tool is freely available and could be applied to any population removal program that uses a single removal method and a single monitoring method, subject to the model assumptions. In addition to disease management, this could include pest eradication programmes (Ramsey et al. 2009; Ramsey et al. 2011; Rout et al. 2014), and capturing a wildlife population for captive breeding or translocation.

Policy recommendations and conclusions

Trade-offs are implicit in any decision to cease monitoring. Analyses such as those presented here help make these trade-offs explicit, and enable decision-makers to assess whether they are acceptable. However, these analyses are still relatively technical, requiring a high level of statistical competency to implement and correctly interpret. It is not the case that managers can, for the most part, be expected to adopt these methods directly from the scientific literature. If conservation scientists wish to see these approaches used in management, they must invest the necessary effort to facilitate application (Artellaz et al. 2010). This project, like most projects in which state-of-the-art statistical approaches are successfully applied in conservation, required a long-lasting, consistent engagement between researchers and practitioners (Whitehead et al. 2016). This engagement occurred with little face-to-face contact between researchers and managers (who are based in different states), and despite differences in work schedules, competing priorities, and regular travel and field work commitments. The development of the excel interface was key to overcoming these barriers and was therefore crucial to the success of this project. Having an accessible interface allowed managers to quickly update their assessments to respond to new information as it arrived, (for example, during a field work session) rather than sending data to researchers and waiting to receive results. The interface was used in this way by managers to inform the duration of both monitoring surveys in 2015. This was enabled by a non-trivial investment of time and money by researchers to develop, tailor and make this tool available for use by managers, and due to the willingness of Program staff to engage with researchers to secure best practice approaches, and consistent effort on behalf of managers to ensure that the tools were exactly right for their needs (Hogg et al. 2016). This highlights the importance of genuine commitment of researchers and managers to conservation outcomes in order for research to be relevant and managers to gain the benefits of recent scientific advancements.

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Tables

Table 1: Details of Tasmanian devil trapping and monitoring sessions on Forestier Peninsula

Session Dates	Session type	No. camera stations	Area (km^2)	No. active trap nights
22 Feb - 20 March 2012	Survey: Pre-removal	28	190	593
10 May – 5 June 2012	Trapping: Removal	N/A	190	3572
12 June - 4 July 2012	Survey: Post-removal	28	190	495
2-23 Aug 2012	Survey: Post-removal	45	45	796
10 March - 7 April 2014	Survey: Post-removal	28	190	785
15 Aug - 07 Dec 2014	Survey: Post-removal	113	190	2901
7 May - 11 June 2015	Survey: Post-removal	31	190	978
19 Oct - 10 Nov 2015	Survey: Post-removal	28	190	1152

Table 2: Results of catch-effort model

	Posterior distribution for effectiveness of pipe trapping (*a*)	Posterior distribution for effectiveness of camera monitoring (*b*)	Probability that no devils remain (2 decimal places)
After 2012 removals and monitoring	μ = 61.98, 95% credible interval (CI) = 44.58-82.20.	μ = 138.73, 95% CI = 115.00-164.67.	1.00
<u>After 2014 monitoring</u>			
Assuming realistic immigration rate	μ = 61.67, 95% CI = 44.00-82.27.	μ = 138.27, 95% CI = 114.65-164.08.	1.00
Assuming high immigration rate	μ = 62.40, 95% CI = 45.69-81.68.	μ = 138.93, 95% CI = 115.31-164.71.	1.00
<u>After first 2015 monitoring session</u>			
Assuming realistic immigration rate	μ = 61.95, 95% CI = 44.53-82.21.	μ = 138.53, 95% CI = 115.93-163.11.	1.00
Assuming high immigration rate	μ = 62.62, 95% CI = 46.06-81.68.	μ = 137.81, 95% CI = 113.62-164.29.	1.00

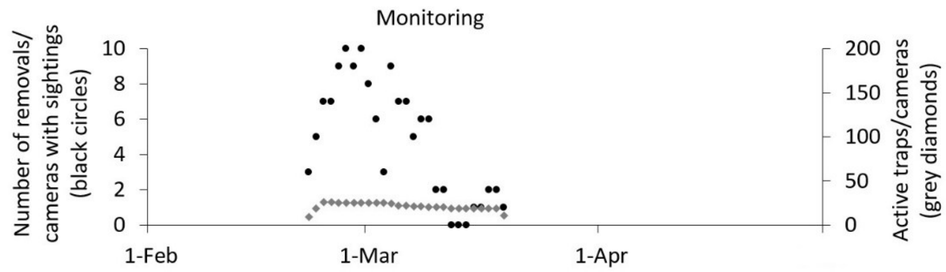
The mean probability of catching a devil is virtually identical for all posterior distributions for the effectiveness of pipe trapping, and the mean probability of detection by a camera is virtually identical for all posterior distributions for the effectiveness of camera monitoring (Supporting Information).

Figure captions

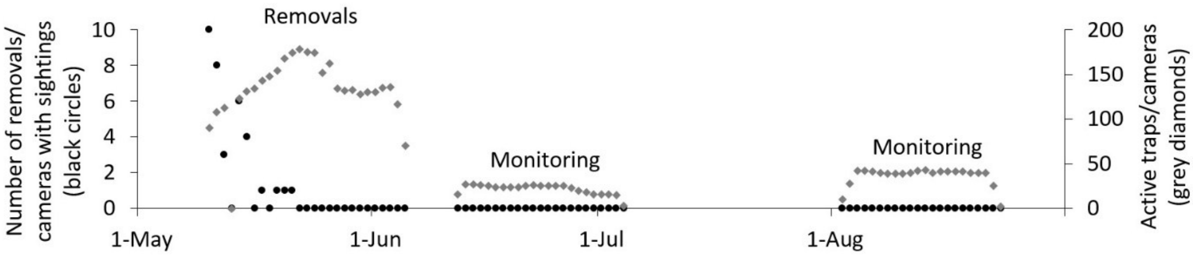
Figure 1: 2012 removal and monitoring of Tasmanian devils on Forestier Peninsula, showing a) pre-removal monitoring data, b) removal data and post-removal monitoring data, and c) population estimates from our model. Estimated population size is shown in grey (solid = mean, dotted = 95%

credible interval), while the probability no devils remain is shown in black. Population estimates could not be calculated for the period before removals occurred, as some removals are required to calibrate the model.

a)



b)



c)

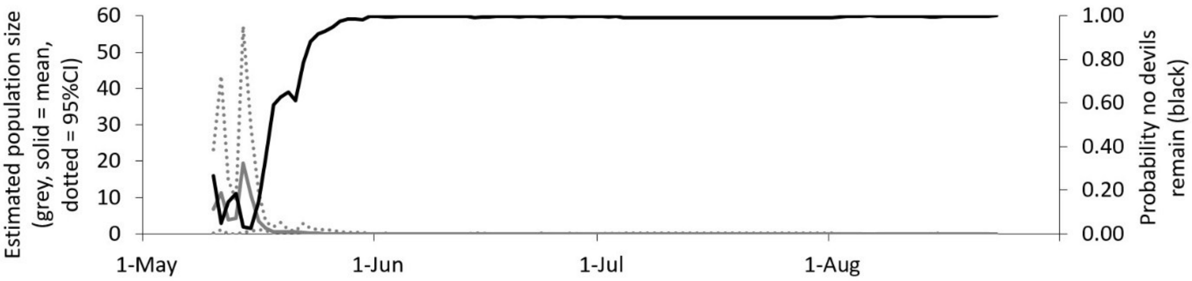
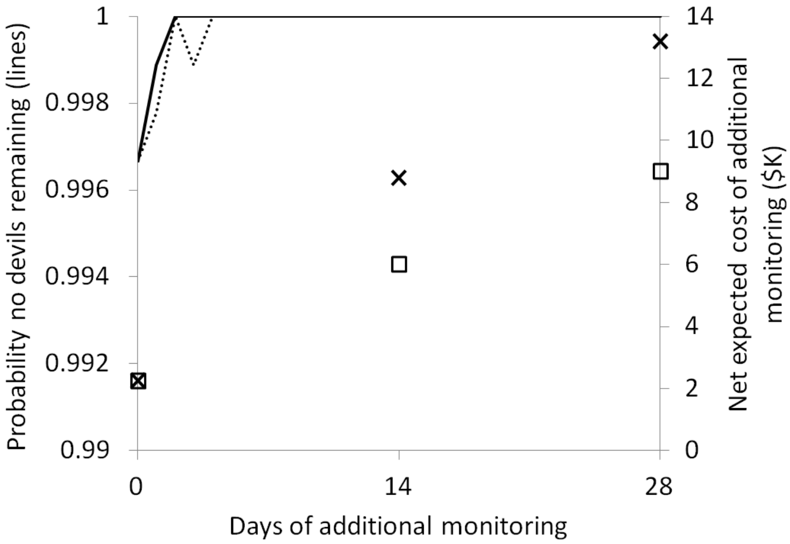


Figure 2: The net expected cost of conducting additional monitoring and the associated probability that no devils remain, under a) a realistic immigration rate, and b) a high immigration rate. Shown for monitoring intensities of 28 cameras (dotted line and square markers) and 58 cameras (solid line

and crosses). Calculated with equation 3, using the baseline estimate for the cost of stopping monitoring prematurely, C_p = AU$676,470.

a)



b)

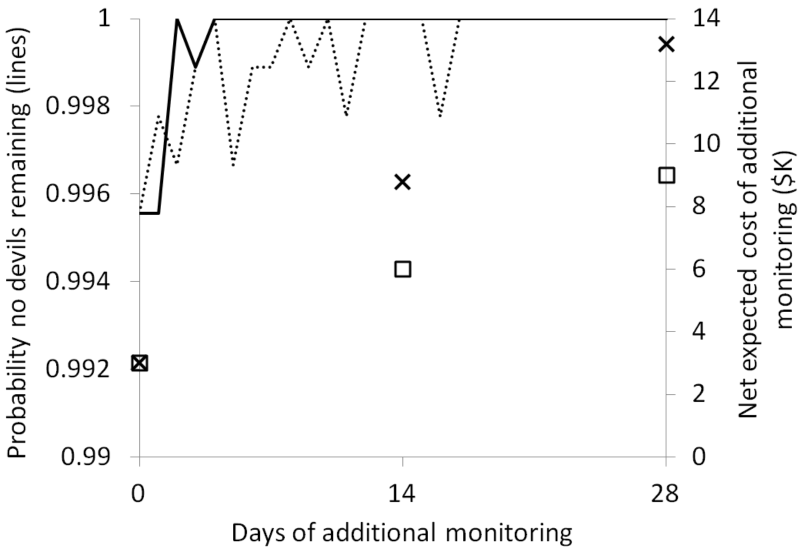
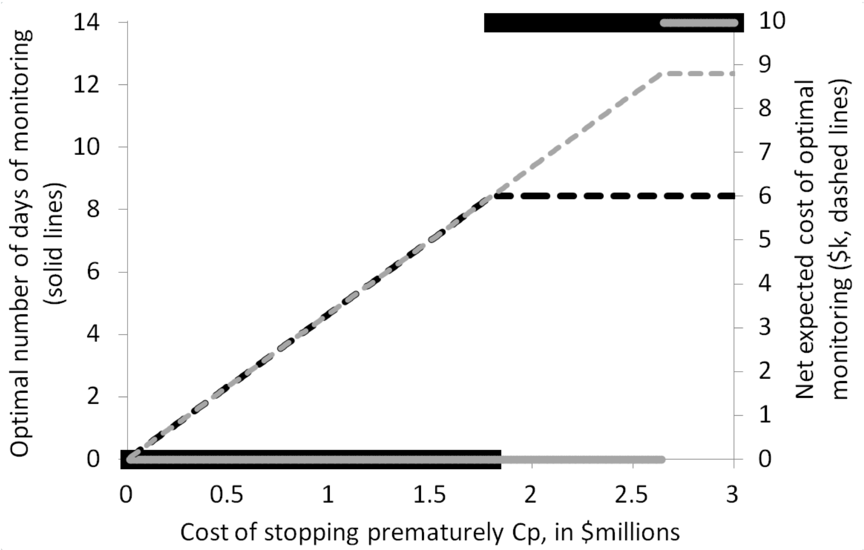


Figure 3: The optimal number of days of additional monitoring and associated net expected cost as a function of the cost of stopping monitoring prematurely (when devils are still present), under a) a realistic immigration rate, and b) a high immigration rate. Shown for monitoring intensities of 28

cameras (black) and 58 cameras (grey). Our baseline estimate for the cost of stopping monitoring prematurely was C_p = AU$676,470.

a)



b)