Data collection requirements to allow analysis and assessment of ungulate control

Envirolink Advice Grant 1095-NLRC145

Landcare Research
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- Northland Regional Council (NLRC) manages several ungulate control projects and in 2012 sought advice from Landcare Research (Envirolink Project 1095-NLRC145) on collection and analysis of data from control operations that can be used to identify optimal intervention strategies.

Objectives

- To describe the particular data collection requirements taken during ungulate management projects that are needed to (a) determine optimal intervention strategies (how often, how intensively to apply control) and (b) stop rules to validate eradication.

Summary

- Data collected during standard ungulate control or eradication projects can be used to answer a variety of management questions. These include (a) the frequency, intensity, and location of ongoing control to optimise intervention strategies, and (b) how many animals are left, including the key question in eradication projects of how to interpret the lack of evidence of survivors and thus claim success and stop the project. Often the answer to this latter question requires data collected much earlier in the project when their immediate purpose is not clear.
- The basic data required include: the number of animals killed or trapped, the effort expended to do this, a description of the area hunted or trapped, the location of the animals killed, the location of animals seen but not killed, and sometimes special-purpose monitoring data from telemetered animals or from search devices such as camera traps or faecal pellet surveys. The development of GPS tools allows these data to be routinely collected by hunters.
- Basic data on the number of animals killed at each control event, control effort and so indices of density such as catch-per-unit-effort can be used to assess trends in population size. With estimates of rates of increase for the species, thecombined information can be used to develop simple models of population size and so harvest rates and thus intervention strategies to either maintain or further lower the current population.
- The development of new analytical techniques allows such data to be used to determine the probability that an absence of animals killed or of their sign at the end of eradication projects means no animals are left, or to prescribe how much more hunting or monitoring is required without finding evidence of survivors, to achieve a nominated probability of success.

Recommendations

- The basic premise is that much spatial information is useful to assess the success of ungulate control projects, but it needs to be collated in the correct context.
- Hunting data (and thus estimation of catch–effort relationships) need to be collected well in advance of later phases of a project to enable adequate analysis of questions relevant to intervention or validation of success.
- We recommend Bayesian models to generate the estimates required to answer these questions.
- Other detection systems (e.g. camera traps, telemetered animals, or surveys for sign) also need to be deployed early during projects so their detection capabilities can be measured while there are still animals to detect.
- Hunts/surveys should be designed to meet model assumptions since more data points (surveys) are required to fit more complex models.
- Survey methods should be standardised within a programme and methodology should be detailed as meta-data associated with GIS files.
- An initial quick validation in a GIS of data collected should be done as they come to hand to:
    - check data have been recorded
    - check they are all in the same spatial projection
    - check for outliers
    - check with a spatial join that each kill point is associated with a track/effort.
- Instances where an animal is sighted but not killed (e.g. missed shot or too many animals in a group to be able to shoot all of them) should still be recorded as waypoints and associated with a track or effort measure.
- Other removals (e.g. landowner culls) should be noted with approximate kill location even though there will be no effort data associated with these kills.

1 Introduction

Northland Regional Council (NLRC) manages several ungulate control programmes and in 2012 sought advice from Landcare Research (Envirolink Project NLRC145) on collection and analysis of control data to identify optimal intervention strategies.

2 Background

Regional councils and the Department of Conservation (and their predecessors) have a long history of controlling ungulates such as feral goats (*Capra hircus*), feral pigs (*Sus scrofa*), Himalayan thar (*Hemitragus jemlahicus*), chamois (*Rupicapra rupicapra*) and deer (Cervidae) in New Zealand, with some projects being among the longest sustained operations in pest management anywhere in the world (e.g. nearly 100 years against feral goats in Mt Egmont National Park; Forsyth et al. 2003). Internationally and within New Zealand the ability of managers to efficiently remove some populations of ungulates has also been increasing (e.g. Parkes et al. 2010; Crouchley et al. 2011).

This growing confidence in the ability to control unwanted ungulate populations efficiently has resulted from improvements across three areas of pest management: improved control technologies or at least improved ways of using old ones, new monitoring tools and data collection methods, and new analytical tools to interpret the data collected to answer questions of interest to the managers.

This Landcare Research review focuses on the last two of these improvements and how such data might be collected and used in typical New Zealand attempts at eradication, extirpation or sustained control of ungulates. It was commissioned by Northland Regional Council under the Envirolink Medium Advice Grant fund of the Ministry of Science and Innovation.

3 Objectives

- To describe the particular data collection requirements taken during ungulate management projects that are needed to (a) determine optimal intervention strategies (how often and intensively to apply control) or (b) stop rules to validate eradication.

4 Ungulate control strategies

4.1 Ungulate control in New Zealand

In New Zealand wild and feral ungulates (the six species of deer, feral goats, feral pigs, Himalayan thar and chamois) are defined as 'wild animals' under the Wild Animal Control Act 1977 and managed as pests, at least in priority places. However, these species are also more (e.g. deer) or less (e.g. feral goats) valued by a range of New Zealanders and overall most of the annual harvest of these ungulates (with the exception of feral goats) is taken by recreational and commercial harvesters (Nugent 1992; Parkes & Murphy 2003). Whether this harvest is in any way effective at limiting their impacts on biodiversity is a moot point (Parkes 2006). Nevertheless, some populations of these animals are controlled specifically as pests by the Department of Conservation and by regional councils and that is the focus of this report.

4.2 Strategic options to manage ungulates

Some wild ungulate populations have limited or patchy distributions and so, in part because reinvasion can be stopped, eradication is a possibility for those populations. The a priori analyses of whether eradication is feasible are assisted by data collected from other similar projects or even earlier phases of projects under review.

For other populations, the rules to achieve eradication cannot be met (Parkes & Panetta 2009) and constraints in each place overcome, so sustained control is the only practical option. A subset of sustained control, termed extirpation, is where the population can be removed but, because reinvasion is certain, sustained effort must be made to keep the density at near-zero. In practice, some long-established sustained-control or extirpation projects might evolve into attempts at eradication (e.g. Auckland Council are currently considering this option for feral goats in the Hunua Ranges) if the immigration problem can be resolved.

For populations that cannot be eradicated the optimal strategy is much more complex. First, some target density of animals has to be set – and usually validated or modified by monitoring.

Generally, for pests that are controlled with methods that have a cost per animal removed (cf. methods such as aerial baiting, which have a cost per hectare treated largely independent of the pest density), the costs per animal increase rapidly as the density of animals decreases (Figure 1). Therefore when budgets are limited there is an opportunity cost in aiming for zero density when the protection goals might be achieved at some higher pest density. In contrast, there are costs if the target density of pests is set too high and the resource is not adequately protected.

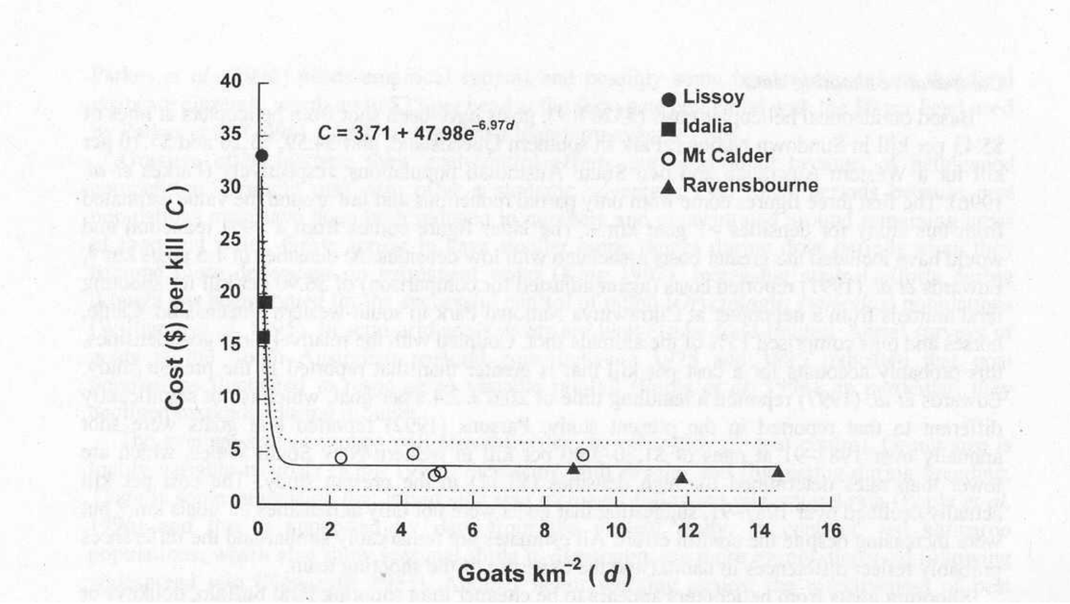

Figure 1 Relationship between feral goat density and the costs to remove each goat by aerial hunting (after Pople et al. 1998). Note: similarly shaped relationships hold for other goat population control by both aerial (Maas 1997) and ground control (Parkes 1993) methods.

So, more usually, the target density is set based on an understanding of the relationship between a pest's density and its impact on the resources we value (Choquenot & Parkes 2001). When a short-term response to ambient pest densities is the issue of interest, e.g. palatable seedling and sapling abundance in the forest understorey, a simple damage function is appropriate. However, where no response is observed despite management or where factors other than the pest are involved, more complex models may be required (Choquenot & Parkes 2001).

For ungulates in New Zealand forests, a damage function model to relate pest (deer) densities to forest regeneration seems to be adequate in many cases. Nugent et al. (2001) have suggested there are real thresholds in the function for palatable species. For example, deer living in forests rely on canopy leaf-fall for much of their diet and so there is no immediate feedback between food abundance and deer density as they cannot affect this food source in the short term. The consequences are that no plants more palatable than canopy foods regenerate in the understorey (other than in inaccessible sites) until deer densities reach very low levels. Thus the threshold deer densities at which these palatable species do survive can become the necessary target for control. However, this damage-function relationship does not always apply in forests and the effects are not always reversed when browsing pressure is reduced by control of ungulates (Coomes et al. 2003; Tanentzap et al. 2009). For example, in some places the ungulate-induced changes in forest understorey have led to irreversible changes in the soil chemistry that preclude simple reversion to the original palatable communities (Wardle et al. 2001).

For ungulates living in grassland or alpine habitats the relationship between an animal's density and its impacts is likely to be more linear, i.e. without clear thresholds (e.g. Parkes & Thomson 1999). This is because the animals have more-or-less permanent access to their food, i.e. the buffering effect of leaf-fall from canopy trees seen in forests is absent.

Despite these complexities, the logic of management–ungulate–vegetation interactions from a monitoring perspective is that:

- Impact declines in some way as ungulate density declines.
- Ungulate density declines with harvest or control.
- Ungulate rates of increase (i.e. recovery after control) increase towards their maximum (intrinsic rate of increase) as the per capita food supply increases with a decline in ungulate density (Caughley 1977).
- The proportion harvested increases in some way as the control effort (intensity and frequency) increases, and the cost to do this is known at all densities.

If this chain of logic (Parkes 1993) is understood, managers can base some planning decisions on the simplest and cheapest input parameters, e.g. budgeted costs, hunting effort or total harvest sizes.

Measuring the post-control or residual densities or the percent change in density is the next step along the logic chain. Changes in catch per unit effort (CPUE) can be used as an index of population change and, with some analysis, to estimate residual densities (e.g. Brennan et al. 1993; Forsyth et al. 2003).

Finally, the effect of the control action on the impacts of the animals can be measured as an outcome that is agreed upon at the outset of the control programme. For ungulates in New Zealand forests such methods vary from relatively simple indices (e.g. the change in seedling ratio method; Sweetapple & Nugent 2004), to measures of short-term responses in places where a quick response is predicted such as light gaps in the forest (Sweetapple & Burns 2002), to long-term trends using permanent vegetation plots and exclosures (e.g. Husheer 2007).

In this report we are interested in the questions that can be answered from data collected during the control operation itself. Our aim is to identify these questions and specify what sort of data are required to answer them. The caveat we place on ourselves is we consider only control operations that result in a known number of animals removed for known effort, i.e. ground-based and aerial hunting and trapping. These are, in any event, the main control methods used against wild and feral ungulates in New Zealand.

4.3 Operational phases

Most ungulate management projects aim to achieve their goals, whether eradication, extirpation or sustained control, by applying a series of control events (hunts, culls, etc.) to reduce the population and then periodically to maintain it at or below the target density.

It helps in planning, conducting, and monitoring to see this process in phases (albeit with fuzzy transitions) that vary depending on the strategic goal (Table 1).

Table 1 Phases of typical ungulate control projects according to the goal of the control

Eradication	Extirpation	Sustained control
Initial knockdown	Initial knockdown	Initial knockdown
Mop-up survivors	Mop-up survivors	Ongoing harvest
Validation of zero	Surveillance and reaction	Validation that impact is mitigated

4.3.1 Initial knockdown

The first task under all strategies dealing with an established population of ungulates is to reduce their number – as quickly as possible to minimise the 'lost ground' as survivors breed (Table 2). It is also best to use methods that leave the least survivors from each encounter and so avoid problems caused by wary animals that have learnt to avoid the control methods. Information on the extent of the target population – a delimitation survey – is desirable either before the initial phase begins or as part of it.

4.3.2 Mop-up of survivors

At some point in the project the population is reduced in numbers and often in distribution. Sometimes wary survivors are a factor, or the last animals may be living in difficult-to-reach terrain/vegetation, or in unexpected places. A change of tactics is often required to remove (mop-up) the last few animals – in part because of this changed behaviour of the animals but also in part because the cost per animal removed can be a major factor (see Figure 1).

4.3.3 Ongoing harvest

In sustained control strategies, an ongoing harvest or cull of animals is required to maintain the population below the target density. The frequency and intensity of these harvests depends on the rate of increase (in situ breeding and, if present, immigration). For the ungulates in New Zealand this varies from about 0.27 for red deer to about 0.7 for pigs (Table 2). To maintain or reduce ungulate density, the rate of removal must be equal to or greater than the population rate of increase, so larger offtakes are needed for highly productive species such as pigs.

Table 2 Rates of increase due to breeding of typical New Zealand wild and feral ungulate populations. Other species of deer, thar, and chamois normally produce one offspring per year and have similar rates of increase to red deer

Species	Intrinsic rate of increase (*r*)	Finite rate of increase (λ)	Doubling time (years)	Reference
Feral pigs	0.7	2.0	1.0	Choquenot et al. 1996
Feral goats	0.425	1.53	1.6	Parkes et al. 1996
Red deer	0.27	1.3	2.6	Caughley 1977

4.3.4 Validation of zero

A difficult issue in the eradication strategy is knowing when success has been achieved and none of the target population is left. This is especially so for ungulate eradications achieved by a succession of culls when there is a cost to remobilise the hunters if success is falsely declared.

At some point it is hoped that the project manager will believe that the population has been eradicated, and a validation phase is required to demonstrate that belief. The validation phase ideally provides the manager with a probability that the lack of further detection or animals killed equals no animals present, and this can sometimes be provided using the data collected during the control phases. Further validation may be required in which a prescribed amount of ongoing monitoring is conducted to increase the probability of eradication to a value that is acceptable to decision makers (e.g. Ramsey et al. 2009, 2011).

4.3.5 Surveillance and reaction to immigrants

In the extirpation strategy, where immigration or deliberate introduction cannot be ruled out, the validation phase may change to an ongoing surveillance programme with the capacity to react (i.e. by restarting the initial knockdown and/or mop-up phase) if new animals are detected.

4.3.6 Validation that impacts are mitigated

It is desirable to assess the effects of the management on native biodiversity under all strategies, but it is only under the sustained-control strategy that this is essential – to additionally validate the target densities. Under the eradication or extirpation strategies the only management responses that might accrue from monitoring mitigation of impacts are (a) that nothing changed and the target pest was not the cause of the problem (Caughley 1994) or (b) the system had changed irreversibly (Coomes et al. 2003) or (c) the problem got worse because either the target pest was not the real problem, as above, or its removal allowed another pest or weed to increase, or (d) any recovery is simply slower than the timeframe between control and monitoring. If this happens the control could be stopped and the system left to revert to its original state (if possible), or the other pest/weed may also need to be managed.

5 Data required to answer management questions

5.1 What sort of data can be collected during a control operation?

Hunting and trapping control projects have long collected simple data such as the number of animals removed and the hunting or trapping effort (e.g. Brennan et al. 1993; Forsyth et al. 2003). However, the recent availability of GPS and telemetry systems and GIS tools to map activities and results has allowed more sophisticated questions to be answered using new analytical tools.

Table 3 Data that can be collected during typical ungulate control projects and its use

Data	Typical unit	Derived statistic	Used to answer
Hunting effort (A)	Hunter-days; flying hours; distance travelled per area searched (km/km^2)	CPUE (B/A) – index of abundance	How many? (relative abundance)
Trapping effort (A)	Trap-days	CPUE versus cumulative kill (∑B)	Sensitivity of detection: How many? (residual population size) True negative? (validation of eradication) How much effort? (to be confident none remaining)
Numbers killed (B)	Number (by sex, age, fecundity)		
Numbers known to survive	Number seen but not killed	Minimum number known to be alive	How many? (residual population size)
Hunting/trapping coverage	GPS tracks or coordinates	% of area searched/treated	Adequate spatial coverage?
Location of kills	GPS coordinates	Distribution of animals across landscape	Where? (population delimitation)
Judas animals	GPS location and no. of wild animals with it	Location Probability of association	Where? Sensitivity of detection
Camera traps	Photographs per unit of time Indivduals per unit of area Presence/absence	Index of abundance Density (MR*) Occupancy	How many? (relative abundance) How many? (density) Where?
Field sign (faeces, tracks, pig rooting)	Presence/absence Counts of sign per area of sampling unit Proportion sampling units with sign	Occupancy Index of abundance	Where? How many? (relative abundance)
DNA analysis (faeces, hair folicles, tissue, blood)	No. of individuals in sample Indivduals per unit of area	Minimum number known to be alive Density (MR*)	How many? (residual population size) How many? (density)

*MR = estimated using Mark-Recapture analysis methods

5.2 Catch per unit effort (CPUE)

Catch-per-unit-effort data can be used as an index of population density (e.g. the residual trap-catch index for possums; NPCA 2011) and thus a measure of control efficacy for initial knockdown or ongoing control, i.e. proportion killed = 1 – (*Post-control Index*/*Pre-control Index*). As with any index method, the assumptions are that the coverage is the same (the same population is sampled) and that the probability of catching an individual per unit of effort does not change with population density or for some other reason between the two sampling periods. Here we assume that a 'catch' means the animal is removed from the population so the probability of capture is equivalent to the probability of detecting *and* killing an individual.

Statistical methods using CPUE data called 'catch–effort' models or 'removal' models can simultaneously estimate the probability of detection per unit of effort and the initial population size, thus enabling calculation of the residual population size (e.g. Barron et al. 2011). This is useful for measuring progress towards threshold density goals and combined with the knowledge of detection probability informs how much effort is needed to achieve those goals.

Estimating the probability of catching/detecting an individual per unit of effort expended is valuable for the validation (in the case of eradication) or surveillance (in the case of extirpation) phase because it enables calculation of the overall sensitivity of the surveillance method. This is required to interpret zero catches obtained in these later phases, i.e. does a zero indicate there are no animals left ('true negative') or you just didn't happen to catch one ('false negative')? By definition, sensitivity is the probability of detecting an individual given there is one present, so hunting/trapping effort and coverage and kill data should be collected in the initial knockdown phase while there are still animals present.

It is important to have a standard measure of 'catch', at least within each area/project. This is not always straightforward if, for example, some animals are killed but the kill is not confirmed. For example, some goats may be killed away from the hunter by hunting dogs, and this may vary depending on whether indicator or hard-bailing dogs are used. Most contract hunters will confirm a kill either by taking a token (the tail) or by noting it when they are certain the animal is killed but cannot reach the carcass.

It is equally important to have a standard measure of 'effort' within each area/project. In the past ground hunters used the unit of a 'hunter-day', being a day on which any active hunting occurred. However, with the current use of personal GPS units the effort data can be logged more accurately as hours-hunted or distance travelled using GPS tracks of hunters and/or their dogs (e.g. Figure 2). Ramsey et al. (2009) used GPS tracks of hunters to calculate track density (the kilometres walked per area (km^2) searched) in a feral pig eradication project on Santa Cruz Island, California. In many ways this parameter is more useful than a simple estimate of hunting effort measured as time spent hunting, as it provides a direct measure of ground covered, which may vary with differences in terrain or vegetation.

The older metrics of effort may need to be retained and used to calibrate against new metrics, at least for a few years. The hours spent flying while actively hunting is of course a simple measure of effort in aerial hunting control and can be compared over time, provided similar helicopters are deployed.

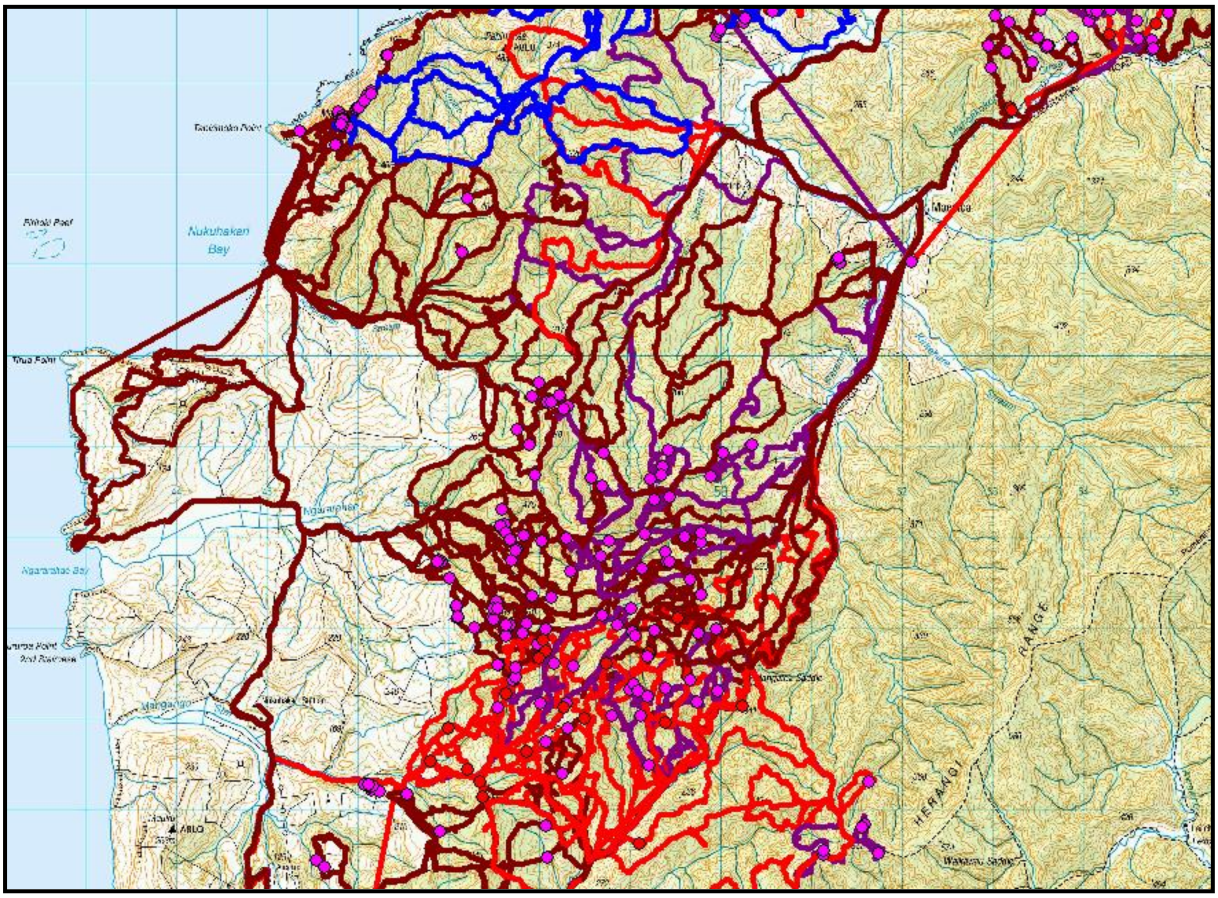

Figure 2 Example of ground hunters' search tracks (red, brown & blue lines) in a feral goat control operation showing location of kills (pink-filled circles). Here the effort was stratified with most in areas known to have goats (data from Backcountry Contracting).

5.3 Coverage of control

Eradication and extirpation require all animals to be placed at risk and the catch–effort models assume the risk of catch is equal for all individuals. In other words, the spatial distribution of hunting/searching effort must be consistent between hunts/searches in order to make inference on the same population. Proportional spatial coverage of a control area can be assessed using GPS tools by placing a 'buffer' or search-swath width either side of the hunter tracks, calculating the area covered by these buffered tracks then dividing this by the area of interest. The size of this buffer, the effective search width of a hunter's track, has not been quantified, and is likely to vary among habitats. Barron et al. (2009) assumed a 150-m-wide buffer either side for ground hunters with dogs looking for feral pigs and calculated spatially consistent and moderate proportional coverage (>0.6) of most hunting zones with each hunting effort with the exception of two very steep zones. Plotting proportional coverage against track density showed an asymptotic exponential relationship indicating the assumption of a random distribution of hunting effort for the catch–effort analysis had been satisfied.

5.4 Location of animals

An essential part of all eradication projects is to delimit the range of the target population. Of course if this exceeds the area where control is possible, then immigration is likely and eradication not possible.

There are several ways to do this, such as a priori surveys of sign (e.g. Parkes et al. 2011), indications from radio-telemetered animals (see below), and mapping of kill and sighting locations as the area is hunted.

Most animals are not uniformly distributed across a landscape; they prefer one habitat type over another. Understanding these preferences will enable more targeted surveillance in the later eradication or surveillance stages. By stratifying the landscape into areas of high and low risk of animal occurrence and allocating more surveillance effort into high risk areas, overall surveillance sensitivity is enhanced (e.g. Anderson et al. submitted). Again this information needs to be collected while animals are still abundant, to be able to characterise their distribution, and makes the assumption that distribution does not change with population density.

5.5 Population size

While the CPUE methods focus on estimating residual abundance through sequential removals of animals, animal abundance can also be indexed using non-lethal methods for the purposes of: identifying if some threshold density has been exceeded in the case of sustained control; detecting new incursions in the case of extirpation; detection and mop-up of survivors in the early stages of eradication and extirpation projects; and validation of zero in the end stages of eradication projects (provided the sensitivity of detection is known). Non-lethal methods include field sign (faeces, tracks, or rooting disturbance) and observation (camera traps, aerial surveys) and these vary in costs, ease of use, duration of monitoring, and detection abilities. Wilson and Delahay (2001) provide a good overview albeit for carnivore monitoring. Most of these methods provide an index of abundance rather than actual population size so caution must be used when comparing indices in time and space since the probability of detection may also vary with time and space. However, methods that identify individual animals, such as camera traps and genetic analysis, can also be analysed with mark–recapture techniques to provide unbiased population estimates. Recently developed analytical methods can even estimate population density from camera traps without the requirement for individual recognition of animals (Rowcliffe et al. 2008). A novel approach to estimating residual abundance using DNA sampling is described by Parkes et al. (2011). Essentially this involves collecting DNA samples from animal sign such as faecal pellets prior to control to obtain individual genotypes, conducting lethal control then, using DNA recovered from the culled animals, estimating the number of animals that were not recovered in the control by comparison with the pre-control genotype sample.

5.6 Telemetered animals

Individuals with GPS and/or VHF transmitters are now commonly used in ungulate control projects. They have three purposes:

1. To measure home ranges and habitat use.

2. To find and kill wild animals associated with the radio-collared (Judas) animal.

3. As a surveillance tool to validate eradication.

The detection probability of a Judas animal can be estimated during the time when wild animals are left and by assuming that one or more of the Judas animals are wild ones and seeing how often (and over what distances) the Judas animals detect each other. For example, Ramsey et al. (2009) found that telemetered pigs on Santa Cruz Island had a 20% chance of being found with another telemetered pig living within the same range within 120 days, but only a 1% chance when the two pigs lived far apart. A more explicit detection probability analysis is currently being done using the hundreds of telemetered feral goats left as surveillance animals on Islas Isabela and Santiago in the Galápagos Islands (Ramsey et al. in prep.).

5.7 Age and sex structure of sampled population

Recording sex, age, and breeding status of hunting kills can indicate the timing and rate of population recruitment, which is useful for planning the timing of operations and in the analysis of kill data. Biases in the sex or age of kills can give some clues to the remaining population's stability and viability (Langvatn & Loison 1999), but it is difficult to determine if these biases are due to the control method preferentially sampling one age/sex class over another or if the bias is indeed due to underlying changes in the residual population.

6 Analytical options

The catch–effort or removal method has been applied to a variety of vertebrate control programmes such as the pig eradication programme on Santa Cruz Island (Ramsey et al. 2009) and the eradication of cats from San Nicolas Island (Ramsey et al. 2011). The method and its assumptions are described in Appendix 1. The method simultaneously estimates the initial population size in the area searched and the probability of detecting an individual animal for each unit of search/hunting/trapping effort and can be used to answer questions about how many animals are left, are there any animals left, and how much monitoring should be done to validate eradication (see sections 5.1–5.3).

We recommend the use of Bayesian statistical methods to fit the catch–effort functions described in Appendix 1 because they enable prior information, for example likely population size, to be incorporated into the analysis. Because parameters are fitted as probability distributions (rather than just point estimates), uncertainty in parameter estimates can be incorporated in the analysis, and is propagated through to the final metric. Also, the ability to specify different process and observation error distributions means that relatively complex functions can be fitted without the need for restrictive assumptions such as normal error distributions.

6.1 How many animals are left?

The estimate of the initial population size provided by the catch–effort function minus the number known to have been killed provides an estimate of how many animals are left, albeit under the same assumptions of the catch–effort model. This approach was used by Barron et al. (2011) to estimate the efficacy of pig hunting across eight different extirpation zones on the Hawaiian Islands.

6.2 Validation of eradication - are any animals left?

Once hunting or monitoring no longer detects any target animals we can use Bayes' theorem (eqn 3, Appendix 1) to estimate the probability that the target population has been eradicated (Ramsey et al. 2009). This formula uses the surveillance sensitivity or probability of detecting an animal for the level of effort expended (estimated from the catch–effort function) to evaluate the likelihood that the failure to detect an animal is due to a true absence rather than just a failure to detect one.

6.3 How much monitoring should be done to validate eradication?

This question can be answered using a variation of the above methodology where the probability of eradication is estimated for a range of hypothetical hunting/monitoring efforts and the effort value where the probability of eradication exceeds some predetermined threshold identifies the minimum required monitoring effort (conditional on not actually detecting anymore animals) to be confident eradication has been achieved.

6.4 Data specifications

Catch–effort models require catch and effort data collected in the initial phases of the control programme while there are still plenty of animals available to be caught. There need to be multiple sequential removal events (hunts), each of which then provides a data point to fit the catch–effort model. The catch–effort method has certain assumptions (Appendix 1), such as all individuals having the same chance of being caught on a single occasion, which ought to be incorporated into the survey design, for example by ensuring the same spatial coverage of the area with each survey. While it is sometimes possible to account for violations in model assumptions during the analysis phase, e.g. by adding covariates to the catch–effort function, this involves fitting extra parameters, which requires more data points, so each approach has its cost.

Handheld GPS technology has greatly improved the measurement of hunting or search effort and several metrics can be derived from recorded hunter's tracks such as hours hunted, track density, and proportional coverage. However, if there are historical catch–effort data available then the old metrics (such as time hunted) should continue being recorded concurrently with the new ones to see if some relationship between the old and new metrics can be derived enabling the use of the old data. Catches (kills) need to be assigned to some unit of effort expended. In practice, since effort is often measured using hunter or dog tracks and kill as a waypoint in a GPS, the kill point needs to be annotated with a track identifier. Where kills cannot be assigned to a measure of effort, e.g. if a landowner adjacent to the control zone kills a target animal, then this should still be recorded, with an approximate kill location, to be taken account of in the analysis. Similarly where animals are sighted but not killed, this should be recorded as a way point and associated with hunting effort (track ID). If this happens frequently, 'catches' in the model could be broken down into detection and kill (conditional on detection) probabilities. The downside of GPS technologies is that they generate much data which need to be downloaded, checked, tidied, annotated, and stored. It is best to do a quick check that the correct data have been recorded as they come to hand, so any omissions or mistakes can be rectified while people can still remember the details.

7 Recommendations

- The basic premise is that much spatial information is useful to assess the success of ungulate control projects, but it needs to be collated in the correct context.
- Hunting data (and thus estimation of catch–effort relationships) need to be collected well in advance of later phases of a project to enable adequate analysis of questions relevant to intervention or validation of success.
- We recommend Bayesian models to generate the estimates required to answer these questions.
- Other detection systems (e.g. camera traps, telemetered animals, or surveys for sign) also need to be deployed early during projects so their detection capabilities can be measured while there are still animals present to detect.
- Hunts/surveys should be designed to meet model assumptions since more data points (surveys) are required to fit more complex models.
- Survey methods should be standardised within a programme and methodology should be detailed as meta-data associated with GIS files.
- An initial quick validation in a GIS of data collected should be done as they come to hand:
    - check data have been recorded
    - check they are all in the same spatial projection
    - check for outliers
    - do a spatial join to check each kill point is associated with a track/effort.
- Instances where an animal is sighted but not killed (e.g. missed shot or too many animals in a group to be able to shoot all of them) should be recorded as waypoints and associated with a track or effort measure.
- Other removals (e.g. landowner culls) should be noted with approximate kill location even though there will be no effort data associated with these kills.

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Appendix 1 – Description and assumptions of catch–effort functions

The probability of detecting (and killing) an animal (θ) for a given level of hunting effort assuming a Poisson process is:

$$\theta_i = 1 - \exp(-\rho H_i)\ , \qquad \text{eqn 1}$$

where H_i is the hunting effort in period i and ρ is the Poisson catchability coefficient. If each animal has the same probability of detection during a hunting period θ_i, then the number removed during that period, n_i, will be binomially distributed:

$$n_i \sim \mathrm{Binom}(\theta_i, N{-}x_i)\ , \qquad \text{eqn 2}$$

where $N{-}x_i$ is number of animals available being the initial population size (N) reduced by the cumulative number of animals killed to date (x_i). To account for extra unexplained variation in the detection probability at each sampling session a lognormally-distributed error term is added to each estimate of θ_i. The method assumes (Seber 1982) that:

1. The population is closed, i.e. the population doesn't change in size due to births, deaths, or dispersal over the sampling period (except for the known number purposefully removed).

2. The per-unit-of-effort catchability coefficient ρ is constant throughout the sampling period, is the same for each individual and the units of effort are independent.

3. All individuals have the same probability θ_i of being of being caught in the ith sample.

Violation of these assumptions can introduce bias into the parameter estimates; e.g. if immigration is constantly occurring, the catchability coefficient ρ may be underestimated. Ideally any sources of bias should be minimised by the experimental design rather than post-hoc analysis but often this is not possible and instead the basic model described above must be modified. Observer/hunter effects and habitat effects can be included as covariates in the detection function (e.g. Chee & Wintle 2010) to account for differences in detectability due to differing hunter skills and experience or the effects of vegetation type on detectability. The assumption of a closed population may be violated if there is the possibility of immigration into the control area occurring or opportunity for the population to breed between control operations. If some prior information on the likely rates of these events occurring is available, then it should be possible to incorporate an underlying population change model into the basic framework. In social ungulates such as goats the assumption of independent and equal probability of capture is likely to be violated because herds of goats are grouped in space so that upon finding one goat the likelihood of finding another nearby is high. Spatial autocorrelation between capture rates due to animals being aggregated in space can be incorporated as a covariate in the detection function (e.g. Nishida and Chen 2004). An alternative approach might be to assume the animals follow a Negative Binomial (aggregated) distribution rather than a Poisson distribution. Another way the equal catchability assumption is likely to be violated is if there is variation within the population in terms of their susceptibility to being caught, e.g. naïve versus wary individuals. Mäntyniemi et al. (2005) describe a Bayesian model where the catchability of individuals' ρ decreases with the total number of animals removed to simulate the naïve animals being picked off first. A different approach was taken by Barron et al. (2011) who used a Weibull detection function so that the probability of detection per sampling period, θ, changed with cumulative hunting effort. In this case they found that detection actually improved with cumulative hunting effort, which they attributed to the hunters becoming more experienced with time.

In summary, it is possible to account for some violation of the model assumptions, but to do this requires estimation of at least one, often more, extra model parameters, which in turn means the number of data points (or degrees of freedom) must be sufficient to estimate all of the model parameters. A related problem we have encountered when fitting such models is that when there are few data points, unless a very constrained prior value for the initial population size N is used, the models will not converge since it deems a low catchability coefficient ρ and a high N is just as likely as a high ρ and low N, i.e. the two parameters become negatively correlated. In this case it might be better to fit the detection-effort function independently of the actual surveys. This has been done for stationary targets such as goat sign (faecal pellets) using multiple searches of the same area by three observers (Parkes et al. 2011) and for invasive weeds by seeing what proportion of 'planted' hawkweeds observers could find in a sampling area (Moore et al. 2010). Using the former approach for a mobile target would probably not work as the first observer would alter success for the second, but a variation of the latter has been attempted for pigs in Australia by firstly radio-collaring a proportion of resident pigs then seeing how many of these pigs a subsequent hunting effort found (McIlroy & Saillard 1989). Something similar could be done with Judas goats by locating them earlier in the day then letting hunters (unaware of the Judas locations) search the area to see what proportion they detect for a given search effort.

The probability of one or more target animals persisting in the control area given none were detected (and the amount of effort expended) can be calculated using Bayes' theorem:

$$f\ \pi^{+}\ D^{-}\ = \frac{f\ \pi^{+}\ f\ D^{-}|\pi^{+}}{f\ \pi\ f\ D^{-}|\pi} \qquad \text{eqn 3}$$

where,

$f(\pi^{+})$ is the 'prior' probability or belief that the some animals persist (see below).

$f(D^{-}|\pi^{+})$ is the probability an individual is not detected given some are present (this is what we estimate from the capture–effort model, and is calculated as one minus the detection probability θ for a given level of effort H).

$\sum f(\pi)f(D^{-}|\pi)$ is the 'marginal' probability, which covers all possible reasons for not detecting an individual – in this case because there are none left or there are some left but they were not detected.

$f(\pi^{+}|D^{-})$ is the 'posterior' or revised probability that one or more target animals are present given none were detected (this is what we are trying to estimate).

Prior probabilities of population persistence can be derived from expert opinion based on the outcomes of similar programmes or deliberately left 'vague' 'or uninformative' so that the posterior probability is determined entirely but the fitted sensitivity (catch–effort) function. Priors are updated over time as new monitoring information comes to hand so, for example, the posterior probability of population persistence calculated in one year becomes the following year's prior. If present, the risk of introduction from adjacent areas can also be incorporated into the annual updating of the priors (e.g. Anderson et al. submitted).