An Analysis of Methodologies used to study Medium and Large Mammals in the Valdivian Temperate Rainforests of central-southern Chile.

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A thesis submitted in partial fulfilment of the requirements for the degree of Master of Science and the Diploma of Imperial College London.

MSc/DIC Conservation Science

September 2008

ABSTRACT

This project was undertaken with the Darwin Initiative Project currently in progress in the Valdivian temperate rainforest of southern Chile, in association with Pontificia Universidad Católica, Chile. This report focuses on the cost-effectiveness of two data collection strategies surrounding threatened species in the region. Their objective is to establish occupancy parameters. The principal data collected, examined and discussed, was collected at 58 camera trap sites with fieldwork spanning three seasons (2007-8). The cost-effectiveness of the limited snow track data has also been examined and compared with that of systematic camera trap data. The latter was analysed using the occupancy software program PRESENCE which indicated the reliability of the existing data. Based upon this factor, key conclusions have been drawn: further camera trap data is required and the data collected so far is insufficient for accurate analysis. The camera trap detection probability was low, suggesting that additional camera trap sites should be included in the future. Given the limited forest fragments and continuous forest in the region, data capture has concentrated on the moist, south facing slopes rather than warmer, drier north facing ones. Budget permitting, up to 60 independent sites could be surveyed each season, extending to north facing slopes and higher altitudes to establish the full range of the species present. The operating costs of camera detection contrasted from £50 (summer 2008) to £75 (winter 2007). Initial data from snow track transects offers an exciting cost-effective alternative for winter data collection. For the 64km surveyed thus far the cost per hit compares favourably at £11. Covariate analysis indicates that occupancy probability for culpeo fox is affected by fragment size, i.e. fragments <100ha are more likely to be occupied than those >100ha. Data for guiña was insufficient as the covariate information did not improve model fit, with the constant model being ranked highest. The high levels of camera trap misfires in winter strongly suggest that the use of snow transects, when feasible, would be more effective.

ACKNOWLEDGEMENTS

I would like to express my sincere thanks to all those that contributed to my MSc project. To the members of Fauna Australis (Pontificia Universidad Cathólica, Santiago, Chile) and Macaulay Institute (Aberdeen, U.K.) – Dr. Cristian Bonacic, Jerry Laker, Nicolás Gálvez; Felipe Hernándeu, Robert Petitpas, Tomás Ibarra and Omar Ohrens who supported me during my time in Chile. To Dr. Marcus Rowcliffe (ZSL) and Dr. E.J. Milner-Gulland (Imperial) for their supervisory and advisory support and to Joseph Smith (Imperial) for all of his advice during my project. I would also like to thank my family, friends and peers who supported and advised me throughout my project. Lastly I would like to thank all those collaborators involved in the Darwin Initiative Project with which I was involved in Chile.

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GLOSSARY

sampling period = the time between visits to a camera station

sampling occasion = a block of trapping days of a set length (i.e. 10-day sampling occasion is a single occasion covering 10 trapping days)

survey (period) = a survey season season

forest site = a geographic named area (forest) where one or more cameras are placed

camera site/station = a specific location of an individual camera

hit = a known detection of an animal with a photographic or print/track record

misfire = a photo (frame) with no sign of an animal, no detection

latency-to-detection = the time between camera deployment and the first detection

detection history = the binary data of hits/non-detections from which occupancy parameters are estimated

detection probability = p = the probability of a minimum of one individual of the target species during a single sampling occasion given the species is present

occupancy probability = psi = the probability of a site being occupied by the target species

trap = camera trap

(track) transect = snow track transect

raw detection rates = the proportion of sites with a detection of the target species from a single survey using a particular method

ABBREVIATIONS

SF = small (forest) fragment

LF = large (forest) fragment

CF = continuous forest

SO = sampling occasion

AIC = Akaike Information Criterion - a goodness-of-fit measure for an estimated statistical model, where the lower the value, the better the fit

ΔAIC = variation in AIC values between a covariate model and the highest ranked model (0.00 ΔAIC)

p = detection probability

ψ = occupancy probability or psi

K = sampling occasion

s = sites (camera stations)

TS = total sampling (*K* * *s*)

S.E. = standard error

[Overall word count: 17,494]

1. INTRODUCTION

1.1. The need for and importance of this research

The fauna of the Valdivian temperate rainforests of the southern Chilean Andes have been little studied. Many native mammals that inhabit these forests are rare and elusive and therefore difficult to detect, measure and monitor. The dense nature of the forests' vegetation and the characteristically steep gradients make many survey methods very difficult to use. Climatic and altitudinal factors play a major part in surveying as the annual temperature ranges between 30°C and -10°C and can drop further still on the higher slopes. In winter, a strong wind ('puelche') from the Andean Cordillera can hamper survey conditions as can rain which, in the author's experience, continues for days. There is a need to assess the population status of the native mammals found here, especially 'vulnerable' species such as *Leopardus guigna*. This surveying could also benefit from various sampling methods, beyond camera trapping, including systematic animal track transects and scat collections.

Little is known about the effectiveness of different data collection methods in general, particularly in this region of Chile. As will become apparent, different methodologies are more relevant to different seasons and in particular autumn/winter when there is frequently inclement weather.

Greater knowledge of the fauna, its preferred habitat and range is also vital for future protection of the local habitats. At present, the region's national parks and nature reserves are predominantly over 1,000m. This trend is reflected throughout Chile possibly because land below this altitude is often owned by local landowners and farmers and used for agriculture. CONAF is an organisation similar to the U.K.'s Forestry Commission, responsible for planting, maintaining and protecting forestry reserves and for plantation/logging operations. Within the National Parks, they work with the rangers who operate exclusively there. The forestry reserves do not have rangers whereas the wildlife reserves/national parks do. Local conservation workers have intimated that a broader Biosphere reserve may be established within this region in the future (Laker, 2008d - personal communications).

This research is integral to a component of the Darwin Initiative funded project no. 15006 ('Darwin Project') being carried out in the study area, relating to habitat use by endemic mammals and birds. For photographs illustrating various aspects of the project, see Fig. 1. The project's purpose is to establish new public-private conservation partnerships, focusing on ensuring habitat connectivity in Chile's Valdivian temperate rainforests. The work supports Chile's National Biodiversity Strategy objectives (Hester *et al.*, 2006).

Chile's National Biodiversity Strategy and Action Plan (CONAMA, 2003) mentioned that national parks and reserves are predominantly on upland sites rather than at various altitudes for biodiversity. This was also highlighted by Armesto *et al.* (1998) who showed that in the most biodiverse areas, over 90% of protected areas are higher than 600m in the Andes, often on active volcanoes, increasing the risk of catastrophic devastation of the local flora and fauna. Armesto *et al.* (1998) also mentioned that these areas are in the southerly latitudes (>43°) missing the highest species richness found in the more northerly regions (35.6°-41.3°S, where the study area is located). Armesto *et al.* (1998) also found other disparities in the coverage of protected areas versus biodiversity. The former is not protecting the latter, even though protected land in the temperate forest region (29%) is the highest percentage of any South American country. Also, in the areas of highest species richness where there are few if any protected areas, one finds the highest human population density and land use. These introduce exotic, invasive species; pollution; logging; hunting; land/soil degradation; habitat fragmentation/isolation and fire to this anthropogenic matrix. In consequence, there is an urgent need in future to focus on 'peripheral areas' found on lower slopes, fragmented forest and connectivity between these and similar or larger protected areas, in the latitudes with the highest biodiversity.

1.2. Aims and Objectives

To gain a greater understanding of the methodologies suitable for studying fauna of Valdivian temperate rainforests.

To assess any preliminary ecological data and evaluate the relative effectiveness of previously used data collection strategies within the Valdivian rainforests.

To ensure the data collected is consistent and comprehensive, enabling researchers to undertake in-depth analyses.

To collect, consolidate and analyse existing data and collection methods.

To determine which methods are appropriate in which seasons and establish the optimum frequency of data collection.

To determine which data capture methods are the most cost-effective based on the limited nature of conservation budgets.

To determine future data collection methods in light of ongoing surveying in this region and previous research in other parts of the world.

1.3. Initial research questions

These were not all fulfilled, due to weather constraints:

1. Are guiña present in the national parks/over 1,000m during winter?

2. Is the probability of finding guiña the same over 1,000m as it is lower down (c. 600m to 800m) in winter?

3. On the lower slopes (*c.*600m to 800m) are camera traps or snow track transects a more effective way of detecting species such as guiña?

1.4. Null Hypotheses

1. Guiña (*Leopardus guiña*) are as abundant on the higher slopes in winter as on the lower slopes of mountains/hills.

2. Tracks are a better survey method on higher slopes than camera traps in winter whereas there is no difference between tracks and camera traps on the lower slopes.

3. There is no correlation between the occupancy of native and introduced species.

As will become apparent in this report, several of the hypotheses listed above will not be proved or disproved on account of unexpected climatic conditions (El Niño's warming effect, causing the winter snow fall to be six weeks late) and will instead form the basis for future research.

1.5. Report structure

This report will commence with an overview of species monitoring, occupancy and data collection methodologies relating to this study and a brief introduction to the study area and species. This will be followed by an explanation of the data collection methods and an analysis of existing data. I will then discuss and review what these results can tell us and suggest how the work in the field can develop further in the future.

2. BACKGROUND

2.1. The need for monitoring threatened species

Monitoring is based on the recurrent observation and recording of fluctuations in the status and population trends of a target species over a specific area (spatial and temporal trends) to assess change in state over time. Variables can include species richness and population size. Its main purpose is to gather data enabling the outcomes of management actions to be assessed and management decisions to be guided. It is important to gain an in-depth understanding of the populations of threatened species and the risks that they face. Accurate distribution maps can delineate suitable areas where species can be protected and reintroduced (Engler *et al.*, 2004). It is also important to establish a baseline for future analyses to assess population trends. This is the basis upon which progress towards the Convention on Biological Diversity ('CBD')'s target of achieving a significant reduction in the rate of biodiversity loss by 2010 can be measured. It has recommended compiling a set of indicators, including one dealing with status changes of threatened species (Baillie *et al.*, 2004).

Monitoring highlights the need for conservation action and species recovery (Campbell *et al.*, 2002) the protection, management, creation and restoration of habitats and management actions to be effective (Kull *et al.*, 2008). Adaptive management requires synergy between management and monitoring and is one of the most important aspects of effective conservation action. Monitoring programme objectives are usually scientific or management based, using insights into a system's behaviour and dynamics or collect information to make informed management decisions (Yoccoz *et al.*, 2001). An additional goal of monitoring is to collate a wide range of knowledge to improve the ability to predict and deal with new situations (Karanth *et al.*, 2002a).

Monitoring is important in order to manage species for conservation and shape conservation decisions. This requires species-specific knowledge of its biology, ecology, range, taxonomy, population and habitat status (Baillie *et al.*, 2004). Information regarding biological systems is also important for biodiversity, for the maintenance of genetic diversity and to identify threats to species and systems and the best way to mitigate against these, hopefully before they have a significant impact on the system. It is also essential to monitor factors such as habitat destruction, fragmentation and degradation and to avoid the degradation of ecosystem services provided by the natural environment. Monitoring is also vital if categorisations such as the IUCN Red-List's 'Data-Deficient' are to be avoided. Species said to be Data-Deficient can be in limited danger, at risk of extinction right through to extinct and, as a result, research and monitoring are essential to protect these species from possible extinction and from being

forgotten by conservation and management. Information derived from monitoring can help to guide conservation planning and priority setting (Baillie *et al.*, 2004). This can include determining what habitats are the most important to conserve in a given system based on the habitat requirements and threatened status of the species residing there.

2.2. Methods used for monitoring (focusing on field and analytical methods for rare and elusive species)

Both direct (invasive) and indirect (non-invasive) methods can be used for monitoring populations. Non-invasive methods are particularly important for rare, elusive, cryptic species and as with carnivores, can have large home-ranges, making it hard to monitor them (Wilson and Delahay, 2001). Analytical methods for monitoring species can be qualitative (mainly based on textual analysis of, for example, questionnaires) or quantitative (using statistical or model based techniques) (Milner-Gulland and Rowcliffe, 2007). The latter include analyses ranging from chi-squared and t-tests to occupancy, abundance and density estimation (using direct observation and/or field signs) analysed in specifically designed statistical packages. PRESENCE uses detection history data and is used for occupancy estimation. It can be set to investigate systems from single-species, single-season to community level and multi-season analyses. It can also take into account survey and site specific variables affecting detection and occupancy probabilities such as lure versus no lure and different habitat types. Another package, DISTANCE, utilises distance sampling data from observations for abundance estimation and other analytical techniques, such as photographic rates for density estimation and twinned camera traps (Fig. 2) either side of a trail, for species with individual markings for abundance and density estimation. Direct field methods include capture and radio-collaring of individuals as well as mark-capture-recapture studies. Indirect methods include track counts (along transects) track plates/scent stations, scat counts and accumulation rates, hair snaring, DNA analysis of hair and scats, counts of structures (e.g. dens) and auditory indices. Indirect or non-invasive observation methods include field and direct observation, aerial observation, spotlight counts, thermal imagery and remote surveillance, i.e. photography and video surveillance. Also included are questionnaires, interviews and sighting reports, road mortality and depredation reports (Gese, 2001).

2.3. Questions over optimal method choice

There are major cost and benefit issues to consider when designing and undertaking any monitoring study. Costs, including labour and equipment, can be measured quantitatively with, for example, raw hit rates, performance metrics and cost per amount of data (hits). Benefits which can be direct and/or indirect (e.g. opportunity costs) can be measured in

monetary terms or qualitatively e.g. cultural or enjoyment value (Milner-Gulland and Rowcliffe, 2007). Whatever the outcome of this analysis, funding of conservation programmes is always a major consideration. The shortfall in current expenditure on conservation is most pronounced in less developed countries but this is where benefit:cost ratio of conservation is much greater (Balmford *et al.*, 2003).

2.4. Occupancy studies

An area is occupied by a target species where that species is present in a closed system (stable population). Each 'station' (point or transect) is representative of the whole area of interest. It is studied multiple times to create a binary history (0/1) (presence/detection = 1; non-detection = 0) and is independent from other stations to avoid counting individuals at more than one station. Following each survey, the station is 'reset' by the researcher. This information is used to determine detection probabilities (p) and occupancy probabilities (psi or ψ) when calculating the site occupancy/proportion of occupied sites (the parameter of interest) (MacKenzie *et al.*, 2002). It can also be combined with environmental covariates to increase model fit and better explain the data. Detection probabilities are based on the likelihood of detecting the target species in a single sampling occasion ('SO') (at time t), given the species is present.

Occupancy studies rely on a set of assumptions: i) occupancy stays constant during the survey season, i.e. sites are 'closed'; ii) occupancy probability is either constant across sites or can be modelled with covariates; iii) detection probability is constant between sites and surveys or can be modelled with covariates; iv) detection and detection histories per site are independent; v) the surveyor's ability to detect animals is consistent across space and time as are the methods used for detection; vi) animals are not falsely detected when absent and when present, they may or may not be detected. Imperfect detection of species and false absences can cause inferences drawn from models to be highly misleading (MacKenzie *et al.*, 2002; MacKenzie and Royle, 2005; MacKenzie *et al.*, 2006). False absences are minimised by carrying out multiple surveys of sampling units over a short time-span (MacKenzie and Royle, 2005). Linkie *et al.* (2007) limited the camera trap sampling period to 3 months per study to minimise the risk of occupancy changes during the study.

MacKenzie and Royle (2005) also indicated that, when sampling rare species, surveying more sites less intensively would be more efficient, whereas the converse was true for more common species. Three different sampling designs can be used to collect occupancy data: 1) in the standard design, all sites (s) are surveyed K times (sampling occasions); 2) in the double sampling design, repeat surveys are only carried out at a selection of sites; and 3) in the removal sampling design, surveying ceases once the species has been detected or after K

surveys have been completed. However, this may be less resistant to assumption violations than the standard design although it is often the most efficient (MacKenzie and Royle, 2005). Variation can occur in both space and detectability. Site and sampling covariates have been investigated by Linkie *et al.* (2006 and 2007), Luiselli (2006) and O'Connell *et al.* (2006) whilst investigating the influence of environmental covariates, e.g. habitat, on the occupancy of tigers, sun bears, gaboon vipers/nose-horned vipers and various species/methods respectively. Covariate models use maximum likelihood to determine which parameters are of most interest to explain occurrence (and detection) (Royle *et al.*, 2005). Sampling covariates can vary both spatially and temporally. Models can investigate various hypotheses being studied (MacKenzie *et al.*, 2003).

Stanley and Royle (2005) looked into how occupancy (and, in some cases, abundance) could be estimated using 'indirect detection indices' (IDIs) such as snow track transects, tracking plates and tubes, scent stations (with lure as an attractant) and hair snare surveys, which are useful when studying rare and/or elusive species. These were based on the ability to 'reset' each station once the investigator has recorded any detections made by the station, by erasing, removing or marking all evidence of the target species having visited the station to avoid re-counting previously recorded data. In the case of snow tracks, this resetting could either be fresh snowfall or marking any tracks with, for example, a spot of spray paint. From the data collected, a binary detection history would be formulated per sampling station and sampling interval. This data could then be analysed to determine detection and occupancy probabilities for each species.

2.5. Field methodologies

Techniques can be categorised as direct, invasive (including capture and radio-collaring individuals) or indirect, non-invasive (including camera traps and snow-track transects). It is also possible to survey prey species (as determined through scat analysis or other means including direct observation) as a surrogate for, or indicator of, species presence.

Direct and indirect methods can be used to estimate abundance of, for example, terrestrial carnivores, the latter of which involve using observation or animal signs as discussed by Wilson and Delahay (2001). Some of these are covered below.

2.5.1. Camera Trapping

Camera trapping has been used to capture some of the world's most elusive mammals. The basic camera trap comprises a pressure-sensitive plate, plus a passive detector or an active infra-red emitter-detector (which detects pressure, movement and body heat or a break in the

beam between the emitter and detector respectively) connected to an automatic, remote camera (as illustrated in Figs. 3a and b). The camera must be strategically positioned (which could itself introduce bias when using only trails and not random placement) and as with the detector, at a height suitable to capture, preferably, the whole body of the target species or group (Wilson and Delahay, 2001). Traps must be spaced taking into account home-range size of the target species (Sanderson, 2004). Once the detector registers a species, it sends a signal to the camera, triggering it and a photo of the area is taken (Swann *et al.*, 2004), providing a permanent photographic record (Gese, 2001) which can be of use to scientists, decision-makers and the general public (Swann *et al.*, 2004). The flash can upset some animals. Studies of animals with distinct pelage patterns can use twinned cameras per station to take photos of both flanks of the animal for identification (Sanderson and Trolle, 2005) and hence abundance/density and even home-range size estimation. For shorter delays between detection and firing, Kelly *et al.* (2008) suggested that, to get high quality photos for identification, cameras should be placed 2 to 3m from where the animal is expected to pass by, during their study of puma density. Trolle and Kéry (2005) used capture-recapture with camera traps to estimate ocelot density in the Brazilian Pantanal and found that the trapping rate varied significantly between roads and trails, factors that should be included as covariates in density analyses. There has been debate over whether photographic rates should be used for animal density analyses (Carbone *et al.*, 2001 and 2002; Jennelle *et al.*, 2002). Rowcliffe *et al.* (2008) reported a new method for determining animal density using trapping rate as well as average species group size, day range, camera sensor angle and distance to the detected animal. Camera traps were used by Linkie *et al.* (2006) to investigate tiger occupancy in Indonesia and showed that occupancy rates were higher, further from roads. Linkie *et al.* (2007) investigated habitat use of sun bears in Sumatra using camera traps and environmental covariates for occupancy estimation and showed that both primary and degraded forests were of importance to them, with significant conservation implications.

Camera trapping has been used in a variety of ecosystems and extensively in tropical forests as illustrated on the TrailMaster website (TrailMaster, 2008). Its varied applications include surveying species' presence in both ecological and conservation situations, deer control and game hunting and abundance estimation. Minimal camera trapping work has been carried out in temperate rainforest systems where studies of mammals are uncommon. A study resembling that in the project area was undertaken in Olympic National Park, U.S.A., using live trapping, camera trapping and sign surveys (scats, tracks, feeding signs and dens) conducted along transects between camera stations (Lomolino and Perault, 2001). Other camera trap population surveys have been conducted on the Olympic Peninsula, north-west U.S.A. (National Park Service, no date).

2.5.2. Sign Surveys

Sign surveys can be used to detect carnivores including felids, canids and mustelids and can use scats and hair samples, tracks, dens or burrows and scratches (Gese, 2001). Occasionally, as in Long *et al.* (2007a and b) scats are located by detection dog teams. These indirect detection indices are very useful when surveying animals such as carnivores who are naturally rare, elusive, found at low densities and difficult to capture repeatedly (Erb, 2005).

Acosta and Simonetti (1999) produced a guide to the tracks of mammals from Chilean temperate forests, using captive animals. Scats, scent stations and opportunistic snow tracks were also used when studying Darwin's fox and detected guiña and puma as well (Jaksić *et al.*, 1990). Scats were used in a study of Darwin's fox from Chiloé Island (Elgueta *et al.*, 2007) and to research guiña, puma, culpeo, chilla and quique (Guerrero *et al.*, 2006).

2.5.3. Animal Tracks and Snow Track Transects

Animal tracks and snow track transects are also examples of indirect detection indices (Stanley and Royle, 2005) and can be used for occupancy studies and to index relative abundance. Tracks can be registered/left in dry river beds, mud and sand/dirt/snow covered roads and trails. The initial registration, clarity /ease of identification and lifetime of each track depends on the substrate, its quality and environmental conditions. In areas that experience snowfall, carnivores, for example, can be surveyed by walking established transects within one to two days following snowfall and counting animal tracks (Gese, 2001). Tracks can be misidentified and the power to detect change can be low with snow tracks, but the latter can be improved with increased survey effort or increased transect length. Factors such as observer experience, depth of snow (as no snow prevents sampling), consistency and condition of snow are all important for reliable data collection. As Wilson and Delahay (2001) highlighted, counts of snow tracks should be carried out along transects soon after snowfall to reduce the impact of freeze-thaw and wind, all three of which can distort tracks, reducing and eventually obliterating the possibility of identifying them and affecting index values. Transects are often divided up into segments or survey units and indices are based on the daily total number of tracks, presence or non-detection of tracks per segment or the distance between each set of tracks. As well as scent-stations, Erb (2005) employed winter snow track indices (of relative abundance) to estimate population trends of marten. For the purposes of this study, the number of tracks of each species per transect was divided by the number of nights since last snowfall, which could affect detection probability in occupancy studies. Stephens *et al.* (2006) also used snow tracks to track deer from their last known location to their new location, 24 hours later, to determine distance travelled for population density estimation using the FMP

(Formozov-Malyshev-Pereleshin) formula. Linnell *et al.* (2007) evaluated three different scenarios for snow track data collection and found that deliberately placed transects, at a higher density and a longer duration since snowfall, increased the detection probability of Eurasian lynx (given presence) and also the index's power to detect change. Becker *et al.* (2004) described how population size could be estimated from animal tracks left in snow using probability sampling of tracks intercepting aerial transects and the length and orientation of each track.

2.5.4. Track Plates and Scent Stations

Track plates and scent stations consist of a square or circle *c.*1m across, filled with a tracking substrate (e.g. fine, sifted soil or a semi-solid mixture acting like mud) sealed with film (Fig. 4) and, in the case of scent stations, lure at the centre. These may be checked as frequently as daily (Acosta-Jamett and Simonetti, 2004) and are an example of an indirect detection index (Stanley and Royle, 2005). Acosta-Jamett and Simonetti (2004) used scent stations to survey habitat use for guiña and culpeo. They used 32 stations across continuous, fragmented and plantation forest, over three to five nights, checked daily, giving 620 scent station nights. Jaksić *et al.* (1990) placed 151 scent stations in subsequent seasons of the same year, along a transect, between 30 and 100m apart when researching Darwin's fox in central southern Chile.

2.5.5. Radio-Telemetry

Radio-telemetry is an invasive technique that involves capturing the species in a trap, anaesthesis and radio-collaring and direct or remote tracking of the animal concerned. It has been used to study habitat use, home-range and daily distance travelled (combined with snow tracks for density estimation - Stephens *et al.* (2006)). Habitat use and home-range have been studied for both guiña (Dunstone *et al.*, 2002; Sanderson *et al.*, 2002) and culpeo fox (Salvatori *et al.*, 1999).

2.5.6. Hair Snares

Hair snares use wire barbs or tape and can be baited with lure to attract animals before collecting a sample of the animal's hair (Stanley and Royle, 2005) for detailed analysis to determine or confirm the species. Long *et al.* (2007b) for example, used 10x10cm carpet pads attached to a tree, with nails/barbs sticking through and lure in the centre.) They are tailored to the target species or group, highlighted by O'Connell *et al.* (2006) who had difficulties detecting certain species, as did Long *et al.* (2007b). Scats and hair snare samples usually undergo laboratory DNA analysis (Long *et al.*, 2007b).

2.6. Cost-efficiency of camera trapping

Long *et al.* (2007b) compared the non-invasive techniques of scat detection dogs, hair snares and camera traps by looking at their effectiveness and cost in a forested, mountainous region (paralleling this study). Detection dogs were substantially more effective at detecting the three target forest carnivores (only needing one visit with no days latency-to-detection), with remote cameras second and lastly hair snares, with the price reducing in the same order. O'Connell *et al.* (2006) also compared remote cameras, hair traps and 'cubby boxes' in a detection array with all three methods baited with lure and running concurrently at each sampling site. They found that cameras were the most effective at detecting species, with cubby boxes second and hair traps least. The effectiveness and cost-efficiency of camera trapping and sign surveys was investigated by Marsh (2008) who found the latter to be more cost-effective.

2.7. The region and habitat

The study area is located within the temperate latitudes of central-southern Chile, South America, 650km south of Santiago, in the IX region, around 39°S, 71°W (Fig. 8b). It includes the southern Andean chain and a lake district. The area is characterised by valleys with very steep, densely forested slopes making access particularly difficult. Altitudes range from *c.*300m on the valley floor to the hill tops (*c.*1,000m) and over 3,000m the highest volcanoes.

The study area is also found within the Chilean Winter Rainfall-Valdivian Forests hotspot (Conservation International, no date) and the Valdivian temperate forests WWF Ecoregion (WWF/Smith, no date) and is characterised by temperate Andean rainforests. The ecoregion covers 248,100km^2 (similar to Oregon, U.S.A.) in a narrow north-south band between 35° and 48° south. The Atacama Desert lies to the north, the Pacific Ocean to the west, the grasslands east of the Andes and to the south, Patagonian plains and sub-Antarctic forests. The Ecoregion is a bio-geographical island and a remnant of Gondwanaland, with a unique collection of species having closer taxonomic similarity to Australasian than to neighbouring Neotropical regions of South America, exemplified by the marsupial, monito del monte *Dromiciops gliroides*. This is a phylogenetic and bio-geographical relict (Himes *et al.*, 2008). The ecoregion is classified as Endangered-Critically Endangered by WWF due to agriculture, deforestation (including large scale burning) (WWF/Smith, no date) and an increasing human population.

The forests feature Coihue (*Nothofagus dombeyi*) and Ulmo (*Eucryphia cordifolia*) trees on the lower slopes (the Coihue-Ulmo subassociation of the Andean Valdivian forest) at 100-850m (Meserve *et al.*, 1982). Around 1,200m, Coihue-Lenga forests with the highest species richness prevail (Elgueta *et al.*, 2006) and above this Lenga-Araucaria predominate (Clavijo Grob *et al.*, 2005). Some of the flora are unique to this region including Araucaria (Monkey Puzzle) trees

(*Araucaria araucana*) found over 1,000m and classified as National Monuments in their own right (Conservation International, no date). These prehistoric trees are unique to the IX Region of Chile. The Valdivian forests usually have a thick under-storey which includes native bamboo (*Chusquea montana*).

The tree-line is at *c.*1,500m, with open grassland above. The snow-line frequently descends to 500-1,000m in winter. This region is well populated below *c.*400m where much of the primary vegetation has been stripped away, removing the habitat for various threatened species. Small fragments of forest remain within the agricultural matrix and this research partly focuses on these. The attached photos in the Appendix depict the differing habitats in which field work has been conducted, camera trap set-ups as well as the gradients of the terrain involved.

The minimum annual rainfall here is 2,000mm (Clavijo Grob *et al.*, 2005) (peaking in winter – June/ July) with average annual temperatures of 2 to 24°C (Lonely Planet Publications, 2008). On higher ground, winter temperatures can drop to -10°C. Pacific westerlies bring rain which is caught by the Chilean slopes of the Andes creating a strong rain-shadow effect.

The study area lies within the Lago Villarrica sub-catchment of the Tolten river system, in the north of the lakes region of southern Chile. Features include lakes Villarrica and Caburga, volcanoes Villarrica, Quetrupillan and Lanin, numerous steep, forested hills and the town of Pucón. It is home to fauna some of which is unique and specific to this region. This report focuses on effective data capture methods and species presence concerning some of these species.

2.8. Mammal species found in the survey area

There are a range of wild, feral or domestic, native or exotic species found in the study area. Carnivores include two felid species (puma and guiña (Fig. 5b)) two fox species (culpeo (Fig. 5a) and chilla), skunk and a mustelid whilst other species including a cervid, suids and bovids are also residents of the forests. The other species encountered during this research include wild fauna (e.g. rabbit and hare) and a range of domestic and feral animals (e.g. dog, horse, cow and goat).

2.8.1. Guiña

Guiña, Chilean Cat or Kodkod (*Oncifelis/Leopardus guigna*) is classified by IUCN as Vulnerable and is listed by CITES (no date) under Appendix II. The genus is likely to be transferred to *Leopardus* from *Oncifelis* (IUCN Red List, 2007). Guiña are smaller than a domestic cat (65cm including tail) and yellowish but can be very dark (melanic) in the south with distinct markings (spots and stripes on the body and bands round the tail) (Campos Cereceda, 1996). They occur

in both Argentina and Chile. *O. g. guigna* are found in the project area and elsewhere, from sea level to *c.*1,900m (Muñoz-Pedreros and Yañez Valenzuela, 2000). Studies on guiña have focused on their home-range (average of 2.69km^2 from Dunstone *et al.* (2002) and 2.12km^2 from Sanderson *et al.* (2002)) and diet as well as a metapopulation analysis of guiña survival (Acosta-Jamett *et al.*, 2003). These studies have taken place in two coastal areas (Sanderson *et al.*, 2002; Acosta-Jamett and Simonetti, 2004) and areas towards Patagonia (Dunstone *et al.*, 2002). Broader studies including species such as *P. culpaeus*, *P. griseus*, *P. concolor* and *L. guigna* have covered the topic of habitat use of threatened carnivores (Correa and Roa, 2005; Acosta-Jamett and Simonetti, 2004; Guerrero *et al.*, 2006). Acosta-Jamett and Simonetti (2004) found that guiña prefer habitats far from roads with dense bush, near large, native forest fragments. However, published studies have not encompassed their distribution, status and ecology in the less fragmented landscapes of the Andean Valdivian forests and have in, certain cases, had limited sample sizes. For example, the radio-collaring studies done by Sanderson *et al.* (2002) sampled seven guiña, two males of whom were shot and killed during the study after raiding hen coups; and Dunstone *et al.* (2002) caught 21 guiña and collected radio-telemetry data for six animals.

In addition to the Darwin Project, one other ongoing study is known from the Argentinean side (Lanin NP) relatively near the project area. This is also investigating guiña and similarly is using scent stations and camera trapping. They use four baited cameras per randomly selected 10km^2 sampling unit and also biennial track counts in scent stations (soil circles, 1m diameter with attractant (lure) in the centre) the latter of which have been successful for detecting guiña in Valdivian forests and culpeo foxes here and on the Patagonian steppe (Monteverde, 2006). Their diet comprises rodents, birds, reptiles, plant material and small mammals (Dunstone *et al.*, 2002).

2.8.2. Culpeo

Culpeo or Andean Fox (*Pseudalopex culpaeus*) is classified by IUCN as Least Concern. Its genus is likely to be transferred to *Lycalopex* (IUCN Red List, 2007). It is also listed by CITES (no date) under Appendix II. It is Chile's biggest fox *c.*48cm tall, 1-1.8m long, weighing 7-10kg and grey-coloured with a whitish chin and long tail. It occurs from Colombia to Tierra del Fuego, either side of the Andes, at an altitude of 0-4,500m (Campos Cereceda, 1996; Muñoz-Pedreros and Yañez Valenzuela, 2000). Studies of the spatial organisation of culpeo foxes in the semi-arid region of north-central Chile determined average home-ranges of 3.5km^2 from a sample size of 13 foxes (Salvatori *et al.*, 1999). A study conducted by Correa and Roa (2005) determined their diet was similar to that of guiña. Acosta-Jamett and Simonetti (2004) found that culpeo, a generalist canid, favours open habitats, near roads but avoids large forested areas. It has been

found in pine plantations, for which it has a preference (Guerrero *et al.*, 2006) indicating that forest fragmentation should not adversely affect it. Its forest preference was (from greatest to least): pine plantations, fragmented and lastly continuous forest. Culpeo prefer habitats on slopes and are comfortable where prey is less abundant (Jiménez *et al.* (1996) cited by Acosta-Jamett and Simonetti (2004)).

2.8.3. Puma

Puma (*Puma concolor*) is classified as Near Threatened on the IUCN Red List (2007) and is listed by CITES (no date) under Appendix II. It is the biggest felid in Chile, up to 2.7m long (including tail) weighing between 25-100kg and ranges from blonde/greyish to reddish brown (Campos Cereceda, 1996). It is widely distributed throughout the Americas. Until recently, pumas in the Central Valley of Chile were considered to be *P. c. araucanus*, with *P. c. puma* in the Atacama Desert and Altiplano to the north and *P. c. patagonica* in the southernmost Andes and Patagonia (Franklin *et al.*, 1999). Genetic studies have since shown that they are all *P. c. puma* (Culver *et al.*, 2000). Chilean studies, mainly conducted in Torres del Paine, Patagonia, have focused on their diet, including that by Franklin *et al.* (1999) (Villalobos, 2006). One study in the Altiplano assessed the state of the population and its impact on livestock (Villalobos, 2006) and analysed habitat use with camera traps and lures (Villalobos *et al.*, 2007). However, it appears that no studies have been published to date from the project area.

2.8.4. Chilla

Chilla, Argentine or South American grey fox (*Pseudalopex griseus* – will be transferred to the genus *Lycalopex* (IUCN Red List, 2007)) is listed by IUCN as Least Concern and is listed by CITES (no date) on Appendix II. It is smaller than culpeo, greyish coloured with a whitish chest, brown and white legs and a light brown/black tail. It is found in southern Peru, Argentina and Chile between 0-3,000m (Campos Cereceda, 1996; Muñoz-Pedreros and Yañez Valenzuela, 2000). This species has mainly been studied using scent stations (IUCN Red List, 2007). Using scat collections, it was concluded that they have no habitat preference (Guerrero *et al.*, 2006).

2.8.5. Chingue

Chingue or Molina's Hog-nosed Skunk (*Conepatus chinga*) is listed on the IUCN Red List (2007) as Lower Risk/Least Concern: it is not listed on CITES (CITES, no date). They are black with a white stripe down each side and black/white showy tail, 54-72cm long and are found in the project area, at 0-2,500m (Campos Cereceda, 1996; Muñoz-Pedreros and Yañez Valenzuela, 2000). It appears that this species has been little studied in Chile. It has been reported in Brazil and Argentina (Cáceres, 2004).

2.8.6. Quique

Quique or Lesser Grisson (*Galictis cuja*) is listed by IUCN (IUCN Red List, 2007) as Lower Risk/Least Concern and is not listed on CITES (CITES, no date). It is brown-black with a white stripe each side of its head-neck and found in South America including the project area at 0-3,800m (Campos Cereceda, 1996; Muñoz-Pedreros and Yañez Valenzuela, 2000). It is a mustelid, similar in size and shape to stoats in the UK. Guerrero *et al.* (2006) were unable to assess its habitat use from scat collection, due to insufficient sample size.

2.8.7. Pudu

Pudu (*Pudu puda* – the Chilean/Southern Pudu) is classified by IUCN (IUCN Red List, 2007) as Vulnerable and is listed by CITES (no date) under Appendix I. It is the world's smallest deer species (Jiménez, 2005) *c.*40cm tall, 75-90cm long, with a brown coat, found in the project area at 0-1,700m (Campos Cereceda, 1996). This species' biology and distribution has been little studied, but distribution modelling indicates that it is much better protected in Argentina (65.9%) than in Chile (16.2%) (Meier *et al.*, 2007).

2.8.8. Jabalí

Jabalí or Wild boar (*Sus scrofa*) is listed on the IUCN Red List (2007) as Lower Risk/Least Concern and is not listed on CITES (CITES, no date). It is 90cm tall, 1.8m long (excluding tail) and up to 300kg (Muñoz-Pedreros and Yañez Valenzuela, 2000). It is an exotic species from Eurasia (IUCN Red List, 2007) and has invaded central Chile (Tognelli *et al.*, 2007). Elgueta *et al.* (2006) noted that livestock, jabalí and feral dogs alter the habitat of and prey on native species.

3. METHODS

3.1. Primary Field Data Collection

3.1.1. Camera Trapping

Camera Trapping as a data collection method has only been used in this region over the past 2.5 years. The initial methodology was derived from past studies and literature on the subject including the protocol set out by Sanderson (2004). The configuration of each trap has been broadly consistent over this period with the cameras being positioned to maximise hits in logistically favourable locations (Laker, 2008c – personal communication). However, the spatial layout (on a landscape level) of the traps has evolved over this period with three successive designs having been used as a development from the preceding one. These will be referred to as Design 1, 2 and 3 throughout this report. All three designs had the aim of studying medium to large mammals in the forests with winter surveys focusing on peripheral areas to protected land (Gálvez, 2008c; Gálvez *et al.*, 2007a and b – personal communications). To the author's knowledge, these were designed by Jerry Laker and Nicolás Gálvez as part of the Darwin Project.

Historically, **Design 1** (the initial fact-finding method from winter 2006) set out to determine what elements of the landscape had the most wildlife activity (Laker, 2008c). Traps were deployed in a variety of habitat types, slopes and altitudes (from 370m to 1272m). The spatial distribution of traps in Design 1 is not known. High and medium TrailMaster sensitivity settings were trialled and analysed by Nicolás Gálvez.

Design 2 (summer and winter 2007) aimed to investigate wildlife activity in areas adjoining the parks and reserves. It progressed with medium sensitivity (following analyses of the early data) and focused on continuous forest and large fragment sites (not small fragments). Several traps were placed in each and trapping was carried out separately in both summer and winter seasons from 400-800m. The number of sites was increased and the design allowed for comparisons between summer and winter habitat use.

Design 3 (summer and winter 2008 - ongoing) surveying progressed to more numerous, carefully chosen sites, examining wildlife activity relative to forest landscape structure (different forest sizes). Sites for camera trap placement in small forest fragments (SF) were selected based on a cover unsupervised classification (5 categories) (which identifies land cover types based on reflectance values) using a mosaic Aster image and cover type was verified with a second mosaic image all imported into a GIS using the GIS programme ArcView v3·2 (ESRI Inc., Redlands, CA, U.S.A.) (Petitpas, 2008a – personal communication). Buffer zones of

2,000 metres from each SF perimeter (Petitpas, 2008b – personal communication) were used to give independent sites (except in one case where a wide, fast flowing river separated sites hence rendering them independent). Small fragments were classified and selected by their size (10-20ha) shape index, isolation and cover quality, accessibility and lastly selected where the owner's permission was granted. Almost all SFs that meet the requirements are being studied.

Design 3 concentrated upon occupancy estimation, using 27 cameras with 9 in each of 3 forest sizes and each at least 2km apart for spatial independence (Gálvez, 2008c – personal communication). Large forest fragments (LF) were selected based on size criteria (>100ha) and had to be surrounded by a farmland matrix whereas continuous forest (CF) areas adjoined one of the forest reserves or national parks. These were of limited availability and accessibility in the study area. They had to have south-facing slopes (more humid and densely forested usually with native vegetation) with an altitude of 400-800m and the sites chosen were the only that fitted these criteria (Gálvez, 2008d – personal communication). (North facing slopes commonly lack vegetation or have forest regrowth, catch the sun so are preferred for animal grazing and are considerably drier and hence prone to fires (Gálvez, 2008c – personal communication).)

The mosaic image was also used in the GIS to get an equal distribution of camera traps between three forest patch sizes and to spread these across the Lago Villarrica Sub-Catchment study area. Here, 9 traps were placed in each of CF, LF and SF sites (within the agricultural matrix) with 3 traps in each of 3 sites for CF and LF and one trap in each separate SF site. The traps were positioned with 1.8-2km spacing (as in Linkie *et al.* (2006) and Maffei and Noss (2008)) as this was regarded sufficient to confer independence between sites (Laker, 2008c) between each station/site, based on the home-range studies carried out for *Leopardus guigna* (Guiña/Kodkod) of 2.1 to 2.7km^2 in other areas of Chile and with the aim of collecting data for occupancy studies for *L. guigna*, as well as presence data for other species found in the forests.

Sanderson (2004) who laid out a Camera trapping protocol, remarked that camera spacing should be at least 1.5-2.0km to keep sample areas independent. The locations chosen for the Darwin Project were usually on slopes with shallow to fairly steep gradients (characteristic of the forests of this region). The camera sites were randomly distributed in each area, being divided into three areas each 2km apart. However, they were tailored to maximise photographic capture and hence detection probabilities of any species present, by being within 20m of a fresh water source on relatively level or gently sloping ground, generally near or crossing natural passages through the forest (Laker, 2008c; Gálvez, 2008c – personal communications). The traps were also baited with WildCat lure (not used in Designs 1 and 2). Locations for each of the camera traps were taken in UTM (Universal Transverse Mercator units) recorded with a

hand-held GARMIN eTrex Legend® (Garmin Ltd., Olathe, KS, U.S.A.) GPS receiver/unit and altitudes were taken with a Suunto Escape203 (Suunto Oy (Suunto Ltd.), Vantaa, Finland).

Two types of weather-proof traps, operational between -40°C to 55°C are used. The active trap (TrailMaster TM 1550) has a thread-like infra-red beam between an emitter and a sensor up to 45m apart and can detect a specific species by setting the beam at its chest height. The passive trap (TrailMaster TM 550) comprises a movement/heat sensitive infra-red sensor with a wedge-shaped area of sensitivity 4° high, 150° wide and 20m deep which is non-species specific, however performs better for all species at a lower height (Swann *et al.*, 2004)(TrailMaster®, Goodson and Associates, Inc., Lenexa, KS, U.S.A.). Once the beam is broken or movement/a heat gradient is detected, it registers an event and sends a signal via an external cable to a remote camera which photographs the area where the beam was broken, within 1-3 seconds. The cameras used are either Canon Sure Shot A1 waterproof cameras (Canon Inc., Tokyo, Japan) or Olympus Stylus 80 cameras (Olympus Corp., Tokyo, Japan) – designed for use with the sensors and included in the Camera Kit from TrailMaster.

Active traps are more appropriate for narrow detection zones e.g. burrow entrance, whereas passive detectors are better for wider detection zones such as waterholes (Swann *et al.*, 2004). Traps were typically positioned across a known or predicted path used by animals, identified by disturbed ground. The infra-red emitter-detector was spaced 2-5m apart with the beam *c.*10-20cm from the ground. The camera was placed strategically to capture an image of the animal without obstruction, typically 1m off the ground and within 0.5-1.5m from the beam. All components of the setup were usually attached to trees, mostly using Velcro straps.

With the latest design (3), lure (Hawbaker's WildCat Lure No. 2 (S. Stanley Hawbaker and Sons, Fort Loudon, PA, U.S.A.)) was also used. Jim Sanderson had determined from his previous work carried out in Chiloé that it was effective for attracting both carnivores and ungulates (cows, jabalí and pudu) and especially for capturing guiñas (Laker, 2008c; Gálvez, 2008c – personal communications). A small amount of this is added into a well in some plaster of paris held in a bottle cap and attached below the emitter or sensor/detector (not the camera) in the field of view of the camera *c.*10cm from the ground. Cameras are loaded with 28 exposure, 35mm, 200 ISO colour print film. They are mostly set to a 2-minute delay (to reduce the likelihood of a single animal triggering many sequential frames and potentially the whole roll of film at once) and record the date and time of frames. To avoid damage to the cables from rodents, the cables are sheathed in rubber tape and barbed with short lengths of wire. To protect from heavy downpours and falling leaves and twigs, the lure is sheltered with a small circle of sheet metal and the camera with two short wooden planks to act as a 'roof'. The Camera Trap is also left with a laminated note including: a short description of the equipment,

that it is for scientific purposes, the contact details of the team/person responsible for it and its maintenance and a request to leave it alone. The cameras were generally in place for *c.*3 months and 'serviced' every *c.*3 weeks on dry days. During this, each piece of the setup was cleaned and checked to make sure that it was functioning correctly, including carefully cleaning all of the seals with propyl-alcohol (for electronics), any film that had either finished or was generally more than half way through (20 frames taken) was replaced and any batteries changed where necessary and any faulty or significantly damaged cables (e.g. as a result of rodents chewing them) were replaced. The films were processed at a local photo laboratory. Where an identifiable photo had been taken, a detection was recorded for the species in question. (This methodology is similar to that used in other studies including Long *et al.* (2007b).)

3.1.2. Other Methods

As in Lomolino and Perault (2001) sign surveys (primarily tracks and scats when feasible) were carried out along transects between camera trap stations. One long, opportunistic snow track transect was walked to, from and between camera traps whenever these were serviced and there was fresh snow (winter 2007) and any intersecting tracks and the species concerned were recorded. These were carried out on dry days due to the conditions needed to check camera traps, which also ensured the reliability and integrity of the animal tracks (i.e. they were not 'reset' by fresh snow the same day before they could be recorded). (Scat counts were also recorded for a range of species concurrently with running snow track transects but not covered in this report.) Locations for each of the track or scat positions, were recorded with a hand-held GARMIN eTrex Legend® (Garmin Ltd., Olathe, KS, U.S.A.) GPS receiver/unit.

Additionally, track plates were used in a small number of locations (Fig. 4). They were placed in front of the lure in a camera trap, opposite the camera, so that any animal that crossed the plate would be recorded by the camera. This also allowed the species' track print to be confirmed using the photographic record.

3.2. Statistical Analyses and Model Design

Once the data had been collected and the films processed, it was collated into a database. From this database, general analyses were carried out, including hit rates, the proportion of sites where the target species was detected during a survey (e.g. 3-month sampling period) and total survey effort mainly for camera trapping but this was also reported for opportunistic transects carried out in winter 2007. These performance metrics (also used by Long *et al.* (2007b)) gave indications as to method efficiency and cost-effectiveness.

The cost of camera trap equipment and related items/labour and transportation for the summer 2008 survey was estimated based on figures provided by members of the research team (Laker, 2008a; Laker, 2008b; Gálvez, 2008b – personal communications). From these, total costs for each site/sampling period (cost per visit) respectively were calculated and the total survey cost determined. The cost per hit was also determined for summer 2008 and from this, winter 2007 trapping and transects cost per hit were calculated and cost-effectiveness determined. From the sampling cost, hypothetical future snow track transect costs were estimated under four scenarios where the level of survey effort was varied. Training costs were excluded as these were factored into initial survey costs, where a handful of experienced workers would be accompanied by (and train) volunteers. The daily cost of camera trap sampling (trap visits) was added to those for new equipment, experienced personnel and trained volunteers, covering 4 transects per survey area or site per day and travelling to and from each site per day. (Similar costs were estimated for camera trapping and scat detection dogs by Long *et al.* (2007b)).

The data from three survey seasons (summer/winter 2007 and summer 2008) were converted into a binary detection history (0 = non-detection,1 = detection or 'hit' - presence) and then input into the occupancy software program PRESENCE v.2.2 (Hines, J.E. USGS-PWRC, Laurel, MD, U.S.A. (2006)) to estimate detection (p) and occupancy (ψ) probabilities (MacKenzie *et al.,* 2002). The detection history used by PRESENCE could be '11010' indicating detection of the target species on the first, second and fourth sampling occasions but not the third and fifth. Single-season, single-species analyses were used throughout.

The data was analysed using first 5- and later, 10-day sampling occasions (where the data was compressed into sequential blocks of 0s or 1s) with a range of scenarios from Designs 2 and 3 and a sample of species from the area (carnivores and ungulates, native and exotic) being reduced to culpeo fox and guiña for 10-day analyses. 10-day sampling occasions were used with the aim of providing an increased proportion of 1's to 0's enabling the limited number of hits to be better analysed by PRESENCE, given the generally elusive nature of the species found in the forests and the relatively low detectability of the species. Sampling occasions were formed from sequential blocks of temporally continuous data. Where there were insufficient days on the end of the sampling period to form a whole sampling occasion, days (almost all 0s) were lost. Where a break in sampling occurred e.g. where a film finished or battery died some days before they were replaced, the same rule applied and sampling occasions were restarted in the next block, i.e. temporally separate detection data was not combined into any one sampling occasion.

10-day sampling occasions, from the summer 2008 data, were then used to carry out covariate analyses. This was to determine which landscape factors (forest unit/fragment size and altitude) alone or in combination, best explained the probability of species presence (ψ). Firstly, site-specific covariates were assessed to determine which parameters best explained occupancy probabilities. One of these analyses was then extended for the species with the most detections and hence the most likely to be more accurately analysed (culpeo fox). This was done using site (psi(x) and/or p(x)) and sampling (p(x)) covariates (potentially affecting occupancy (psi) and detection probabilities (p)) to determine which model best explained occupancy and detection probabilities (MacKenzie *et al.*, 2002). The covariates (site-specific only) used in the models were different combinations of habitat type (i.e. fragment size (psi(frag) and p(frag)) (a categorical variable) and altitude (psi(alt)) (a continuous variable) and these were compared with the constants (psi(.) and p(.)) (MacKenzie *et al.*, 2002). These were carried out to determine which model (and hence parameters) best explained occupancy and detection probabilities. PRESENCE ranked each model according to its AIC (Akaike Information Criterion) value – a goodness-of-fit measure for an estimated statistical model, where the lower the value, the better the fit (Burnham and Anderson (1998) cited by Linkie *et al.* (2006)). The highest ranked model(s) (all within 2.0 delta-AIC (Δ AIC) of the top model) were used to determine which covariates best described occupancy and/or detection probabilities. (Features of this analysis were reflected in Linkie *et al.* (2006).) Different data treatments were also analysed, i.e. how variable selection of binary data when condensing sampling occasions could affect the outputs from PRESENCE. (The data was derived from the winter 2007 data for guiña. The binary data from camera units 10B and 14B (5-day sampling occasions) was grouped for analysis in three different ways and this altered the number of 10-day sampling occasions registering a hit. Finally, low detection and occupancy probabilities were modelled to calculate the survey effort that would be required based on the standard model. Based on the cost per sampling period and the number of sites from the summer 2008 survey, the resulting cost of each modelled survey effort was calculated. This was based on MacKenzie *et al.* (2006)'s standard design.

4. RESULTS

4.1. Methodological analyses

Methods used for camera trapping between summer 2007 and 2008 and survey effort, productivity and performance metrics are assessed below.

4.1.1. Camera Trapping

Camera trap summary data & Performance indicators/metrics	Design 2		Design 3
	Summer '07	Winter '07	Summer '08
Total known active trapping days (sampling effort)	479	1030	2067
Total functioning camera stations	11	20	27
Total frames taken (excluding tests)	185	436	352
Total hits (known detections – mammals + birds)	125	110	230
Total mammal hits	55	97	200
Total misfires (no species detected)	60	326	122
Active traps: misfires of total frames (%)	37.5%	77.0%	34.6%
Passive traps: misfires of total frames (%)	28.6%	40.7%	35.3%
Total field days (that trap visits made)	8	16	51
Total full field days (when 4 traps visited per day)	6.25	15.25	29
Percentage: hits of total frames (productivity)	67.6%	25.2%	65.3%
Percentage: misfires of total frames (performance)	32.4%	74.8%	34.7%

Fig. 6: Performance metric data for camera traps over 3 trapping seasons (surveys).

During the summer 2008 survey, 27 camera trap stations yielded a total of 2067 trapping days (see Fig. 6 above) (minimum - based on functionality of camera traps, i.e. as some traps may have been operational until later but as the date was unknown, the last known date was recorded). Using 5- and then 10-day sampling occasion length (for occupancy analyses) this reduced to 405 and 195 sampling occasions respectively. The number of single, 5- and 10-day sampling occasions for Design 2 during winter 2007 were 1030, 195 and 90 and for summer 2007: 479, 91 and 40 respectively.

Of the 11 summer 2007 camera stations, 9 were active and 2 passive with 30/80 and 30/105 misfired frames of the total frames taken. In winter 2007 of the 20 active stations, 16 were active and 4 passive with 315/409 and 11/27 misfires respectively and in summer 2008 of 27 stations, 25 were active and 2 passive with 116/335 and 6/17 misfires respectively. Percentages are given in Fig. 6 above.

Remote camera surveys were conducted across all sites (11 (summer 2007), 21 (winter 2007) and 27 (summer 2008) some located very close to, or in the same place, as a previous survey period). Of these, camera or sensor failure occurred for the duration of the survey in one site (winter 2007 and not included in occupancy calculations) and 4/14, 13/40 and 15/88 sampling periods (32/142 total). Camera failure occurred over 3/11, 10/20 and 13/27 (including 8 of 9 SF sites) camera sites (26/58 total) respectively. At other sites, where the battery or film ran out at least once before being checked, 38 (7.5%), 219 (17.3%) and 295 (23.2%) (552 (13.5%) total) days were lost between total sampling time and effective sampling time.

Failures included cases in which the entire roll of film was wrongly shot in one day. These were apparently as a result of snowfall (blocking the sensor) battery failure, chewed cables and camera misalignment. Detection histories for occupancy analyses were based only on days when the trap was known to be fully operational (similar to Long *et al.* (2007b)).

The discrepancy between the number of days and 5- and 10-day SO blocks was a result of how SOs were selected (see methods).

Guiña and culpeo were photographed at one common camera station in each survey. Guiña were detected at 12B in each survey and culpeo were detected at 8B and 7N in the first two. Both species were also detected in the same forest site during each survey and at more than one camera in each (see Figs. 5a a/b and 7). Over the course of the three surveys, guiña were detected at Barda, Huellemolle and Currarehue LF forest sites; Namuncai and Kawellucó CF sites and 91, 93 and 95 SF sites. Culpeo were detected at Barda, Huellemolle and Currarehue LF sites; Namuncai and Kawellucó CF sites and 92, 93, 94 and 98 SF sites. (For the camera trap site map, see Fig. 8).

The minimum latency-to-first detection (the time from camera deployment until the first detection (as used by Long *et al.* (2007b))) for summer 2007, winter 2007 and summer 2008 for culpeo was 9, 4 and 51 days and for guiña was 9, 4 and 22 days respectively. The mean values (across cameras with hits only) increased to 29, 48 and 79 (culpeo) and 19, 37 and 65 (guiña).

During summer 08, culpeo were detected at 9/27 sites (LF and SF only) and guiña at 9/27 sites (all three sizes); whereas in summer 2007 this was 3/11 and 2/11 (both in LF and CF) and winter 2007 was 5/20 (LF and CF) and 3/20 (only LF) (operational cameras only) respectively. (Long *et al.*, 2007b)

The end date of surveys, where the battery or other component failed part-way through the sampling period, was taken as the last known operational date (i.e. when the last detection was made).

The species detected throughout the camera trapping process are listed in Fig. 9.

4.1.2. Track Plates

At a selection of sites both huelleros (track plates) and camera traps were deployed concurrently. At trap 97F, three quique prints were picked up by the huellero but not by the camera trap (summer 2008) as found on 29 February 2008 camera check. On two occasions a huellero track was confirmed with a photo taken by the camera trap above it. On 15 February 2008 a guiña track in a huellero at trap 91F was confirmed with a photo from 13 February 2008 (only a guiña was captured during this sampling period). Also, during 26 February 2008 check of 94F, a track was found and confirmed as culpeo fox with a photo as only culpeo foxes were captured during this sampling period.

4.1.3. Snow Track Transects

During the 18H camera check on 28 August 2007, a pudu track was found. This must have formed between 23 August 2007 (most recent snow) and 28 August 2007 however, as the camera film had finished on 13 August 2007 with snow (23 frames on same day), there was no photographic record of this. During 9 July 2007 check of 5N, a puma track was detected 20m from the camera but the animal had not passed through the camera trap which was still active.

Three different (forest) locations were used for snow track transects: Huellemolle (5 SOs of 4,000m); Kawellucó (3 SOs of 2,600m) and Namuncai (4 SOs of 9,064m (including 2,510m grassland)) with the assumption that there was snow cover for the entire transect. Opportunistic snow track transects and successful track identification showed the presence of guiña, pudu, jabalí, lagomorph (hare/rabbit), puma, dog, chingue and fox (culpeo/chilla) in winter 2007. Each snow transect walked in winter 2007 had at least 1 hit (Gálvez, 2008e – personal communication).

In total, 12 transects were walked covering 64,056m with 52 mammal hits, including 19 guiña and 4 fox (culpeo or chilla). To record a detection, a minimum average distance of 520m up to a maximum of 9,064m was required with an average across sites and days of 1,232m. An average distance of 3,640m was required to detect a guiña (minimum 1,000m). Guiña were detected at all sites with a total of 19 detected, whereas fox were only detected at 2/3 with 4 detections in total (Fig. 10).

4.2. Methodological-based costs

4.2.1. Estimated cost of summer 2008 sampling

An estimated £11,612.20 has been spent on the collection of camera trap data over the 27 camera stations/sites during summer 2008 (230 hits total), £9,783.73 of this on site costs (camera equipment and maintenance) across the 27 camera stations and £1,828.48 on sampling (period) costs (labour and transportation) across 116 camera visits.

4.2.2. Cost-effectiveness of systematic camera trapping compared with initial, opportunistic snow track transects

Summer 2008 camera trapping gave the cost per hit as £50.49 (total survey cost/total hits). Based on site and sampling costs from summer 2008, the winter 2007 sampling would have cost £8,208.73 with a cost per hit of £74.62. Early opportunistic winter 2007 transect data indicated one would have to walk an average of 1,232m per hit. Assuming *c.*6km could be walked in one day, this would give *c.*5 hits per day costing £11.34 per hit (daily sampling cost/5), contrasted with £74.62 per hit from the systematic camera trap data for the same season.

4.2.3. Estimated cost of potential future snow track transects

The costs of camera trap sampling (trap visits) at £56.70/day was added to those for equipment to give the total survey cost. (Equipment costs are low compared to those for camera trapping overall and the equipment is used for all sites surveyed so only incidental costs such as spray paint, acetates and pens have to be replenished occasionally. These costs were estimated at £536 (based on: callipers *c.*£6; spray paint, acetates and pens *c.*£10; 2 pairs of snow shoes (for the field team) *c.*£120 for 2 pairs; small digital camera *c.*£150 and a GPS *c.*£250).

Scenario	Repeats	Transects	Total days	Sampling cost	Total Survey cost
1	3	32	24	1,361	1,897
2	3	64	48	2,721	3,257
3	5	32	40	2,268	2,804
4	5	64	80	4,535	5,071

Fig. 11: Estimated survey costs (GBP/£) (sampling and equipment) for four scenarios of snow track transect survey effort.

4.3. Occupancy analyses

4.3.1. Five-day sampling occasion species analyses and 5- *vs.* 10-day analyses

Data from a range of camera trapped mammal species was analysed using 5-day sampling occasions (SOs). Different scenarios from Designs 2 and 3 showed varying occupancy and detection probabilities. Individual scenarios (and detection histories) were comparable between species, but as each detection history within each species was different, the scenarios were not directly comparable within each species (Fig. 12). Given that the overall sample size was small, data from both years and seasons, with different treatments (lure/no-lure) and habitat types (altitude and forest size) were pooled. They were also analysed in PRESENCE along with the separated data whose results varied substantially depending upon the fluctuating levels of detections in different years and seasons. (This was the basis of the different scenarios.) What these preliminary analyses highlighted were the large standard errors surrounding each estimate and the low occupancy and detection probabilities. Consequently, two species were selected, guiña (a small endemic threatened wild cat) and culpeo fox (a wide ranging species with the highest detection probabilities with relatively low standard errors for each scenario across all the wild species). Five-day SOs were converted to 10-day SOs (closer to the 2-week SOs used by Linkie *et al.* (2007)) in an attempt to increase the proportion of 1s (hits) to 0s (non-detections) to give the best estimate of occupancy and detection probabilities for each (Fig. 13). From the single season scenarios, both the estimates themselves and the standard errors for the *p* and psi estimates increased when going from 5- to 10-day SOs. The data from summer 2008 gave the highest detection probabilities for each species and were therefore used for analysing the site-specific covariates.

4.3.2. Ten-day site covariate analyses using occupancy probability parameters

The covariate analysis of guiña with fragment size and altitude combinations as occupancy covariates against the constant model, showed no clear 'highest ranked model'. Five of 8 models were 2.0 or less Δ AIC from the highest ranked model which was the constant (psi(.)p(.)). The same models run for culpeo were re-run with just LF and SF data, as the data for CF had zero hits across all 9 cameras and could thus not be analysed properly (Figs. 12a and b).

4.3.3. Ten-day site covariate analysis using occupancy and detection probability parameters

Model	AIC	Δ AIC	AIC wgt	Model Like'	no.Par.	-2Log(Like')
psi(Alt),p(Frag)	79.21	0.00	0.3139	1.0000	4	71.2100
psi(Frag+Alt),p(Frag)	79.72	0.51	0.2433	0.7749	5	69.7200
psi(Frag),p(Frag)	79.78	0.57	0.2361	0.7520	4	71.8000
psi(.),p(frag)	80.06	0.85	0.2052	0.6538	3	74.0600
psi(.),p(.)	91.11	11.90	0.0008	0.0026	2	87.1065
psi(Frag),p(.)	93.09	13.88	0.0003	0.0010	3	87.0906
psi(Alt),p(.)	93.10	13.89	0.0003	0.0010	3	87.1039
psi(Frag+Alt),p(.)	95.09	15.88	0.0001	0.0004	4	87.0897

Fig. 15: Results of covariate analysis for culpeo fox using LF and SF, 10-day SOs, summer 2008 Design 3 data. (AIC (Akaike Information Criterion); Δ AIC (variation in AIC values between each model and the highest-ranked, lowest AIC model); AIC wgt (model weight); Model Like' (model likelihood); no.Par. (number of parameters); -2Log(Like') (-2Log(Likelihood)).)

Four of the eight models explaining culpeo occupancy and detection received strong support (Fig. 15) and together these had an Akaike weight of 0.9985. All of these models included fragment size as an important parameter for detection probability. Model 1 had the highest Akaike weight and suggested that culpeo occupancy was positively affected by altitude (Linkie *et al.*, 2006).

The covariate analysis of culpeo occupancy indicates that culpeo foxes are positively influenced by the site conditions (highest ranked model(s)). The 10-day SO, summer 2008 data analysed for large and small fragment at various altitudes, indicates that altitude and fragment size affect occupancy but as the constant psi (psi(.)) is <1 AIC unit away, there is no substantial effect of either on occupancy. There is a very small positive effect of altitude on occupancy (highest ranked model – psi(alt),p(frag)). There is a clear effect of fragment size on detection probability and as the effect is negative, it is more likely that the species would be detected in small fragments than in large ones.

Detection probability (p) for LF and SF was calculated as 0.0561 and 0.5043 respectively (almost a 10-fold difference). The psi β-coefficient for altitude (model 1) was 0.0138 with 0.0137 standard error and a 0.0411 to -0.0136 95% confidence interval. As this includes 0, there is no clear effect of altitude on occupancy. The psi β-coefficient for fragment size (model 3) could not be calculated fully as it appeared there was too little data. The *p* β-coefficient for fragment size was -2.8408 with 0.6850 standard error and -1.4708 to -4.2109 95% confidence interval. As

this does not include 0, it shows that fragment size affects detection probability. Culpeo were not detected in CF which does not prove absence (MacKenzie *et al.*, 2002) but indicates that if they are there, they are difficult to detect.

As the psi for altitude has a large standard error, similar in size to the actual β-coefficient estimate for psi, it indicates that more data is required to reduce this error and to gain a value closer to the 'true' value for psi. This would increase the likelihood of determining if there really is no effect of altitude on occupancy or whether there is a positive or negative correlation. There is a 95% probability (σ2), and hence near certainty, that the detection of culpeo is affected by fragment size. As this value is negative, the bigger the fragment size, the more negative the detection co-efficient and hence the less likely the probability of detection.

4.3.4. Data treatment effects illustrated with guiña winter 2007 data

Model	AIC	-2log(like)	*p*	p(S.E.)	*ψ*	Ψ(S.E.)	Naïve Est'	sites	SOs	pm's
Original	30.3061	26.3061	0.0333	0.0189	1.0	0.0	0.15	20	8	2
Revised	35.4298	31.4298	0.1433	0.1215	0.3111	0.2501	0.15	20	8	2
Revised2	35.1812	31.1812	0.1618	0.1332	0.2822	0.2179	0.15	20	8	2

Fig. 16: Comparison of 3 different data selection methods (models) and the resulting *p* and *ψ* values. (All with an equal total sampling occasions and all modelled with constant sites model (2 parameters p(.) and *ψ*(.))). AIC (Akaike Information Criterion); -2log(like) (-2log(likelihood)); *p* (detection probability); p(S.E.) (standard error of p); *ψ* (occupancy probability); *ψ*(S.E.) (standard error of *ψ*); Naive Est' (Naive estimate); SOs (sampling occasions); pm's (parameters).

The above table demonstrates that subtle alterations in how data are selected for grouped sampling occasions (detection history) can substantially alter the resulting estimates of *p* and *ψ* from results that cannot be used for further analyses to ones that can (O'Connell *et al.*, 2006) (p sub 0.1-0.03 to *c.*0.15 to 0.3 (and a corresponding increase in standard errors S.E.)). The original model (with very low *p* and inestimable (PRESENCE defaults to 1 and 0) *ψ* values) was that used throughout for sampling occasion data selection (a consistent, scientific and ethical method). (As the data were temporally discontinuous for Revised and Revised2 (see Fig. 17) the approach was considered unscientific and unethical.)

4.4. Occupancy-based costs

4.4.1. Modelled survey effort and expected cost required to gain precise parameter estimates at low occupancy and detection probabilities

p	*ψ*	var (*ψ*)	*K*	no. visits	*s*	TS	cost tot' *K*	cost tot' *s*	cost tot' TS
0.1	0.1	0.05	14	8	58	812	126.08	21,027.84	21,153.92
0.1	0.2	0.05	15	9	101	1,508	141.84	36,426.31	36,568.15
0.1	0.3	0.05	16	9	130	2,077	141.84	47,042.38	47,184.22
0.1	0.4	0.05	17	10	147	2,507	157.60	53,444.91	53,602.51
0.2	0.1	0.05	7	5	56	391	78.80	20,226.19	20,304.99
0.2	0.2	0.05	7	5	104	725	78.80	37,553.51	37,632.31
0.2	0.3	0.05	8	5	125	996	78.80	45,126.21	45,205.01
0.2	0.4	0.05	8	5	150	1,200	78.80	54,370.51	54,449.31
0.3	0.1	0.05	5	4	50	251	63.04	18,208.54	18,271.58
0.3	0.2	0.05	5	4	92	462	63.04	33,518.20	33,581.24
0.3	0.3	0.05	5	4	127	634	63.04	45,928.98	45,992.02
0.3	0.4	0.05	5	4	153	765	63.04	55,440.88	55,503.92
0.4	0.1	0.05	3	3	61	182	47.28	21,939.25	21,986.53
0.4	0.2	0.05	4	3	84	335	47.28	30,349.86	30,397.14
0.4	0.3	0.05	4	3	114	455	47.28	41,176.47	41,223.75
0.4	0.4	0.05	4	3	136	542	47.28	49,104.20	49,151.48

Fig. 18: Modelled comparison of survey costs (GBP/£) at different values of *p* and *ψ* at a precision of 0.05 standard error. (Cost/survey effort (sites and sampling occasions) using cost per visit and site.) [*p* (detection probability); *ψ* (detection probability or psi); var(*ψ*) variance or error around *ψ* value; *K* (number of sampling occasions); no visits (number of visits needed at 3 weekly checks (as in summer '08); *s* (number of sites); TS (total number of surveys or *K***s*); cost tot' *K* (total cost of all Ks, based on number of visits from summer '08 costing; cost tot' *s* (total cost of all sites based on summer '08 costing) and cost tot' TS (total survey cost).]

To achieve reliable estimates of detection and occupancy probabilities, using a standard error (S.E.) of 0.05 (an acceptable level of precision based on examples given in MacKenzie *et al.* (2006) and as used in Linkie *et al.* (2007)), a range of *K* and *s* would be required at an estimated total cost given in cost tot' TS. Modelled above are the number of surveys to conduct at each site (K) and number of sites (s) that would be needed at a total cost of all surveys (TS) based on costs from summer 2008, Design 3 camera trapping. For example, based on data collected over seven sampling occasions, a predicted 104 site surveys would be required (TS (total surveys) of 725) at a total cost of £37,632 (when *p* and *ψ* are both very low at 0.2) (Fig. 18 above). (Further costs with S.E. = 0.1 are published at the end of this report (Fig. 19).)

5. DISCUSSION

5.1. Discussion of Results

5.1.1. Methods

5.1.1.1. Camera performance and winter misfires

Remote cameras give an immediate positive identification of a species, can be deployed over a long period, prove current use, enable multi-species monitoring and provide a permanent record for scientists and the public alike (Long *et al.*, 2007b).

Performance metrics from camera trapping carried out over the past three survey seasons have highlighted some important issues to be considered for ongoing and future surveys in the study area.

In winter 2007 the overall rate of misfires was 75%, contrasted with *c.*33% for the two summer seasons, potentially due to the abnormally harsh conditions. However, these results were unreliable and inconclusive due to the comparatively low sample size of passive sensors. However, for active traps, close to ground level, it is probable that snow, or fallen leaves or branches, blocked the beam triggering continuous misfires, whereas misfires for passive traps were substantially lower due to their raised height. Passive detectors perform well at lower heights but poorly higher up. Unless the breadth of the beam is reduced from 150°, misfires can occur when an animal is outside the field of view of the camera. They may also have been partly due to direct sunlight, mimicking body heat, to small animals crawling on the sensor, precipitation (both very common in the forests of the study area: movement of vegetation or the sensor itself (Swann *et al.*, 2004). The high rate of misfires in winter indicated that camera trapping may not be the most appropriate or cost-effective method to use in this season.

Swann *et al.* (2004) noted that some traps are more effective at detecting smaller animals than others whereas all detect large animals. Unless the active TrailMaster sensors are very close to the ground, smaller animals can pass under the infra-red beam and hence go undetected, as was the case with quique (picked up by a track plate). When the beam is too low to the ground, pumas could step over it and be missed entirely.

The raw data shows that passive cameras may be better placed in more remote mid-altitude locations (eg. Kawellucó) as otherwise cows can trigger the sensor repeatedly and even cause premature termination of the film, especially if there is no programmed delay of, for example, 2 minutes on the camera.

5.1.1.2. Camera and equipment failures

Swann *et al.* (2004) noted that failure can affect all remote camera systems but the nature of failures differ between systems. Although the TrailMaster's range is -40°C to 55°C, the infra-red sensors operate optimally at <26°C but become less sensitive at higher temperatures.

It is likely that using remote cameras in high altitude National Parks, susceptible to deep snowfall, would be highly unsuccessful as the equipment would be more exposed and most likely get snowed up or frozen and fail. Data shows that trapping in forests in winter has a poorer success rate than in summer for similar reasons (here trees/branches can fall rendering the traps useless). One extreme example of snow's effect was when a camera trap was checked and the sensors were found 1m off the ground and misaligned. Subsequent analysis of the film showed that snow had most probably triggered firings on three separate days, leaving the sensor high up as the snow level receded.

Lure may lose its efficacy more rapidly due to humidity in the forests and frequent, heavy rains in autumn-winter. It seems to have increased the trap rate of species such as guiña and puma (Gálvez, 2008f - personal communications) based on ongoing studies in the field. Rodents have been photographed by camera traps in the lure and cows have been captured sniffing the lure.

However, the cables are increasingly attacked (being chewed and at times severed) by rodents in the forest, occasionally rendering traps inoperative and leading to much increased maintenance times. Swann *et al.* (2004) noted that reinforcing cables with materials such as duct tape can minimise damage to them from chewing. Cabling has had to be barbed due to more frequent rodent attacks following the introduction of lure, increasing project costs (labour and maintenance). There is again, a trade off between more hits and more failures of equipment and more field time protecting equipment. It is expected that the barbing will prove cost and time effective as long as hit rate is increased with lure introduction.

5.1.1.3. Latency-to-detection

The mean latency-to-first detection (the time from camera deployment until the first detection) indicated that cameras need to be in the field for over 80 days in summer and 50 in winter (based on upper mean values for guiña and culpeo) in order to register a detection when a species is definitely present. This may be coincidental or could be explained by varying activity levels for example. Based on the non-existent detections in continuous forest for culpeo and the 80-day latency, the sampling time especially in CF may need to be substantially increased to say 120 days in summer whereas based on the converse habitat preferences (Acosta-Jamett and Simonetti, 2004) of guiña, the latency of 65 days in summer may be best increased to say

120 days in small fragments. This would help to determine whether these species have very low occupancy rates in these habitats or just are less detectable there.

The latency in winter for camera traps compared to the very early signs from track transects (i.e. latency-to-first detection *vs.* finding tracks on each transect walked regardless of length. Culpeo and guiña had a minimum latency of 4 days each and a mean of 48 and 37 days respectively. This indicates that track transects could be a more effective way of sampling in winter.

5.1.1.4. Track plates

The author found huelleros (track plates) covered in leaves in late autumn, rendering them far less effective at this time of year than they may be in spring. They also have to be checked regularly (every few days rather than weeks, in contrast to the camera traps). In summer 2008, camera trap photos confirmed tracks left in huelleros to be those of guiña and culpeo.

Pumas (as shown by one camera trap photo) can step straight over the huelleros (if they are too small) and miss stepping on them completely so they need to be made bigger and less conspicuous. Studies such as Rau *et al.* (1992) gave dimensions of $1m^2$ or 1m diameter (if circular) for track plates or scent stations.

Quique have been picked up by the huelleros but not the camera traps, possibly as they pass under the beam, going undetected.

5.1.1.5. Snow track transects

From preliminary data, it appears that snow track transects would be a valuable method for studying habitat use and occupancy in the study area in winter (given appropriate weather conditions). Based on raw hits alone, the very early signs from the opportunistic data corroborate the covariate information gained from the camera trapping for culpeo as there were very few (grouped) fox hits from the transects carried out in large fragments and continuous forest (none in CF from summer 2008). This showed that they prefer small fragments and open terrain rather than guiña who have the opposite tendencies (in line with Acosta-Jamett and Simonetti (2004)) also corroborated by this data having the most hits of any species.

In these cases, snow tracks would provide a viable alternative, especially when done one or two days (allowing time for tracks to register) after fresh snowfall, as long as there are favourable environmental conditions (i.e. it remains cold, no further snow falls and there are only light breezes).

The very early transect data showed a high number of guiña hits compared to other species. This could be explained by life history traits, such as propensity to follow tracks/paths through the forest or a higher activity level. The low detection of culpeo could be explained by what Acosta-Jamett and Simonetti (2004) found, i.e. that they prefer open habitats to those of forest, especially larger extents of forest. As a result, the data currently only shows that transects appear to be an efficient and cost-effective way of monitoring. Camera traps and transects both have disadvantages, i.e. cost, misfires and component failure *vs.* weather conditions for example (which the author experienced first-hand).

Snow tracks were found in the vicinity (20m away) of one trap (5N) however the animal (puma) had not passed through the trap and thus not been detected by it although it was still active. Therefore, the cameras are not entirely effective at attracting and detecting the species present in the survey area; in this case, however transects were. This may not have been the case if lure had been in use by this stage, as it was only introduced in summer 2008.

Snow track transects are believed (based on opportunistic data) to have a higher hit rate than camera traps as every time a track survey was done, at least one detection was made. This implies that there is a higher detection probability with the latter so cannot be directly compared to traps unless estimated occupancy probabilities are used (which could also vary, to a greater extent if there is less data for traps) with raw hit rates.

5.1.1.6. Method-based and occupancy-based costs

The cost for snow track transects could be reduced by repeat sampling some sites 2 days running and staying overnight at a refuge in the field.

The equipment must be monitored regularly depending on the environment in which it is placed (e.g. every 3 weeks) to ensure that the trap is fully functioning and to change batteries and or film where necessary (Karanth *et al.*, 2004). The initial set-up costs (equipment costing <USD300 to >USD600 (Swann *et al.*, 2004)) can be substantial, but fully functioning cameras can operate 24 hours, 7 days per week. They can also be vulnerable to damage, theft and equipment failure. Delay between detection, triggering the camera and the taking of a photo, as well as time and costs of film development are also disadvantageous. The introduction of digital cameras is now overcoming some of these drawbacks (Gese, 2001). This is in contrast to a method such as snow transects which, depending on the field team size and their experience at identification, can only sample sites soon after fresh snowfall. Where sites are isolated they can take some time to reach, even with snow shoes, reducing the survey effort and increasing the labour costs even without expensive equipment. (Wilson and Delahay, 2001)

The frequency of visits in the study area being 3-weekly could be in part explained by the fact that the Olympus cameras' batteries last approximately 3 weeks, whereas Canon's last 3 months (Laker, 2008b). The TrailMaster batteries are expected to last 8 months in active traps and over one year in passive traps.

5.1.1.7. Method comparisons

Early signs show that camera trapping in winter is much less cost-effective than a method such as snow track transects due to the high number of misfires resulting from snowfall versus the very early signs from snow track transects where a track was detected on every transect walked. The rate of hits from summer to winter, *c.*66% to 25% shows a very marked difference in productivity. (It is likely that many hits in winter are also due to cattle grazing in the forests whilst the pasture is poor and hence further mislead the results for hits.) It appears to be more cost-effective to do snow track transects in winter, especially on higher ground and to focus on camera trapping in summer, as weather conditions permit.

Firm conclusions about method effectiveness cannot be drawn from the data for various methods in order to make direct, meaningful comparisons, but these data illustrate several interesting points.

5.1.2. Occupancy

The data for the culpeo co/multivariate analysis corroborated Acosta-Jamett and Simonetti (2004) who found that culpeo favour areas with less cover preferring plantations/fragments to continuous forest, as was the case in this study.

The use of covariates is usually employed to improve model fit. However this appears to be fairly ineffective with datasets with minimal data or missing covariates where an important factor is not taken into account to improve model fit. In these instances, the constant model (no covariates) can come out as the highest ranked model as there the covariates do not help to explain the data any better. This was the case with the preliminary data obtained from guiña (Fig. 14a).

Big errors around an estimate in covariate analyses can result from limited data or one or more factors/covariates not having been taken into account (or both). Linkie *et al.* (2006) made the assumption that prey biomass varied with habitat type, the former being one such covariate which could play a role. Covariates such as trap type (active or passive) could also affect detection probabilities.

Single season, single year, single species was chosen for the analyses as summer 2008 was more targeted to occupancy due to the camera spacing, as this violates fewer assumptions or to a lesser extent than other scenarios (MacKenzie *et al.*, 2006).

As culpeo's home-range has been found to be *c.*3.5km^2 (Salvatori *et al.*, 1999) this would violate the assumption of independence of camera sites for this analysis. This estimate was from semi-arid Chile and, in the study area, is likely to be less in the forests if there is a higher prey abundance, but their range may still be too great to satisfy the independence assumption.

There is a trade off between the number of reduced SOs (sampling occasions) and the potentially increased proportion of 1s to 0s (hits to non-detections) i.e. starting at daily and working through to say 20 days. I modelled 5 days and for 2 selected species (the most common and one of the least which is also threatened) additionally 10-day blocks (this was increased to 15 days or 2 weeks by Linkie *et al.* (2007)).

A clear disadvantage of condensing hits into blocks is that, for example >1 hit in a 5-day block is considered to be the same as 1 hit as was the case when condensing species such as dog *vs.* guiña. As a result it can be misleading for occupancy analyses (Fig. 13). Days are grouped arbitrarily to reduce any bias. Condensing blocks from say 1- to 5- to 10-days can reduce 0:1 ratio, hence increasing the proportion of 1s and can increase data quality. It is the same for 10-days blocks. Where the number of sampling occasions is not exactly divisible by the sampling period the data was selected for best proportion of hits without combining temporally separate SOs or losing SOs, such that, in some cases, hits were lost (amalgamated). The better case scenario was used without manipulating the data in an unethical way. Hits were also lost where occasions or daily hits were too close or the data immobile. Hits on adjacent sampling occasions which were condensed would have had a greater impact on rarer species with the fewest hits and could potentially have a clearly measurable negative impact on the results. An example of this was seen from the guiña winter 2007 data with 3 models assessed with the data moving from immeasurable data to more respectable estimates of p and ψ (Fig. 16).

Very low detection probabilities were commonly seen when analysing species presence in this study. O'Connell *et al.* (2006) who compared methods including remote cameras and hair snares, questioned p values between 0.05 and 0.15. MacKenzie *et al.* (2002) warned of estimates of psi very close to 1 when p is low (<0.15) especially when the number of SOs is small (<7) as detection/non-detection data is limited making it difficult for models to separate absence from non-detection. This was apparent, frequently in the data, for example, in the case of guiña summer 2008 data (10-day SOs) where psi was estimated at 0.7 (with a large standard error of 0.34) and p at 0.08 with an average of 7.5 SOs (1-14 SO range, one being too low to

analyse). The results would otherwise be misleading and inconsistent with a rare, elusive species such as guiña due to insufficient data. PRESENCE usually gives a psi of 1.0 when detection probabilities are very low (especially where data are very limited). Therefore psi may be overestimated as the programme assumes the species could be prevalent even though it is virtually impossible to detect (Smith, 2008 – personal communication). As PRESENCE cannot separate true absences from pseudo-absences, it is more appropriate to consider occupancy to be usage (MacKenzie and Royle (2005) cited by Linkie *et al.* (2006)).

Long *et al.* (2007b) used a 14-day sampling duration as this was believed to be sufficient for animals to find the attraction-based detection devices on their home-ground. Based on the fact that traps were out for 75 days and very few detections were made of guiña, for example, it indicates that they are found at very low densities (in their preferred habitat (Acosta-Jamett and Simonetti, 2004)) across the whole study area. They noted that it is difficult to calculate the actual surveyed area using these methods.

Methods that search for signs of animals (active, e.g. scat detection dogs and transects) may permit more accurate estimation of parameters such as home-range size and habitat use. They have reduced bias compared to station-based methods, using lure, which record them (passive, e.g. hair snares, scent stations (track plates) and remote cameras). These attraction-based methods can draw animals into hazardous situations (e.g. forest clearings or road edges) and create male/female bias depending on their preference for particular scents (Long *et al.*, 2007b).

Foxes are generalist species and as a result the plantation-dominated landscape in Acosta-Jamett and Simonetti (2004) may partly explain the culpeo habitat preference. The abundance of roads in more fragmented landscapes such as those around the SFs facilitates movement and may help when hunting. Fragmented landscapes appear to not have an adverse impact on them. Their study showed that guiña depend on large forest patches, being native forest specialists and their conservation status would be severely affected by forest fragmentation and a conversion to an agricultural matrix.

When calculating precision equation to get *s* and *K*, the occupancy and detection probabilities had to be rounded to nearest (0.1) and thus the values for *K* from the table used for reference may be far from appropriate as values were typically very small, and hence *K* differs widely, e.g. *p* (detection probability) of 0.15 and psi (occupancy probability) of 0.37 rounds to 0.2 and 0.4 respectively. With psi of 0.4 and *p* of 0.2, *K* is 8 rather than 17 if *p* is 0.1. The closer the probabilities get to 1, the less variation there is in the value of *K*, e.g. *K* of 3 or 4.

Currently there is insufficient data to assess the comparative presence of one species versus another to see if there was any positive or negative effect.

5.2. Recommendations and Future Direction

It is critical to define a survey's objectives, the availability of personnel, climate and topographical constraints and other factors in order to determine the most suitable method(s) to use for a study (Long *et al.*, 2007b). To avoid compromising the integrity of the study one must select sampling units, conduct sufficient repeat surveys and time these appropriately to obtain quality data from which reliable conclusions can be drawn (MacKenzie and Royle, 2005).

5.2.1. Analyses

The ongoing research in the study area could in future be extended as more data is collected, enabling new analyses, potentially more reliable, to be carried out.

Ecological analyses, including investigations of any interactions between the presence of different species could be conducted. Any variation in species richness between higher sites and lower sites in winter could be researched. Latency-to-detection data could provide information about factors such as activity level in different seasons or species-specific characteristics. Relative abundance of the species found in the forests could be assessed. Assessing the occupancy of the national parks, over 1,000m, would provide valuable data to determine whether the parks are providing protection for species such as guiña.

With regard to **Ecological and Occupancy** analyses, further examination of results with more detailed covariate information could be carried out especially with an increased data set and species detections. These covariates could include distance to rugged terrain, roads, distance to forest edge and for smaller (and larger) fragments, distance to the edge of (large fragments or) continuous forest. They could also include proximity to rivers, as many of these are too small to be recorded on the available maps of the area and represent temporal or permanent water sources. This mapping or analysis may only be possible with aerial or hydrographic surveys. Rivers (depending on size, depth and potential for crossing) can also represent barriers for dispersal of species such as guiña although species such as puma may only be restricted by wide, fast flowing rivers. Further habitat type, forest extent and vegetation covariates could be analysed from the data that has already been collected from the study area by members of the Fauna Australis research team (linked to the Darwin Project). Analysis of the continuous variable, altitude at which the species are detected, could be repeated once more information is available. Trap type (active or passive) could also affect detection probabilities and hence should be considered in future covariate analyses. Using season as a covariate could provide

information as to effects on occupancy and detection probabilities. This could be extended to multi-season analyses conducted in PRESENCE to assess the population dynamics over time.

The work carried out by Stanley and Royle (2005) could also be used to evaluate any differences in species visitation to camera traps with and without lure, i.e. covariate information, as seen in their second example with scent stations and coyote visitation. Acosta-Jamett and Simonetti (2004) carried out a logistic regression analysis using covariate information to assess the effects of vegetation types and landscape variables on culpeo abundance. Similar analysis could be applied here.

Occupancy from Design 2 could be recalculated using grouped traps per site where these are not independent due to their spacing.

Once current efforts to trap a wild guiña are successful, the team plan to put a radio collar on the individual which will enable better understanding of home ranges in the study area of the Andean foothills, an area and habitat not covered by previous studies. The collar(s) would also allow for analyses of density/abundance to be made using the daily distance travelled information, key to the Formozov-Malyshev-Pereleshin formula discussed by Stephens *et al.* (2006).

Detection rates in future could be calculated focusing only on cameras fully operational throughout the study as in Long *et al.* (2007b) and a comparison be made between this and total camera survey stations.

In future, effectiveness could be assessed for a range of methods using performance metrics such as raw detection rates at sites where the combined methods are used (paired samples), sampling effort, performance and productivity (Long *et al.*, 2007b). The productivity per site could be assessed by calculating the ratio of total hits to the number of hits per site. Mammal hits could also be compared with detections of tracks on transects. Mean latency-to-first detection could be calculated for fully active cameras only (as seen in Long *et al.* (2007b)) and compared to that for the distance travelled before the first track (detection) on a transect. Comparison of method costs would also be a worthwhile evaluation to extend from this study in designing further surveys, once designs have been decided upon and are in place and as an evaluation once they have been completed to maintain cost-effectiveness and to systematically and adaptively sample. This could be between (e.g. camera traps *vs.* snow track transects) and also within methods (e.g. active *vs.* passive traps).

In future, analysis could be conducted once more data has been collected to compare the rate of return of data per method, the survey costs for each and then to assess the rate of return of

records per unit cost (a cost-benefit analysis). This would also look at the value of the research to science and to conservation of the species and habitats involved.

Further research is needed to look at other techniques to see how camera trapping ranks against others in terms of cost-effectiveness of collecting data, including assessing snow track transects compared with camera trapping in winter once more transect data has been collected.

The camera-based occupancy analysis provides an indication of how well this method performs in providing precise, reliable estimates. For culpeo the data collected backs up the Acosta-Jamett and Simonetti (2004) paper which shows that fragmentation would not adversely affect them, but for guiña there is insufficient data and hence more needs to be collected to draw conclusions and impact for conservation. However, based on the same paper it shows that the integrity of native continuous forests are very important from a species conservation perspective.

By calculating the amount of each habitat type for each forest site, carrying capacity could be estimated (Linkie *et al.*, 2006).

Detection-non-detection data from repeated surveys at various sites enable estimation of detection probabilities (MacKenzie *et al.*, 2002). By taking into account method-specific detection probabilities from different methods, the resulting data from the same site can be amalgamated given that survey periods are set and methods are independent. To attain a specific detection probability, the number of surveys per method or a combination of them can then be calculated (Long *et al.*, 2007b). Long *et al.* (2007b) compared the effectiveness of remote cameras, scat detection dogs and hair snares, comparing p values from each method and determining the probability of detecting a species in a single visit or survey. MacKenzie and Royle (2005) recommended a minimum of 3 for K (sampling occasions) when $p>0.5$ and a greater number when p is lower except when using a removal design. Methods can also be compared by running each simultaneously in a detection array at the same site (Long *et al.*, 2007b). In future different methods could be assessed in this way. This was modelled as part of this present research for low detection and occupancy probabilities and the required survey effort and cost proved to be substantially higher than has thus far been feasible. The data also showed that these levels of effort would be difficult to achieve within most conservation research budgets.

Camera data could, over the longer-term, be used to map the probability of occurrence and hence distribution of captured, target species (especially ones towards whom the design has been focused). A standardised method to calculate species range area was introduced by Nowell

and Jackson (1996). To capture the target species the beam is set at its chest height, hence close to the ground is good for smaller species but larger species could step over the beam and not be detected whereas, higher placement could capture all larger animals but smaller ones could pass under the beam (Swann *et al.*, 2004; Linkie *et al.*, 2006).

Sanderson (2004) noted that camera traps should be sited along travel routes or at favoured spots for animals, identified following at least 1 month's pre-surveying before the actual survey starts.

5.2.2. Camera Trapping

If camera traps were revisited every *c.*10 days (closer to the 2 days to one week mentioned by Long *et al.* (2007b) to replenish lure and check the set-up) this could raise detection probabilities, as could deploying cameras at higher densities, increasing the actual area surveyed. Increasing the number of camera traps per independent site to at least two (Kelly *et al.*, 2008) in an array (an independent sampling unit) (as in Smith (2007) regarding track plates) could increase the capture probability. This could also greatly reduce loss of data if one camera failed as others will be operational within the same area. The overall failure rate and number of days lost between effective sampling and camera deployment time could therefore be reduced. Traps could also be moved around every few weeks within each independent survey area. This could however substantially increase project costs.

In future, sites could be grouped in arrays per sampling unit to hopefully increase detection probabilities and reduce the total number of sites and sampling duration needed. This would violate the independence assumption unless the detection histories are combined per site. In the project area it would be extremely difficult, if not impossible to site the 135-150 independent camera stations needed based on the modelled data (Fig. 18). Camera traps could be placed in the National Parks (particularly in summer when snow is not an issue) as they are currently only positioned on the slopes below and peripheral to the parks. Cameras could also be distributed across a wider area irrespective of habitat in case the various species surveyed are occupying landscapes previously thought inappropriate (Smith, 2008 – personal communication; Smallwood and Schonewald, 1998).

Hawbaker's WildCat Lure #1/#2 (already in use)/a combination of both/a control (no lure) could be trialled across the sites to attempt to determine which lure/combination was best (taking into account habitat and other variables).

The use of adaptive sampling is especially important for rare species (McDonald, 2004). A camera trap study carried out by Maffei and Noss (2008), showed a valuable example of

adaptive sampling, when the spacing of some traps was reduced from 2km in the first study to 1km in the second systematic survey for estimating ocelot density. They recommended standardising the spacing of cameras across an area at least four times the average home-range of the target species when estimating density (Maffei and Noss (2008) cited by Kelly *et al.* (2008)).

In future, as Canon cameras have longer battery life, these should be used in preference to Olympus. Digital cameras should also be trialled due to their significantly increased data recording capabilities and enhancing reliability. A camera system appropriate to the target area (i.e. active or passive for a narrow or wide area) and species in question (i.e. small or large) should be selected.

For successful trapping, the equipment must be mounted on very stable supports and placed where there is no vegetation in the foreground or nearby branches, otherwise false triggers could occur. Specifically clearing vegetation should be avoided as this could affect (positively or negatively) detection probabilities, as could human scent left on the equipment. As cameras are autofocus they should be set 2-5m from the target area (Swann *et al.*, 2004).

MacKenzie and Royle (2005) indicated that the optimal removal design would be the most efficient but less robust than the standard design to assumption violations.

5.2.3. Snow Track Methodology

In order to carry out systematic snow track transects it is vital to collect temporal (time and date) and spatial (geographic position) data for the start, end, detection location and transect route walked. A minimum of three repeat transects should be walked along the same route at a number of locations. Transects and detection locations can be given a track index (e.g. 'logging track' to 'no defined path') to be used as a covariate along with habitat information and altitude. The date of last snowfall should also be recorded as detection probabilities would be expected to increase up to 2-3 nights after snowfall (the optimum interval). Current and disused logging tracks in the survey area could be used for access to and transects themselves.

For a pilot study, the survey area could be divided into a home-range sized grid with similar transect lengths in each square and e.g. 40 squares surveyed with 4 repeats of each. Alternatively, transects per grid square could be divided up into 4 set length portions (acting as repeats for the detection history) rather than carrying out repeat transects on separate days (Rowcliffe, 2008 - personal communication).

Repeat surveys are often conducted on different days ('discrete visits'); however, this is not always necessary. Also, multiple surveys can be conducted on one visit, multiple observers can

carry out independent surveys (same day or different visits) or multiple transects can be surveyed over a larger site in one visit. Which to choose depends on meeting model assumptions based on species biology, sampling logistics and the study objective. The timeframe is a primary consideration: studies over a short time provide a snapshot of the occupancy of the system and hence surveying all sites as rapidly as possible is preferable, to minimise the potential for change and to satisfy the assumption that sites remain closed to changes in occupancy (MacKenzie and Royle, 2005).

Linnell *et al.* (2007) used 264 lines, deliberately placed to maximise detection probability, each with a transect length of 3km from valley floor to a height above the lynx's usual range, with a transect density of 1/38km^2. In extreme cases, where access and habitat availability is scarce, as few as 40 transects may be sufficient to carry out occupancy analyses given high detection probabilities, although a minimum of 80 are preferable (Rowcliffe, 2008 – personal communication). In contrast, the Snow Leopard Conservancy (2006) recommended a maximum transect length of 1.5-2km when surveying snow leopards.

I walked one snow track transect in the study area (July 2008) and made the following observations: snow tracks must be very fresh to be readily identifiable; fresh snow must be 1-3 days old, unless there is no disturbance and little wind to blow snow off the trees and no warming leading to freeze-thaw; tracks can easily be confused between species; in uniform snow, tracks and the direction of travel are very easy to see. Since snow does not discriminate between tracks and paths, whether the transect is run on or off track may make little difference as it all looks the same, unless the path is clearly defined by the surrounding vegetation. Animals can use the same paths as humans and even in patchy snow or mud, tracks can be detected and hence presence inferred. The ability to carry out snow track transects is highly weather dependent and was not possible in the first half of the winter season this year in the study area. According to Dirección Meteorológia de Chile (2008) (Chilean Meteorological Office) in Temuco, July 2008 has seen maximum temperatures up by 0.5°C from 'normal' and minimum up by 2°C. (Temuco is the nearest city to the project area for which data was available and is found in the Central Valley 80km NW of Pucón in the foothills of the cordillera.)

In future, survey effort can be increased by using volunteers and rangers to do simultaneous or sequential (over a very short time frame) landscape level survey transects. This greatly reduces or obliterates the issue of violating the closed population assumption and maintains independence between sites. Transects are more likely to be truly independent if they are completed in a very short time span (Karanth *et al.*, 2002b) and involve multiple surveys which together minimise false absences (MacKenzie and Royle, 2005).

The use of inexperienced volunteers increases sampling bias but gaining the data in a short time frame and using volunteers, with limited training and variable observation skills compared to skilled, trained personnel, is a trade-off that must be considered.

5.2.4. Other Methodological Aspects

The use of scat detection dogs can be considerably more cost-effective and have high detectability compared to remote cameras. It has minimal bias contrasted with transects, but has substantial logistical (and cost) implications. However, it would be advisable to consider their use in future research along with hair snares which continue to be developed for non-invasive, species-specific sampling. These have long deployment potential, low equipment costs, require minimal personnel training and (like remote cameras and track plates) measure current use. (Long *et al.*, 2007b)

As scats persist in the environment, they indicate presence sometime in the recent past, i.e. days, weeks and even months (Long *et al.*, 2007a). Conversely, tracks do not persist for long and are easily erased but like scats they do not detect actual animals, just signs of their past presence. Remote cameras however, detect presence in real-time (Long *et al.*, 2007b).

Acosta-Jamett and Simonetti (2004) recorded data including GPS position and the distance to features such as the nearest road, the centre of the reserve, the nearest native forest patch and the nearest creek or fresh water source.

5.3. Conclusion

Based on the cost of camera trapping calculated from the data for summer 2008 and compared to the very rough estimate for snow track transects, it would appear to be much more cost-effective to carry out transects rather than or as well as camera trapping in winter, especially on higher ground and to focus efforts in summer on camera trapping but with a (substantially) increased survey effort where possible. Tracks can be difficult to identify and require favourable weather conditions to preserve the integrity of the track whereas cameras provide an instant identification of the species.

With deeper snow usually occurring at higher elevations, camera traps are unlikely to be a viable option there in winter.

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APPENDIX

Fig. 1: Photos illustrating various aspects of the Darwin Project. All photos taken by the author unless otherwise stated.

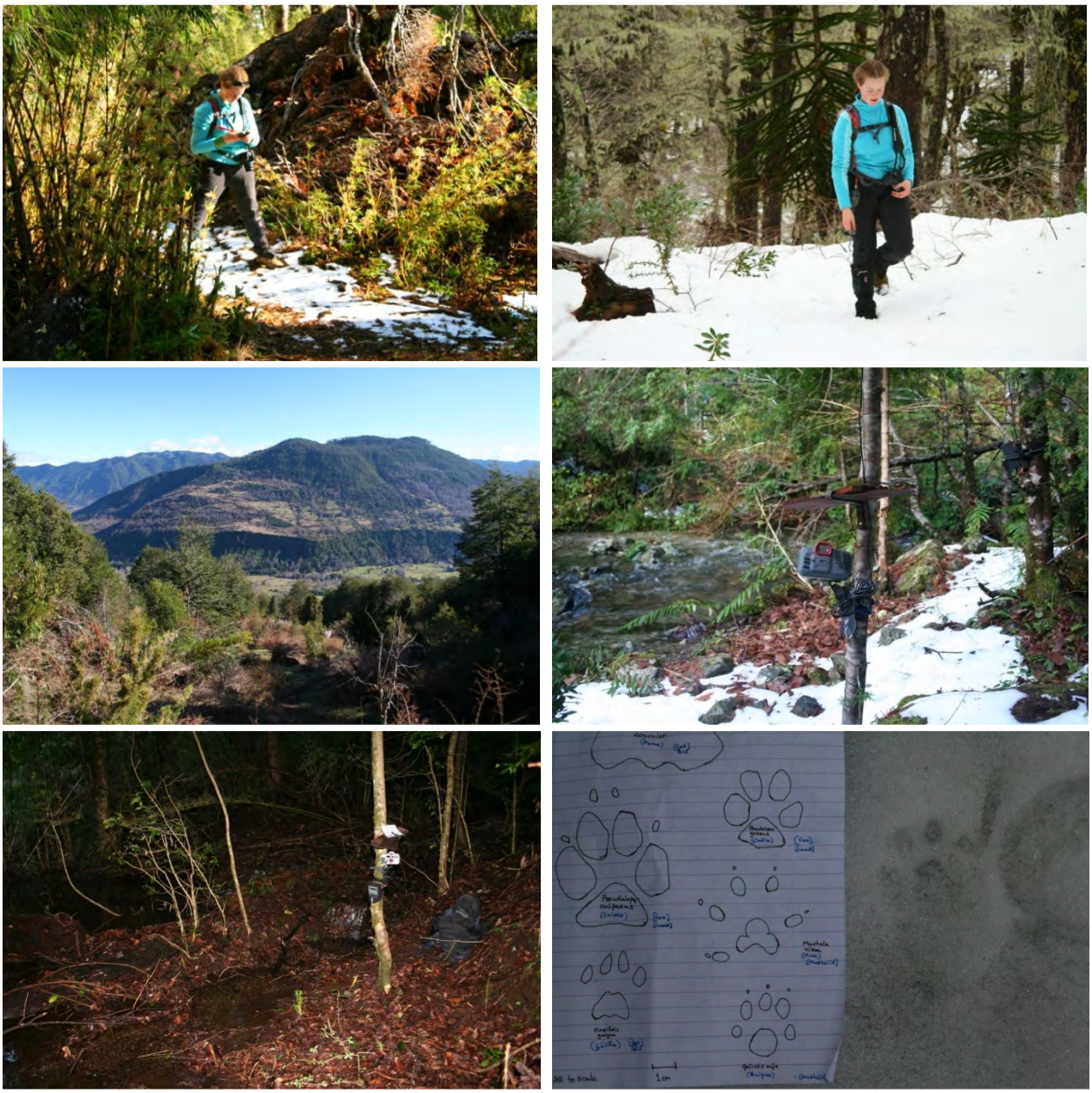

Fig. 1: From top-down: *Left*: Field work – author recording the location of a snow track; La Barda (LF) site taken from El Cañi showing dense, steep south facing slope (left) and facing its fragmented east slope with near-vertical glacial-formed cliff at the base; Passive camera trap by a small forest stream and pools (note the sensor higher off the ground (below the camera) than active sensors). *Right*: The author walking a trial snow track transect (El Cañi); Passive camera trap (camera and sensor from behind) by a mountain stream; A track similar in shape to *L. guigna* but possibly *P. culpaeus or griseus* – an example of identification difficulties with tracks.

Fig. 1 (continued): *Top left* - Typical Habitat >1,000m with *Araucaria* trees – Puesco/Lanin – Chile/Argentina boarder; *top right* – track found at El Cañi, suspected to be *P. culpaeus or P. griseus*; *bottom left* – typical forest under-storey with dense bamboo; *bottom right* – active camera trap showing camera, barbed cables and infra-red sensor firmly secured by being strapped to a tree.

Fig. 1: *Left* - Location of Kawellucó (site on the right, Volcan Villarrica on the left); *Right* – A track plate/scent station *in-situ* and below infra-red emitter and lure (below the emitter). The plate being covered in leaves and damaged and displaced by an animal suspected to be horse or cow (taken late autumn-early winter 2008).

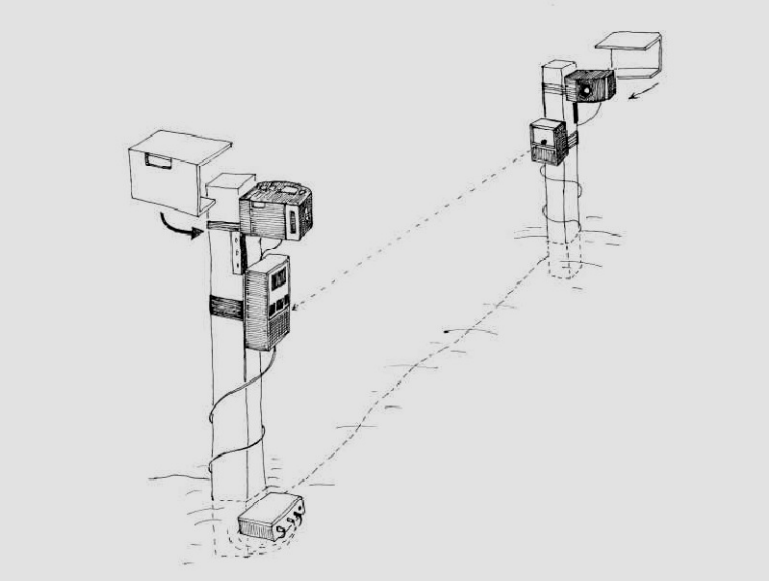

Fig. 2: A twinned camera trap set-up. Note the beam at the chest height of the target species, in this case Tigers (Karanth *et al.*, 2002c – ch.12 figure 12.1 p.141).

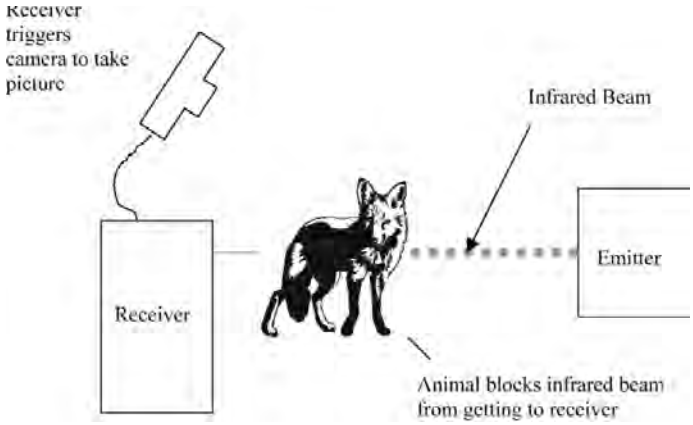

Fig. 3a: Systematic diagram showing the typical configuration of an active infrared camera trap (Swann *et al.*, 2004).

Fig. 3b: An *in-situ* active camera trap.

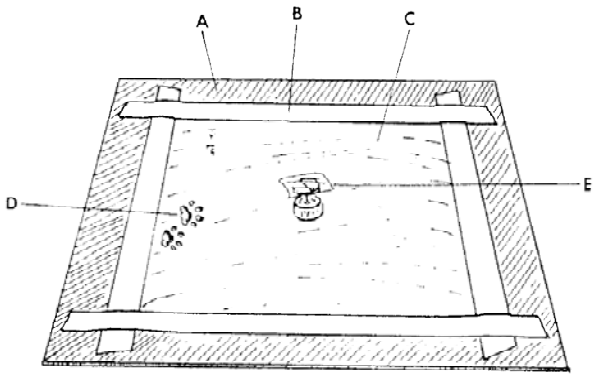

Fig. 4: Scent station design. (Design used by Darwin Project from 'Figure 1' in Rau *et al.* (1992) p.323.) Station consists of a) ply-board covered in a mix of plaster of paris and oil (to prevent setting); edged with b) duck tape; overlaid with c) cling-film (to maintain the moisture levels and integrity of the plate); d) puma tracks; e) a porous block of cheese on a wooden base, held in place with a clip and partly covered a sheet of plastic.

Figs. 5a and b: Photos of culpeo fox - *P. culpaeus* (*left*) and guiña - *L. guigna* (*right*) courtesy of the Darwin Project.

Fig. 7: Location of Guina and Culpeo hits. [Camera unit codes (B = Barda (LF), N = Namuncai (CF), H = Huellemolle (LF), K = Kawellucó (CF), RV = Currarehue (LF), F = fragment (SF)]

Guiña:
Summer 2007: 12B, 6N
Winter 2007: 10B, 12B, 19H
Summer 2008: 50K, 51K, 22K, 3N, 80RV, 12B, 91F, 93F, 95F

Culpeo:
Summer 2007: 8B, 6N, 7N
Winter 2007: 8B, 18H, 19H, 27K, 7N
Summer 2008: 70H, 15H, 71H, 81RV, 7B, 92F, 93F, 94F, 98F

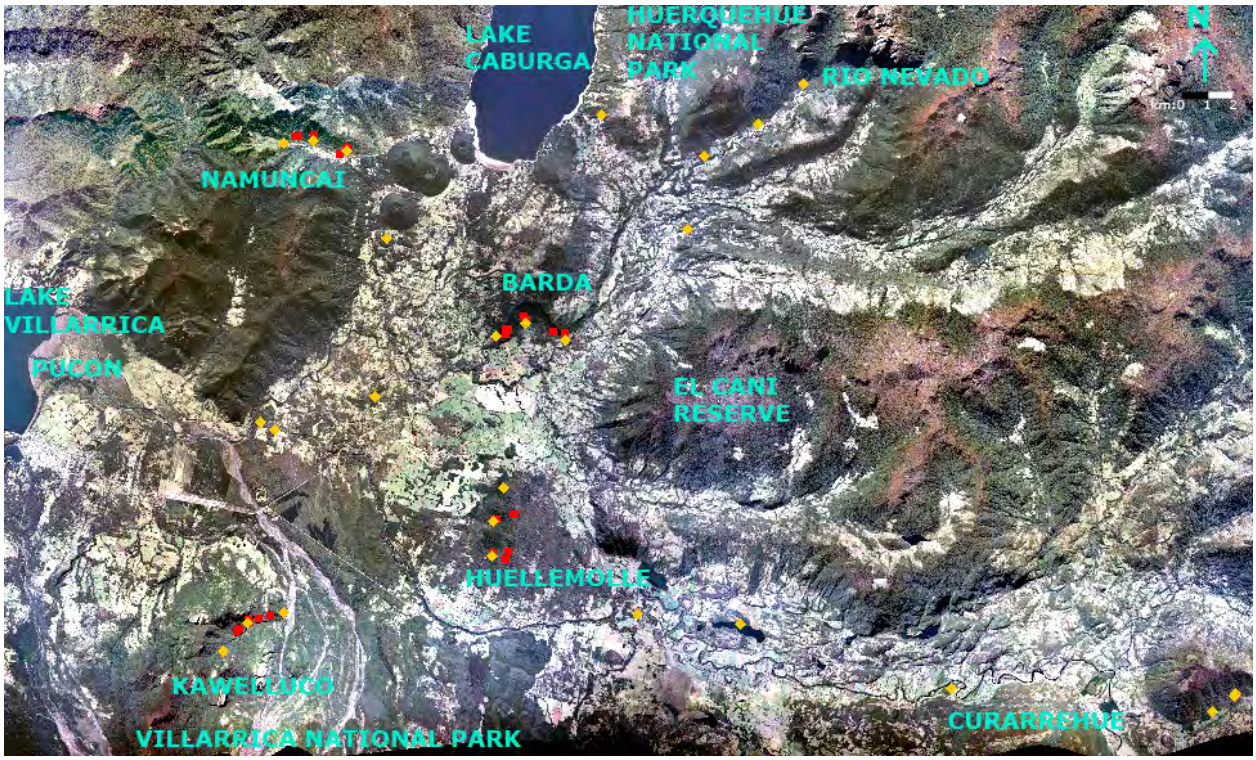

Fig. 8: Map (Mosaic image courtesy of the Darwin Project) of the locations of the Design 2 and Design 3 camera traps, around the study area. (Design 2 traps in red, Design 3 traps in orange and with forest site names (Namuncai, Kawellucó and Rio Nevado (CF sites); La Barda, Huellemolle, and Curarrehue (LF sites) with the SF sites dotted across the agricultural matrix on the valley floors and lower slopes of hills), selected National Parks and Reserves (Huerquehue and Villarrica National Parks and El Cañi Reserve) and key features (Pucón town and Lakes Caburga and Villarrica) marked.)

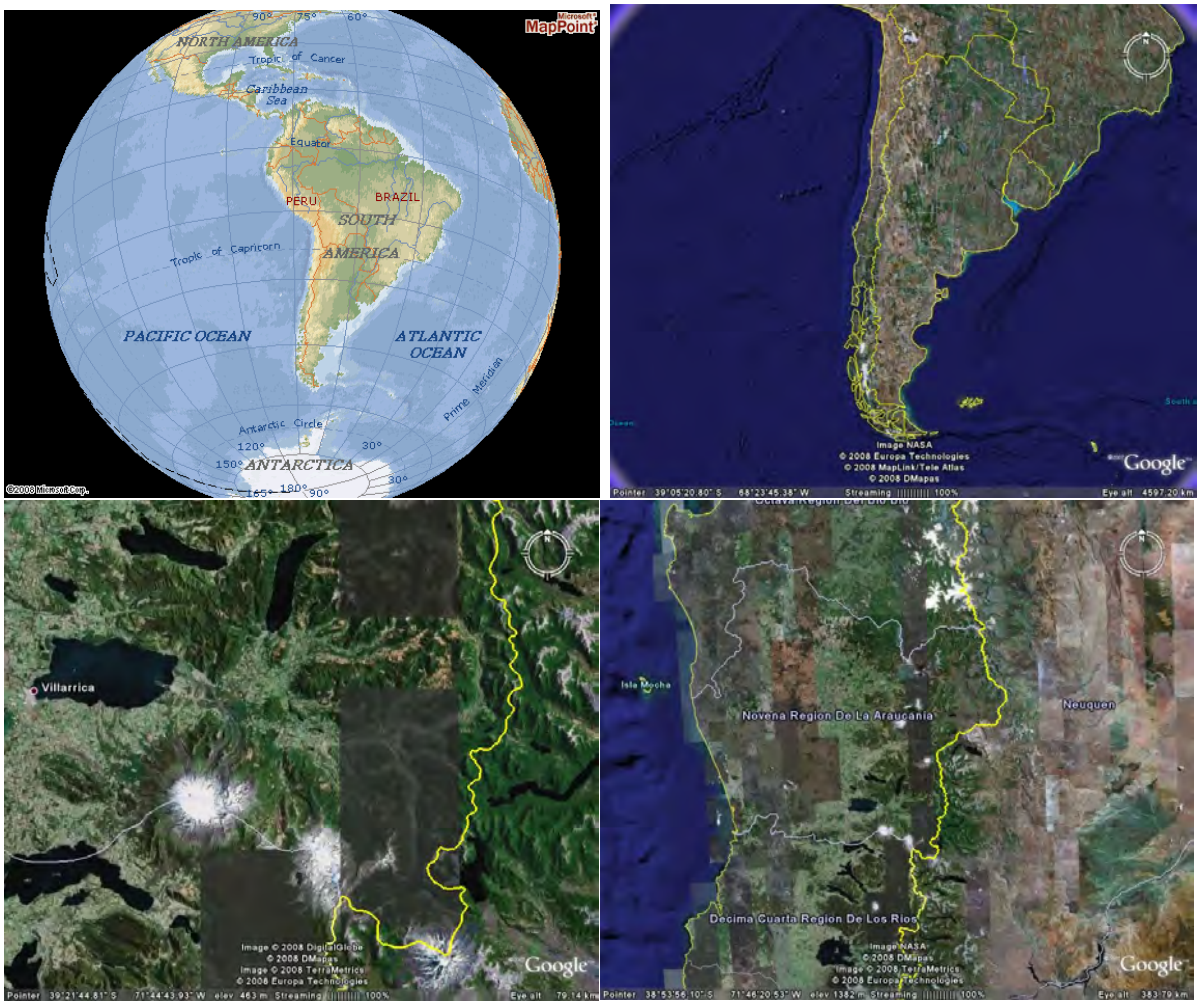

Fig. 8b: Location of study area from a Global Perspective. (Clockwise from top left: South America (Microsoft, 2008); Chile, IX region – just over half way down Chile, Study area – Lakes Villarrica and Caburga, Volcanoes Villarrica, Quetrupillan and Lanin and the Chile-Argentine boarder (Google Earth, 2007).)

Fig. 9: Species detected through camera trapping (Designs 2 and 3).

Mammals:
Puma concolor (Puma), *Oncifelis/Leopardus guigna* (Guiña/Kodkod/Chilean Cat), *Pseudalopex/Lycalopex culpaeus* (Culpeo/Andean Fox), *Pseudalopex/Lycalopex griseus* (Chilla/Argentine/South American grey fox), *Canis lupus familiaris* (Perro/Dog), *Conepatus chinga* (Chingue/Molina's Hog-nosed skunk), *Galictis cuja* (Quique/Lesser Grison), *Pudu puda* (Pudu/Chilean or Southern Pudu), *Sus scrofa* (Jabalí/Wild boar), *Sus scrofa domestica/scrofa* (Pig), *Bos taurus* (Vaca/Cow), *Capra aegagrus hircus* (Caprino/Goat), *Equus caballus* (Caballo/Horse), *Oryctolagus cuniculus* (Conejo/Rabbit), *Lepus capensis* (Liebre/Hare), species unknown - multiple (Roedor/Rodent) & *Homo sapiens* (Human).

Birds:
Pteroptochos tarnii (Huet-huet/Hued-hued/Black throated Huet-Huet), *Scelorchilus rubecula* (Chucao/Chucao Tapaculo), *Turdus falcklandii* (Zorzal/Austral thrush), (Cometocino), *Caracara plancus* (Traro/Southern Caracara), *Columba/Patagioenas araucana* (Torcaza/Chilean Pigeon), *Zenaida auriculata auriculata* (Tórtola/Chilean Eared Dove), *Nycticorax nycticorax obscures* (Huairavo/Black-crowned night heron), *Sephanoides sephaniodes* (Picaflor/Green-backed firecrown), *Colaptes pitius pitius* (Pitio/Chilean flicker)

Site	dist (m)	reps	total dist	tot hits	top day	tot dist/hit	min dist/hit	tot G	top G day	tot dist/G	min dist/G	tot C	top C day	tot dist/C	min dist/C
Hu	4,000	5	20,000	14	7	1,429	571	5	4	4,000	1,000	0	0	-	-
Ka	2,600	3	7,800	9	5	867	520	2	2	3,900	1,300	1	1	7,800	2,600
Na	9,064	4	36,256	29	17	1,250	533	12	8	3,021	1,133	3	2	12,085	4,532

Fig. 10: Snow track transect summary, winter 2007. [Site: Hu = Huellemolle LF site; Ka = Kawellucó CF site; Na = Namuncai CF site; G = guiña; C = fox (culpeo or chilla). Dist (m) = transect length per site; reps = number of times transect walked; total dist. = dist (m) * reps; tot hits = total number of mammal hits across all reps per site; top day = number of hits on day with the highest rate; tot dist/hit = total dist/tot hit; min dist/hit = top day/dist(m); tot G = total number of guiña hits across all reps; top G day = number of guiña hits on day with the highest number; tot dist/G = total dist/tot G; min dist/G = top G day/dist(m); tot C *to* min dist/C – as per G but for culpeo or chilla fox (species not distinguished).

Species	Scenario	Models	AIC	(*p*)	(*p*) SE	(psi)	(psi) SE	Naive est	SOs	sites	pm's	-log2(like)
Puma (5 day)	S07	1 gp,cst p	17.4	0.3898	0.2318	0.0924	0.0882	0.0909	9	11	2	13.402683
	S08	1 gp,cst p	48.9	0.0099	0.0049	1	0	0.1481	29	27	2	44.901105
	W07	1 gp,cst p	-	-	-	-	-	-	-	-	-	-
	S-W07	1 gp,cst p	19.52	0.3883	0.2308	0.033	0.0325	0.0323	17	31	2	15.52245
	S07-8	1 gp,cst p	68.29	0.027	0.0253	0.4498	0.3887	0.1316	29	38	2	64.294218
	W07+S07-8	1 gp,cst p	71.96	0.0251	0.0238	0.3416	0.3095	0.0862	29	58	2	67.959971
Guina (5 day)	S07	1 gp,cst p	23.2266	0.022	0.0154	1	0	0.1818	9	11	2	19.226568
	S08	1 gp,cst p	104.74	0.0377	0.0235	0.7446	0.3754	0.3333	29	27	2	100.744593
	W07	1 gp,cst p	41.92	0.061	0.0552	0.3366	0.2829	0.15	17	20	2	37.916633
	S-W07	1 gp,cst p	61.79	0.0413	0.0385	0.5091	0.44	0.1613	17	31	2	57.786975
	S07-8	1 gp,cst p	124.21	0.0327	0.0206	0.8142	0.4329	0.2895	29	38	2	120.206256
	W07+S07-8	1 gp,cst p	162.95	0.0367	0.0191	0.6816	0.3086	0.2413	29	58	2	158.95485
Culpeo (5 day)	S07	1 gp,cst p	30.37	0.033	0.0187	1	0	0.2727	9	11	2	26.373477
	S08	1 gp,cst p	149.66	0.1539	0.0352	0.3725	0.1026	0.3333	29	27	2	145.655323
	W07	1 gp,cst p	42.51	0.0412	0.039	0.4736	0.4418	0.15	17	20	2	38.511575
	S-W07	1 gp,cst p	69.73	0.0294	0.0282	0.8283	0.754	0.1935	17	31	2	65.728615
	S07-8	1 gp,cst p	179.04	0.1363	0.0315	0.3855	0.0959	0.3158	29	38	2	175.040726
	W07+S07-8	1 gp,cst p	221.1	0.1155	0.0267	0.3564	0.0866	0.2586	29	58	2	217.098872
Chingue (5 day)	S07	1 gp,cst p	15.01	0.011	0.0109	1	0	0.0909	9	11	2	11.01069
	S08	1 gp,cst p	104.14	0.0864	0.0308	0.3264	0.1277	0.2222	29	27	2	100.140987
	W07	1 gp,cst p	26.3	0.0103	0.0072	1	0	0.1	17	20	2	22.298826
	S-W07	1 gp,cst p	37.31	0.0105	0.006	1	0	0.0968	17	31	2	33.312698
	S07-8	1 gp,cst p	116.28	0.0798	0.0288	0.3052	0.1192	0.1842	29	38	2	112.284534
	W07+S07-8	1 gp,cst p	139.63	0.0748	0.0271	0.2767	0.1031	0.1552	29	58	2	135.626155
Sus scrofa (5 day)	S07	1 gp,cst p	23.23	0.022	0.0154	1	0	0.1818	9	11	2	19.226568
	S08	1 gp,cst p	61.07	0.0603	0.0323	0.1997	0.126	0.1111	29	27	2	57.072126
	W07	1 gp,cst p	26.3	0.0103	0.0072	1	0	0.1	17	20	2	22.298826
	S-W07	1 gp,cst p	46.1	0.014	0.0069	1	0	0.129	17	31	2	42.101373
	S07-8	1 gp,cst p	81.85	0.0539	0.0287	0.2746	0.15	0.1316	29	38	2	77.852196
	W07+S07-8	1 gp,cst p	104.55	0.0497	0.0264	0.2799	0.1447	0.1207	29	58	2	100.551407

Fig. 12: (continued below)

Species	Scenario	Models	AIC	(*p*)	(*p*) SE	(psi)	(psi) SE	Naive est	SOs	sites	pm's	-log2(like)
Pudu	S07	1 gp,cst p	23.23	0.022	0.0154	1	0	0.1818	9	11	2	19.226568
(5 day)	S08	1 gp,cst p	39.41	0.0074	0.0043	1	0	0.1111	29	27	2	35.409371
	W07	1 gp,cst p	-	-	-	-	-	-	-	-	-	-
	S-W07	1 gp,cst p	27.84	0.007	0.005	1	0	0.0645	17	31	2	23.83736
	S07-8	1 gp,cst p	59.92	0.0101	0.0045	1	0	0.1316	29	38	2	55.920807
	W07+S07-8	1 gp,cst p	-	-	-	-	-	-	29	58	2	-
Cow	S07	1 gp,cst p	15.01	0.011	0.011	1	0	0.0909	9	11	2	11.01069
(5 day)	S08	1 gp,cst p	66.11	0.0264	0.0247	0.5665	0.481	0.1852	29	27	2	62.112216
	W07	1 gp,cst p	107.1	0.1274	0.0423	0.624	0.1782	0.45	17	20	2	103.104233
	S-W07	1 gp,cst p	123.06	0.1167	0.0403	0.4848	0.1551	0.3226	17	31	2	119.058737
	S07-8	1 gp,cst p	77.24	0.0245	0.0231	0.5798	0.5034	0.1579	29	38	2	73.239559
	W07+S07-8	1 gp,cst p	191.77	0.0822	0.026	0.4154	0.1161	0.2586	29	58	2	187.766357
Dog	S07	1 gp,cst p	15.01	0.011	0.011	1	0	0.0909	9	11	2	11.01069
(5 day)	S08	1 gp,cst p	90.56	0.0948	0.0392	0.2913	0.1161	0.2222	29	27	2	86.56485
	W07	1 gp,cst p	16.54	0.0051	0.0051	1	0	0.05	17	20	2	12.540862
	S-W07	1 gp,cst p	27.84	0.007	0.005	1	0	0.0645	17	31	2	23.83736
	S07-8	1 gp,cst p	102.7	0.0847	0.0358	0.2792	0.1129	0.1842	29	38	2	98.69658
	W07+S07-8	1 gp,cst p	117.31	0.0781	0.0336	0.232	0.0955	0.1379	29	58	2	113.308593

Fig. 12: 5-day sampling occasion length occupancy analyses with a range of species from the study area using various scenarios. All used single-season, single-species analyses and the constant model (1 group, constant *p* ('1 gp,cst p') = psi(.)p(.)). Scenario: S07 (summer '07), S08 (summer '08), W07 (winter '07), S-W07 (summer and winter '07 combined), S07-8 (summers '07 and '08 combined), W07+S07-8 (winter '07 and summers '07 and '08 combined); AIC (Akaike Information Criterion); *p* (detection probability); pS.E. (p Standard Error); ψ (occupancy probability); ψS.E. (ψ Standard Error); Naive est (Naive Estimate); SOs (sampling occasions); sites (camera stations); pms (parameters); -2log(like) (-2log(likelihood)).

Species	Scenario	Models	AIC	*p*	*p*S.E.	ψ	ψS.E.	Naive est	SOs	sites	pms	-2log(like)
Culpeo	S07	1 gp,cst p	30.37	0.033	0.0187	1	0	0.2727	9	11	2	26.373477
(5 day)	S08	1 gp,cst p	149.66	0.1539	0.0352	0.3725	0.1026	0.3333	29	27	2	145.655323
	W07	1 gp,cst p	42.51	0.0412	0.039	0.4736	0.4418	0.15	17	20	2	38.511575
	S-W07	1 gp,cst p	69.73	0.0294	0.0282	0.8283	0.754	0.1935	17	31	2	65.728615
	S07-8	1 gp,cst p	179.04	0.1363	0.0315	0.3855	0.0959	0.3158	29	38	2	175.040726
	W07+S07-8	1 gp,cst p	221.1	0.1155	0.0267	0.3564	0.0866	0.2586	29	58	2	217.098872
Culpeo	S07	1 gp,cst p	25.31	0.075	0.0416	1	0	0.2727	4	11	2	21.310757
(10 day)	S08	1 gp,cst p	100.04	0.2138	0.062	0.3804	0.1182	0.3077	14	26	2	96.036147
	W07	1 gp,cst p	36.16	0.0923	0.0848	0.4545	0.4179	0.15	8	20	2	32.163874
	S-W07	1 gp,cst p	58.43	0.0712	0.066	0.7514	0.6625	0.1935	8	31	2	54.426194
	S07-8	1 gp,cst p	122.78	0.1966	0.0571	0.4145	0.1173	0.2973	14	37	2	118.781056
	W07+S07-8	1 gp,cst p	156.43	0.1714	0.0489	0.3985	0.1115	0.2456	14	57	2	152.426715
Guina	S07	1 gp,cst p	23.2266	0.022	0.0154	1	0	0.1818	9	11	2	19.226568
(5 day)	S08	1 gp,cst p	104.74	0.0377	0.0235	0.7446	0.3754	0.3333	29	27	2	100.744593
	W07	1 gp,cst p	41.92	0.061	0.0552	0.3366	0.2829	0.15	17	20	2	37.916633
	S-W07	1 gp,cst p	61.79	0.0413	0.0385	0.5091	0.44	0.1613	17	31	2	57.786975
	S07-8	1 gp,cst p	124.21	0.0327	0.0206	0.8142	0.4329	0.2895	29	38	2	120.206256
	W07+S07-8	1 gp,cst p	162.95	0.0367	0.0191	0.6816	0.3086	0.2413	29	58	2	158.95485
Guina	S07	1 gp,cst p	19.88	0.05	0.0345	1	0	0.1818	4	11	2	15.881219
(10 day)	S08	1 gp,cst p	88.21	0.0829	0.0498	0.7078	0.3445	0.3462	14	26	2	84.2118
	W07	1 gp,cst p	30.31	0.0333	0.0189	1	0	0.15	8	20	2	26.306054
	S-W07	1 gp,cst p	46.39	0.0385	0.0169	1	0	0.1613	8	31	2	42.386144
	S07-8	1 gp,cst p	104.29	0.0742	0.0448	0.763	0.3863	0.2973	14	37	2	100.286635
	W07+S07-8	1 gp,cst p	131.46	0.0598	0.0374	0.8311	0.459	0.2456	14	57	2	127.45787

Fig. 13: 5- *vs.* 10-day sampling occasion length occupancy analyses. Two selected species from the study area were selected and analysed under various scenarios – all using single-season, single-species analyses and the constant model (1 group, constant *p* ('1 gp,cst p') = psi(.)p(.)). Scenario: S07 (summer '07), S08 (summer '08), W07 (winter '07), S-W07 (summer and winter '07 combined), S07-8 (summers '07 and '08 combined), W07+S07-8 (winter '07 and summers '07 and '08 combined); AIC (Akaike Information Criterion); *p* (detection probability); pS.E. (p Standard Error); ψ (occupancy probability); ψS.E. (ψ Standard Error); Naive est (Naive Estimate); SOs (sampling occasions); sites (camera stations); pms (parameters); -2log(like) (-2log(likelihood)).

Model	AIC	Δ AIC	AIC wgt	Model Like'	no.Par.	-2Log(Like')
psi(.),p(.)	88.21	0.00	0.2385	1.0000	2	84.21
psi(l2,a),p(.)	88.27	0.06	0.2315	0.9704	4	80.27
psi(l2-lf),p(.)	89.05	0.84	0.1567	0.6570	3	83.05
psi(l1-cf),p(.)	89.90	1.69	0.1025	0.4296	3	83.90
psi(a),p(.)	90.21	2.00	0.0877	0.3679	3	84.21
psi(l1,l2,a),p(.)	90.27	2.06	0.0852	0.3570	5	80.27
psi(l1,l2),p(.)	91.04	2.83	0.0579	0.2429	4	83.04
psi(l1,a),p(.)	91.78	3.57	0.0400	0.1678	4	83.78

Fig. 14a: Results of covariate analysis for guiña using all three forest sizes. Data from: CF, LF and SF, 10-day SOs, summer 2008 Design 3 data. (AIC (Akaike Information Criterion); Δ AIC (variation in AIC values between each model and the highest-ranked, lowest AIC model); AIC wgt – model weight; Model Like' – model likelihood; no.Par. – number of parameters; -2Log(Like') - -2Log(Likelihood).)

Model	AIC	Δ AIC	AIC wgt	Model Like'	no.Par.	-2Log(Like')
psi(l1-cf),p(.)	93.11	0.00	0.5034	1.0000	3	87.11
psi(l1,l2),p(.)	95.09	1.98	0.1871	0.3716	4	87.09
psi(l1,a),p(.)	95.10	1.99	0.1861	0.3697	4	87.10
psi(l1,l2,a),p(.)	97.09	3.98	0.0688	0.1367	5	87.09
psi(l2,a),p(.)	100.02	6.91	0.0159	0.0316	4	92.02
psi(.),p(.)	100.04	6.93	0.0157	0.0313	2	96.04
psi(l2-lf),p(.)	100.55	7.44	0.0122	0.0242	3	94.55
psi(a),p(.)	100.80	7.69	0.0108	0.0214	3	94.80

Fig. 14b: Results of covariate analysis for culpeo using all three forest sizes. Data from: CF, LF and SF, 10-day SOs, summer 2008 Design 3 data. (AIC (Akaike Information Criterion); Δ AIC (variation in AIC values between each model and the highest-ranked, lowest AIC model); AIC wgt – model weight; Model Like' – model likelihood; no.Par. – number of parameters; -2Log(Like') - -2Log(Likelihood).)

Fig. 17: Data Treatment selection method.

Varying the selection of binary data from camera units 10B and 14B: 5-day sampling occasions were '00110000' (temporally linked) and '0000' (the first being temporally separate). The original (standard) model selected '0100' and '0' (for temporal continuity); the revised was '1100' (latter being SO's 1 and 8 combined, temporally discontinuous) and '0' and lastly revised2 was '110' and '00' (temporally discontinuous).

p	*ψ*	var (*ψ*)	*K*	no. visits	*s*	TS	cost tot' *K*	cost tot' *s*	cost tot' TS
0.1	0.1	0.1	14	8	15	203	126.08	5,256.96	5,383.04
0.1	0.2	0.1	15	9	25	377	141.84	9,106.58	9,248.42
0.1	0.3	0.1	16	9	32	519	141.84	11,760.59	11,902.43
0.1	0.4	0.1	17	10	37	627	157.60	13,361.23	13,518.83
0.2	0.1	0.1	7	5	14	98	78.80	5,056.55	5,135.35
0.2	0.2	0.1	7	5	26	181	78.80	9,388.38	9,467.18
0.2	0.3	0.1	8	5	31	249	78.80	11,281.55	11,360.35
0.2	0.4	0.1	8	5	38	300	78.80	13,592.63	13,671.43
0.3	0.1	0.1	5	4	13	63	63.04	4,552.14	4,615.18
0.3	0.2	0.1	5	4	23	116	63.04	8,379.55	8,442.59
0.3	0.3	0.1	5	4	32	158	63.04	11,482.25	11,545.29
0.3	0.4	0.1	5	4	38	191	63.04	13,860.22	13,923.26
0.4	0.1	0.1	3	3	15	45	47.28	5,484.81	5,532.09
0.4	0.2	0.1	4	3	21	84	47.28	7,587.46	7,634.74
0.4	0.3	0.1	4	3	28	114	47.28	10,294.12	10,341.40
0.4	0.4	0.1	4	3	34	136	47.28	12,276.05	12,323.33

Fig. 19: Comparison of survey costs (GBP/£) at different values of *p* and *ψ* at a precision of 0.1 standard error. (Cost/survey effort (sites and sampling occasions) using cost per visit and site.) [*p* (detection probability); *ψ* (detection probability or psi); var(*ψ*) variance or error around *ψ* value; *K* (number of sampling occasions); no visits (number of visits needed at 3 weekly checks (as in summer '08); *s* (number of sites); TS (total number of surveys or K*s); cost tot' *K* (total cost of all Ks, based on number of visits from summer '08 costing; cost tot' *s* (total cost of all sites based on summer '08 costing) and cost tot' TS (total survey cost).]