CONTENTS

LIST OF FIGURES

LIST OF TABLES

ABBREVIATIONS

ACA	Annapurna Conservation Area
WPA	World Pheasant Association
BBS	North American Breeding Bird Survey
USFS	United States Forest Service
IBA	Important Bird Area
GLM	Generalised Linear Model
MDS	Multidimensional Space Analysis
ANOVA	Analysis of Variance
SE	Standard Error
DF	Degrees of Freedom
TWINSPAN	Two-way Indicator Species Analysis
DCA	Detrended Correspondance Analysis

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ABSTRACT

Surveys designed to understand trends in populations of different species seek to maximize statistical power but, in the real world of inadequate conservation budgets, usually need to minimise financial costs. Between 1979 and 2009, intermittent surveys in the Pipar Reserve in Nepal were designed to assess trends in Galliforme populations, using call-count data; two vegetation surveys in 1983 and 2005 characterised their habitat. The present analysis aims to increase the effectiveness of investment in this long-term monitoring by assessing the statistical significance of trends within the data sets and suggesting improvements in collection protocols. Generalised Linear Modelling confirms declining trends in abundance over three decades for Satyr Trogon and Koklass Pheasant, but not for Hill Partridge. Issues of pseudoreplication and detectabilty are discussed, together with options for increasing sample size and reducing the time period required to detect significant change. Multidimensional scaling of percentage cover data from vegetation plots does not suggest a significant shift in composition. The power of long-term data collection at Pipar could be improved by minor adjustments to current protocols focused on increasing the sample size, adjusting the collection and analysis of replicate data to account for differences in detectability between sites, and increasing the detail collected on vegetation. Discussion of the implications of these findings for conservation management choices at the local level are set within the context of international requirements for monitoring of trends in biological diversity.

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1. INTRODUCTION

1.1 Meaningful monitoring of dispersed species in difficult terrain.

Demonstrating trends in biological systems is particularly problematic due to the range of variables influencing them, important specifics of context for different habitats and populations as well as the need for a long enough time period within which change can be reliably detected (Farnsworth et al., 2002, Beissinger et al., 2006, Seavy & Reynolds, 2007, Humbert et al., 2009, Teilmann et al., 2010). As an understanding of change, and the factors driving change, is of value to conservation strategies, a plethora of survey designs and analytical approaches have been developed (Tyre et al., 2003, Field et al., 2005, Kéry et al., 2005, Royle & Link, 2005).

Bailey et al. (2007) advocate that a well defined study objective can be easily translated into mathematical models, and that clear objectives can be set by first determining why monitoring is needed in any given system. As outlined by Yoccoz (2001) this will then inform exactly what system state should be monitored, which will in turn inform how it should be monitored and the details of survey design.

Monitoring can be established to serve the requirements of different stakeholders in the system from conservation staff seeking evidence to guide adaptive management to donor agencies wishing to assess the impact of their investment. **Determining why monitoring is needed** in any given system will help focus on the requirements to achieve a useful end product – confidence in the data and a sustainable level of effort. In order not to waste time or resources, investing both in establishing the rationale for monitoring trends in any given situation will improve survey design.

With the monitoring goal established, the most **appropriate state variable** - the variable used to characterize the status of the system – can be determined. State variables include species abundance or population size but occupancy (the proportion of area, patches or sampling units occupied by a species) can provide comparably robust indications of change if careful attention is paid to the spatial aspects of survey design (MacKenzie & Royle, 2005, Royle et al., 2007). Different state variables however generate different inferences about the system.

To monitor change in a particular state variable Bailey et al. (2007) suggest that survey design should be tailored not only to specific goals, but also to the biology of the target species and the logistic or economic constraints that present themselves for each situation. Further, Field et al. (2005) suggest that optimal survey design must either maximize statistical power or minimize financial costs, taking account of the constraints of management objectives, budgets and the particular idiosyncrasies of the system under study. These authors agree that survey design involves trade-offs between the real and the ideal – the reality of resources available (time, skills and finance) and the peculiarities of the species or habitat under study, set against the ideal size , scope and repetition of survey protocols. Issues for survey design therefore include spatial and temporal replication and detectability of different species.

Detectability refers to the fact that individuals or species may be present in surveyed areas but remain undetected. Rarer and less detectable species (whether due to habitat distribution or behaviour) will likely require a larger investment of resources over time than ubiquitous and highly detectable species, in order to provide a substantive evidence base for decision-makers (MacKenzie & Royle, 2005, MacKenzie, 2009). Regarding temporal replication, Field et al. (2005) explain how too few visits to each site incurs greater loss in statistical power than sampling too few sites, with two visits per site usually producing robust results. They also suggest that if the landscape is close to fully occupied (p=1.0) then it is more useful to sample intensively at relatively few sites than it is to sample widely across the landscape as this reduces uncertainty around any zeros in the sample.

There are particular complications for survey design in areas where it is difficult to obtain a large sample size. Bailey et al. (2007) consider that as the studies by MacKenzie (2005, 2006) are based on asymptotic methods (large sample size), their results may not hold for studies where a smaller sample size is obtained. The focus of this thesis is a small area located on the west bank of the river Seti in the Annapurna Himalaya, Nepal. The Pipar Reserve (28o25'N 83o57'E) covers approximately 46 km^2 and lies in a depression known as the “Pipar bowl” at 3,300 m of altitude on a spur running southwards from the Machapuchare peak at 6,990 m (Kaul & Shakya, 2001). The vegetation ranges from sub-tropical near the River Seti through temperate to alpine grasslands. Dominant trees in the canopy include *Quercus and Rhododendron* species. Vegetation surveys carried out across the Pipar bowl in 1983 and 2005 involving transects and sample plots have helped characterise the vegetation cover (Picozzi, 1983, Pouydal, 2005)..

Monitoring Galliforme populations at Pipar has particular challenges due to difficulties of access in the steep mountain terrain, but the calling behaviour of the male birds in the Spring breeding season provides a means to assess population levels. Call count data obtained over almost three decades provides a good temporal spread but the terrain imposes limits on sample size. Overcoming such issues to ensure that statistically robust evidence is available to conservation managers at this site is one subject of this study.

1.2 Aims and objectives of this study

The **aim** of this analysis is to increase the effectiveness of investment in long-term monitoring of Galliforme populations in the Pipar area of Nepal.
The specific **objectives** of this analysis are to:

1. Interrogate the periodic call count data for several Galliformes occurring in the Pipar Pheasant Sanctuary for any evidence of change in apparent abundance.
2. Evaluate the limitations of abundance studies from review of the available literature.
3. Interrogate the vegetation surveys carried out in 1983 and 2005 for any evidence of shifts in vegetation composition in the Pipar bowl
4. Propose improvements to the monitoring protocols that will allow for easier

detection of change in Galliforme numbers, and allow for inferences about any change.

1.3 Structure of thesis

The **methods** of statistical analysis applied to periodic call count data sets from Pipar are described. This analysis aims to assess whether an apparent change in call frequency provides a substantive evidence base for change in population densities for three of the five species of Galliforme in the Pipar area (Satyr Tragopan, Koklass Pheasant and Hill Partridge). Changes in vegetation cover between two surveys by Picozzi in 1983 and Poudyal in 2005 are compared.

Results are provided in two sections. Firstly, the results from the call count analysis are described for all three species in question. Secondly, comparative analysis of the vegetation data is described.

The **discussion** focuses on the strengths and weaknesses of the monitoring protocols for the long-term data collected at Pipar, and the limitations around inferences that can be drawn on trends in abundance of Galliformes. Reasons for the statistical methods applied are outlined and possible future analysis highlighted, with reference to the literature on other species and habitats. Adjustments to monitoring protocols for both call-count and vegetation data collection are proposed. The interaction between ecological data and the information requirements for conservation management are discussed, given the pressures potentially driving change in Galliforme populations and habitat in the Pipar area. The issues are drawn out in relation not just to the ecology of Galliformes and their habitat, but also to the impact of a changing social context and a changing climate. The social context includes both human demographics and livelihood options that impact on Galliforme habitat. These include use of non-timber forest products that provide income or other livelihood benefits to forest-adjacent communities, as well as changes in human use of the habitat. The changing climate has both ecological (habitat and range change) and human (ecosystem services and potential migration) elements. The information to be derived on Galliforme abundance and habitat use from new data-collection protocols is set in the context of human use of the habitat. The value of collecting robust monitoring data for species abundance and habitat health over time is described with reference to the trade-offs conservation managers make in choices at the local level. A rationale is drawn out for local level trend data being able to inform national and international commitments to monitoring trends in biodiversity.

2. BACKGROUND

2.1 Abundance and detectability.

Many authors support the importance of accurate data on abundance and population trends in order to improve conservation and management strategies (Farnsworth et al., 2002, Beissinger et al., 2006, Seavy & Reynolds, 2007, Humbert et al., 2009, Teilmann et al., 2010). In some instances, such as for European Union Species of Community Interest, the ability to detect a population trend (annual decline of 1% in a 6 year reporting period) is described as a legal obligation (Hovestadt & Nowicki, 2008).

Land birds are often surveyed using point counts, as it is one of the more efficient methods of obtaining abundance data. (Farnsworth et al., 2002, Forcey et al., 2006, Thompson & La Sorte, 2008). However, the use of point counts as indices of abundance to detect population trends has attracted scrutiny due to the assumptions and errors associated with it and authors such as Thompson (2002, 2008) and Hovestadt & Nowicki (2008) suggest it should be treated with caution. One of the main assumptions is that detection probability is not taken into account when explaining count variations. Some models such as the double-observer count approach, created by Cook & Jacobsen (1979) for aerial surveys, have been incorporated into land based surveys that account for the probability of detection for each bird species (Nichols et al., 2000). Forcey et al. (2006) compare two of these approaches and describe how corrected abundance estimates can be calculated to account for those individual birds that are present but not detected. They address feasibility of monitoring, higher detection probabilities and associated accuracy in analysis. Although Forcey et al. (2006) report the improvements that these models bring to abundance data, Thompson & La Sorte (2008) and Beissinger et al. (2006) suggest that there is still some resistance to incorporating them into monitoring strategies. Through investigation of data from the United States Forest Service (USFS) Southern Region bird monitoring programme into the effects of detection probabilities for monitoring surveys Thompson & La Sorte (2008), conclude that there is strong evidence to show that detection probability differs between species, years and observers; they suggest that investigators need to incorporate this into survey design.

Non-uniform detection probabilities across a landscape can introduce bias when determining population status and change over time (Royle, 2006). Non-detection of a study species can provide “false negative” data for a certain site and not necessarily mean that it does not occur there at the time of data collection. Such data can have substantial effects on precision of population parameter estimates (Tyre et al., 2003). This imperfect detection remains one of the biggest problems to overcome (MacKenzie & Kendall, 2002, MacKenzie, 2006, Royle & Link, 2006). Underestimates in population numbers can also lead managers to expend scarce resources on a system where populations are healthy (MacKenzie, 2009). Overcoming Imperfect detection has been a major subject of concern for population modelling in recent years and several survey techniques have been developed to reduce bias and improve accuracy of estimates for population size and habitat use, particularly for rare species with detection probabilities of less than 1 (Farnsworth et al., 2002, Kéry et al., 2005, MacKenzie & Royle, 2005, Royle, 2006, Royle et al., 2007, MacKenzie et al., 2009)

Kéry et al. (2009) report on models run for sand lizard population counts from the National Dutch Reptile Monitoring Scheme. This work illustrates the importance of detectability and shows that over the first 5 years of the study, detectability increased and was highest for the most experienced observers. Interestingly they note that when detectability was not included in the analysis, the increase in the inland population was not detected.

As a result of needing to account for imperfect detection, many recent studies have included spatial and temporal replication into the survey method (MacKenzie & Royle, 2005). Repeated site visits can improve assessment of detectability and this is explored by Royle & Nichols (2003). For survey designers and conservation managers this creates a trade-off between time and resources expended on spatial replication (the number of sites that are to be surveyed in the area) and temporal replication (the number of repeated surveys at each sampling site) (Field et al., 2005, MacKenzie & Royle, 2005, MacKenzie, 2009, Field, 2007). Seavy & Reynolds (2007) refer to various aspects of survey design that affect the power to detect trends, including temporal replication. They argue that alerting managers to factors in survey design that can reduce bias can help them focus effort and resources on minimising their impact.

Teilmann et al. (2010) investigated how survey frequency affects the statistical power to detect trends on 30 years of data for 7 subpopulations of harbour seals in southern Scandinavia. They found that carrying out annual surveys typically doubled the power of detecting a trend compared to surveys carried out every two years (Teilmann et al., 2010). Replicate surveys within each year also increased the power to detect trends. These results indicate that, given natural variation within ecological systems, more replication is required in order to improve inferences about change within those systems. For Harbour seals they suggest that mean results from at least three replicate surveys per year are necessary to provide robust data on which conservationists could make more informed decisions for management and allocation of resources. Dennis et al. (2010) supports the conclusion that replicate surveys for monitoring biological populations increases their efficiency in detecting trends.

Observer differences as well as species detection rates with distance are recognized as sources of error when conducting surveys to determine population numbers for terrestrial birds (Cunningham et al., 1999, Diefenbach et al., 2003). Diefenbach et al. (2003) investigated the effects of different observers when identifying several species of terrestrial birds, and found that as distance increased beyond 25metres, many species had a detection probability of less than 1, and at 50 meters 60% of the species in question were missed entirely. In addition to their results on changes in detection probability with distance, Diefenbach et al. (2003) also concluded that there was a large variation in detection rates between observers. Moore et al. (2004) go further and recommend that for count data to reliably give an index of abundance or density, correction should not only be for variation in detection probabilities and observers, but also environmental conditions.

2.2 Characteristics of the Pipar area

The country of Nepal lies in the transitional zone between the eastern and western parts of the Himalaya. With large altitudinal variation and diverse climatic conditions, this small country hosts the richest diversity of bird species in Asia and 27 Important Bird Areas (IBA's) (Baral & Inskipp, 2005, Subedi, 2010). Of the 281 species within the 77 genera of the Order Galliformes, 22 are found in Nepal.

The Annapurna Conservation Area (ACA) is one of the world's most biologically diverse reserves (Kaul & Shakya, 2001) hosting 485 bird species (Baral & Inskipp, 2005) ACA is the only protected area of Nepal where all six species of Himalayan pheasants occur. Twelve Galliforme species have also been reported in the Seti Khola watershed areas of Kaski district (Poudyal, 2009).

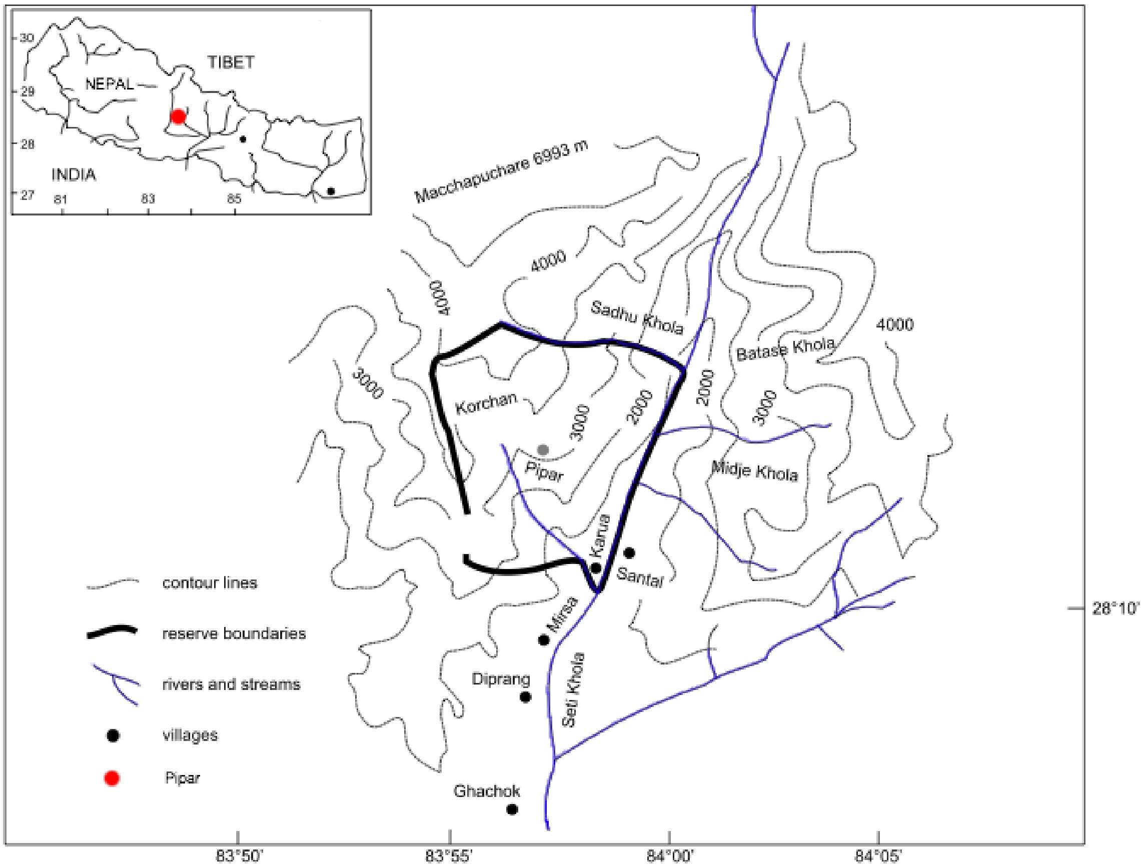

Figure 2.1 *Map from Kaul and Shakya (2001) showing the Pipar reserve and the surrounding area.*

The Pipar reserve is an area of 42 km^2 which hosts five of the Himalayan pheasant species, Blood Pheasant, Koklass Pheasant, Himalayan Monal, Kalij Pheasant and the globally near-threatened Satyr Tragopan. The World Pheasant Association (WPA) has been involved with this area since their first survey in 1979. Over that time a variety of ecological data has been gathered, including: periodic call count surveys, vegetation sampling (in 1983 repeated in 2005) and a wider survey in the upper Seti Khola forests that predicted the distribution of suitable habitat. The ecological research is aimed at identifying population status and trends of Galliformes,, and relating this to habitat preference and health. The area has been under conservation management during the last thirty years, more recently with awareness and income generating activities linked to habitat conservation. Changes in the population status of Galliformes could be due to either ecological change, shifting patterns of resource use in this habitat by local people, or changes in management priorities. A clearer understanding of observed changes, their significance and the drivers behind them, would be of value to conservation managers as they seek to adjust conservation strategies to retain the value of the habitat in the face of both natural and human-induced change.

2.3 Characteristics of Galliformes at Pipar relevant to survey design.

Three species of Galliformes from Pipar are the focus of this study:

2.3.1 The Koklass pheasant (*Pucrasia macrolopha nipalensis)* is the most vocal of the pheasants at Pipar giving loud pre-dawn calls during the breeding season and autumn. This is fortunate for a call count survey, as it is otherwise difficult to detect, being the shyest and most secretive of the Pipar pheasants. It occurs mainly in Rhododendron and mixed broadleaf forest and between 2930 and 3350 m altitude. Koklass remain in pairs or small family groups throughout the year.

2.3.2 The Satyr Tragopan (*Tragopan satyra)* is also shy but easier to observe than the Koklass. It occurs in thick Rhododendron and mixed forest on both gentle and steep slopes at Pipar, between 2750 and 3345 m altitude. Males also call before dawn during the breeding season between April and June.

2.3.3 The Hill Partridge (*Arborophila torqueola)* occurs in tropical moist montane or lowland forests and is more widespread than the other Galliformes monitored. It also occurs in pairs or small family groups, and breeds between April and June. The Hill Partridge utters hen-like contact calls when feeding, making it easier to detect. (Birdlife International, 2010)

Common to all three is the assumption that as only calling males are detected in call counts, numbers of other adults can only be deduced.

3. METHODS

3.1 Pheasant call count data collection

Since the Late 1970's and early 1980's, after the Pipar Pheasant Sanctuary was declared as a reserve, several ecological surveys were carried out every few years. Gaston's (1980) technique has been replicated in subsequent surveys such as that by Kaul and Shakya in 1998, in order to allow for comparative analysis for change. The protocols involved positioning observers at 4 pre-determined listening stations, across the Pipar Bowl before dawn. After Kaul and Shakya's survey in 1998, two more listening stations were added to the surveys. On a few occasions, some listening stations were not surveyed due to difficulties in reaching the listening stations before dawn, or due to difficult weather conditions (Kaul and Shakya, 2001). Observers listened for the distinctive calls of the species in question and recorded the apparent position and time of the calling individuals onto a recording sheet. This technique has been used in many pheasant surveys throughout the Himalayas (Gaston and Singh, 1980, Garson, 1983, Khaling et al., 1998). Double counts between adjacent listening stations were eliminated by matching observers' recording sheets and not counting any birds that were heard at the same time and place. Another precaution taken to eliminate double counts was to limit the time spent recording birds to 15 minutes after the first bird of a particular species was heard. This is due to the fact that the birds tended to move around after calling for 15 – 20 minutes from a stationary point (Kaul and Shakya, 2001), and the chances of adjacent observers recording the same bird as a separate one would increase. Distant calling birds that were only faintly heard, and therefore difficult to determine position and distance were not counted. Poudyal et al., (2009) estimated that the audible range from each station was approximately 300 meters.

Observers reached the listening stations approximately 20-30 minutes before the first birds started calling (04h15); although dusk call counts were attempted, the birds did not call consistently and so these counts were abandoned. Throughout all the surveys, except for the first in 1979, listeners recorded call counts for at least 3 consecutive mornings to account for inconsistencies in calling birds, and adverse weather conditions affecting the number of birds heard. In 1979 recordings were only made over two mornings.

Several species were recorded, Satyr Tragopan, Koklass Pheasant, Hill Partridge and the Himalayan Monal, although the Himalayan Monals calls were very sporadic and inconsistent so this species was less amenable to counts (Kaul and Shakya, 2001, Poudyal, 2009). Also, counts for the Hill Partridge only began after 1998. However this provides an opportunity for an interesting comparison between the longer time series and data on a species with a set that has been collected over a shorter time series. This could provide an indication of how many years of surveying are necessary to pick up any trend that there might be.

All surveys were carried out in the spring month of May when the calling birds are at their most active, coinciding with the breeding season when the males call regularly. This is a consideration for abundance assessments as only the males are counted.

3.1.1 Statistical modelling

All analyses and graphics were carried out using the statistical package R version 2.10.1.

Standard linear regressions are often used to statistically analyse and detect trends in time series data that follows a Gaussian distribution. Count data is not normally distributed however as there can be no counts lower than 0 which produces a skewed distribution with non-constant variance that increases with the mean. This and the fact that counts only produce integers means normal Linear regression is not suited for analysis of change (Crawley, 2007). Generalised Linear Models (GLM) of the Poisson family were therefore used to detect trends in the call count data, with the response variable being 'call counts' measured against the explanatory variable 'Year'. When running the GLM's, the significance of the trends that were detected were assessed using a likelihood ratio test by nesting, within the more complex model, a simpler model that contains only a subset of the variables that are also present in the more complex model. As previously described, this type of count data is only an index of abundance and so caution is required when describing any change in apparent abundance.

Within site variation and the problems of Pseudoreplication (repeated observations from the same sample not being treated as independent) associated with the data were addressed by summarising across repeat visits to give a single value per site (maximum counts) then re-running the GLM's for each species. Mean counts were also used but due to non-integers being produced this way, GLM with normal error distribution was carried out for comparison. The graphical illustration of the trends detected were created using the average of the means for all sites, this allowed for clearer graphical representation of the trends detected, and error bars were fitted to give visual support to statistical significance or lack thereof, of trends detected.
Justification for using maximum or mean counts for this analysis is highlighted in the discussion.

Environmental variation was not considered in analysis, due to the fact that there was a lack of information regarding weather conditions for each survey carried out.

3.2 Vegetation survey methodology

Ordination analysis of vegetation data was carried out using the statistical package "vegan" in R version 2.10.1.

Two intensive vegetation surveys were carried out in 1983 by Picozzi and in 2005 by Poudyal. Picozzi marked out a 750m transect (T1) across the higher part of the Bowl (starting at an altitude of 3310m) running due North, and surveyed 10x10 meter plots at intervals of 50 meters along this transect. This was then repeated by creating transects down the valley in an easterly direction, from the 10x10meter plots every 100meters along T1, creating subsequent transects (V1, V2, V3…etc). These were then surveyed in less detail and at intervals of 30 meters along each one, ending when Picozzi and his colleagues decided that there was no longer any clear evidence of the recent effects of man or domestic stock, or the slope was too steep to survey safely. At this final point, detailed information was once again collected in the same manner as for the 10x10 meter plots on T1. The data collected at each $100m^2$ plot consisted of detailed information of the number of different plant species found as well as estimations of ground cover for 5 main groups, Plants, Moss, Leaf Litter, Rock and Bare Ground. Geographical information such as slope and aspect were also recorded but are not used in this study. In 2005, Poudyal repeated this survey using the same methods to allow for comparison between the two years. However, he also took more detailed information for 10x10 meter plots at every 30 metre interval along the subsequent transects down the hill (V1, V2, V3, etc…). Due to the fact that Picozzi (1983) did not take GPS measurements for all the plots, Poudyal's (2005) survey may not have exactly matched all the plots although care was taken to be as accurate as possible to obtain comparable data. Also, by taking GPS points at each plot, Poudyal has provided valuable information for future vegetation surveys trying to match the methods. These limitations are not ideal for comparison between the two surveys as paired samples provide more statistical power for evidence of change. However estimated percentage ground cover of the 5 groups described does provide a general description of the landscape and allow for a general comparison; the lack of exact overlay of plots between the surveys is not considered critical for this analysis. Should the precise species information be required for a comparison the lack of exact overlay would be important.

3.2.1 Statistical modelling

The main point of interest was to investigate any change over the 22 year period between the sets of data. Plot by plot data on percentage ground cover independently estimated visually by several observers, and then averaged out provides an indication of the vegetation composition over the Pipar Bowl. Percentage ground cover for all the plots in 2005 and for most of the plots in 1983 are available for a comparison analysis. For this analysis ordination statistics Multidimensional Space analysis (MDS) were applied to the vegetation percentage cover data for the plots that have this data for both years. Plot location may have an influence on cover type so only the plots that were surveyed in this manner during both years were compared. The influence of location that new plots would have on the ordination was controlled for by excluding them from the MDS analysis and so ensuring that Year is the only explanatory variable. Ordination gives each plot coordinates in multivariate space based on vegetation cover. These coordinates were than plotted graphically. Due to uncertainty as to whether the plots in the two surveys overlayed exactly (as described in Methods) a degree of caution is taken when making inferences of change or lack thereof.

Analysis of Variance (ANOVA) tests were performed on each cover type, testing to see if there was any significant association between type of cover and Year. As described, the two surveys are separated by 22 years and although ANOVA may not be the most informative test, due to there being only two explanatory variable data points (1983 and 2005), it was considered useful to statistically check any evidence of change shown by the ordination.

4. RESULTS

The first step of the analysis, based on consideration of the different options described above , was to carry out generalised linear models (GLM) on the Pipar call count data across all years of collection.

4.1 Analysis of pheasant call count data – Detecting trends

By running GLM's on the call count data for the three species Satyr Tragopan, Koklass Pheasant and Hill Partridge independently, including **all** counts and treating them as independent replicates, there appear to be significant declines in numbers for Satyr Tragopan and Koklass Pheasant since monitoring in Pipar began in 1979 (Table 4.1). This suggests that the response variable "counts" is significantly lower over the unit of time, in this case years.

Table 4.1 *Results representing Poisson GLM's of* ***all*** *counts for the three study species at Pipar in relation to year, with statistical values showing the significance of the trend over time*

Species	Parameter Estimate	Standard Error	Chi squared	Degrees of Freedom	p-value
Satyr Tragopan	-0.011	0.004	8.29	1	0.004*
Koklass Pheasant	-0.038	0.005	61.40	1	0.0001*
Hill Partridge	-0.077	0.04	3.54	1	0.06

* Significant test statistic

Table 4.2 *Results representing Poisson GLM's of the* ***maximum*** *counts for all three study species at Pipar in relation to year, with statistical values showing the significance of the trend over time.*

Species	Parameter Estimate	Standard Error	Chi squared	Degrees of Freedom	p-value
Satyr Tragopan	-0.008	0.006	1.98	1	0.16
Koklass Pheasant	-0.03	0.007	18.2	1	0.0001*
Hill Partridge	-0.03	0.063	0.28	1	0.6

* Significant test statistic

Table 4.3 ***Table 4.3** Results representing normal error GLM's of **mean** counts for all three study species at Pipar in relation to year, with statistical values showing the significance of the trend over time*

Species	Parameter Estimate	Standard Error	Chi squared	Degrees of Freedom	p-value
Satyr Tragopan	-0.055	0.033	13.4	1	0.104
Koklass Pheasant	-0.13	0.033	75.74	1	0.0003*
Hill Partridge	-0.29	0.23	4.46	1	0.23

* Significant test statistic

Pooled means of the call counts for each year were used to represent the decline graphically. Standard errors for each point were calculated to fit the y error bars.

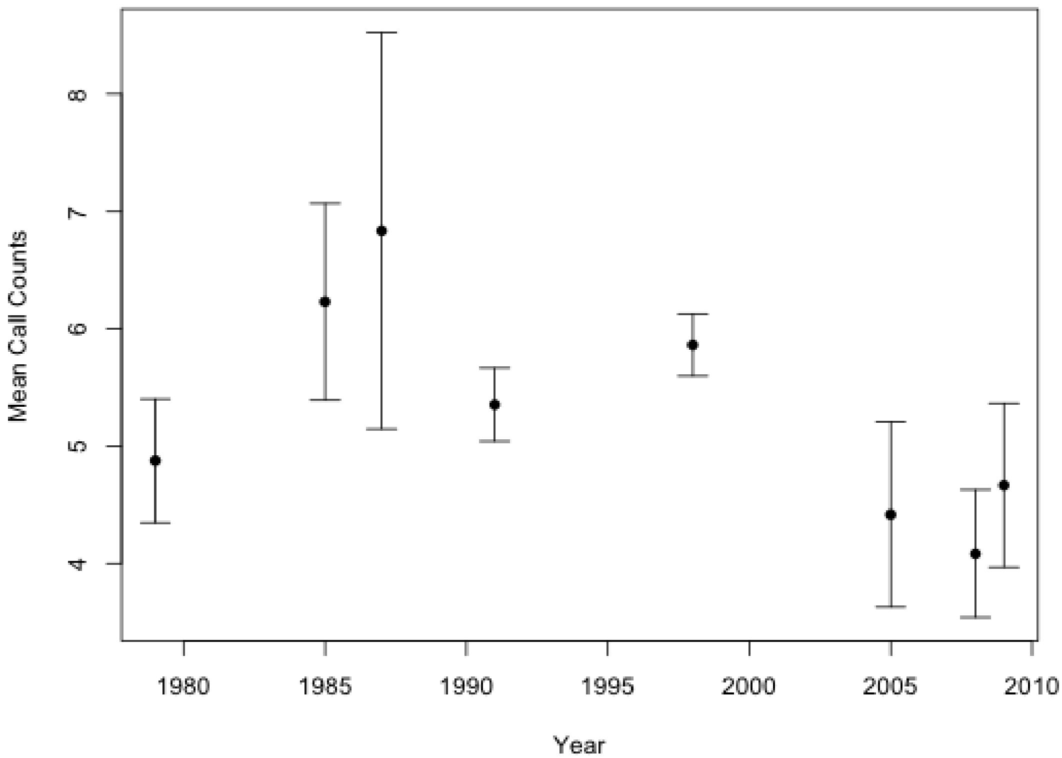

***Figure 4.1** The graph represents apparent decline in numbers of Satyr Tragopan at Pipar, based on the pooled means for all counts during each year of survey. Y-error bars are fitted from calculations of Standard Errors.*

Figure 4.1 allows for visualisation of the decline in numbers for Satyr Tragopan, however the GLM using the **mean** counts shows this to be a non-significant decline (**Table 4.3**). Only by performing GLM's using **all** the call count data for Satyr Tragopan can a significant decline be detected.

The overall trend seems to be one of a decline however (negative parameter estimates in Table 4.3 support this). Removing the data from 1979 (a possible anomaly in the data, due to some unseen effect) and re-running the GLM using **maximum** and **mean** counts shows a significant decline is detected (**Table 4.4**).

Table 4.4 *.Results representing GLM's using* ***all, maximum*** *and* ***mean*** *counts for Satyr Tragopan at Pipar, in relation to year, with statistical values showing the significance of the trend over time, excluding data from 1979.*

Counts used	Parameter Estimate	Standard Error	Chi squared	Degrees of Freedom	p-value
All	-0.016	0.005	12.63	1	0.0004*
Maximum	-0.017	0.007	5.3	1	0.02*
Mean	-0.09	0.04	23.7	1	0.03*

There is a much clearer trend seen with the Koklass data (**Figure 4.2)** and it is supported by the statistical values from the GLM analysis (**Tables 4.1 - 4.3**.) The data from year 2009 appears to be very different from the other years. Between 1979 and 2008, there appears to be a decline of about 30% and then there is a dramatic drop in 2009 of a further 30%. The last three surveys revealed visibly lower numbers of pheasants. so the data was re-tested using the same GLM models as in the previous analysis (for all, maximum and mean counts) but sequentially removing the more recent data, to see at what point the GLM loses its significance, and therefore suggesting how long it is taking for a trend of this magnitude to be detected with this data set (**Tables 4.5 – 4.7**).

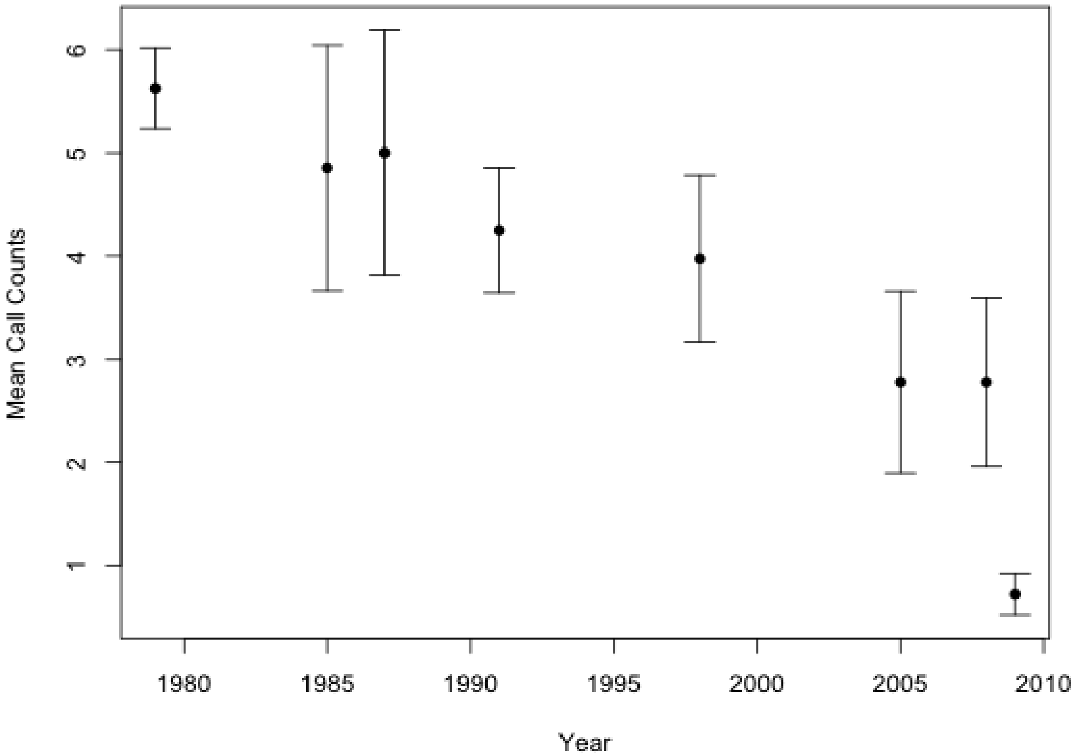

Figure 4.2 *The graph represents decline in numbers of Koklass Pheasant at Pipar, based on the pooled means for all counts during each year of survey. Y-error bars are fitted from calculations of Standard Errors.*

Table 4.5 *Results representing 3 Poisson error GLM's of* ***all*** *counts for Koklass Pheasant at Pipar, sequentially excluding the most recent 3 years of survey*

Excluded Years	Parameter estimate	Standard Error	Chi Squared	Degrees of Freedom	p-value
2009	-0.03	0.005	23.63	1	0.0001*
2008,2009	-0.02	0.007	5.3	1	0.0004*
2005,2008,2009	-0.06	0.011	23.7	1	0.16

* Significant test statistic

Table 4.6 *Results representing 3 Poisson error GLM's of* ***maximum*** *counts for Koklass Pheasant at Pipar, sequentially excluding the most recent 3 years of survey*

Excluded Years	Parameter estimate	Standard Error	Chi Squared	Degrees of Freedom	p-value
2009	-0.02	0.008	6.86	1	0.009*
2008,2009	-0.02	0.01	5.61	1	0.02*
2005,2008,2009	-0.01	0.016	0.58	1	0.45

* Significant test statistic

Table 4.7 *Results representing 3 normal error GLM's of* ***mean*** *counts for Koklass Pheasant at Pipar, sequentially excluding the most recent 3 years of survey*

Excluded Years	Parameter estimate	Standard Error	Chi Squared	Degrees of Freedom	p-value
2009	-0.10	0.04	34.78	1	0.01*
2008,2009	-0.11	0.05	23.7	1	0.04*
2005,2008,2009	-0.09	0.09	5.67	1	0.32

* Significant test statistic

From this sequential analysis it is evident that the significant trend is detected after the 2005 survey was carried out (using **all**, **mean** and **maximum** counts). This suggests that collecting data in this manner for the area of Pipar for approximately 25 years would be enough to detect a trend. Another important factor here however is not just the **time** to pick up a trend but also the **magnitude** of change that is detectable.

It appears there is an approximately 30% decline in Koklass pheasant abundance over the first 25 years (**Figure. 4.2**). A decline of 30% in numbers is a large decline not to be detected by the statistics applied here. If it is important to the reserve managers to have significant evidence behind any change in management practice, then monitoring in this way is not sufficient and protocols need to be altered so that trends can be detected more rapidly.

One method of improving the statistical power in order to pick up these sort of declines as quickly as possible is to increase the sample size. Using the data that was collected for Koklass pheasant, the data were re-sampled to include an imaginary increase in the sample size from 6 listening stations to 10. Even a small increase in the number of listening stations increased the significance of the test statistic by 3 decimal places. (p=0.0001 for 6 stations to p=1.10e-07 for 10 listening stations).

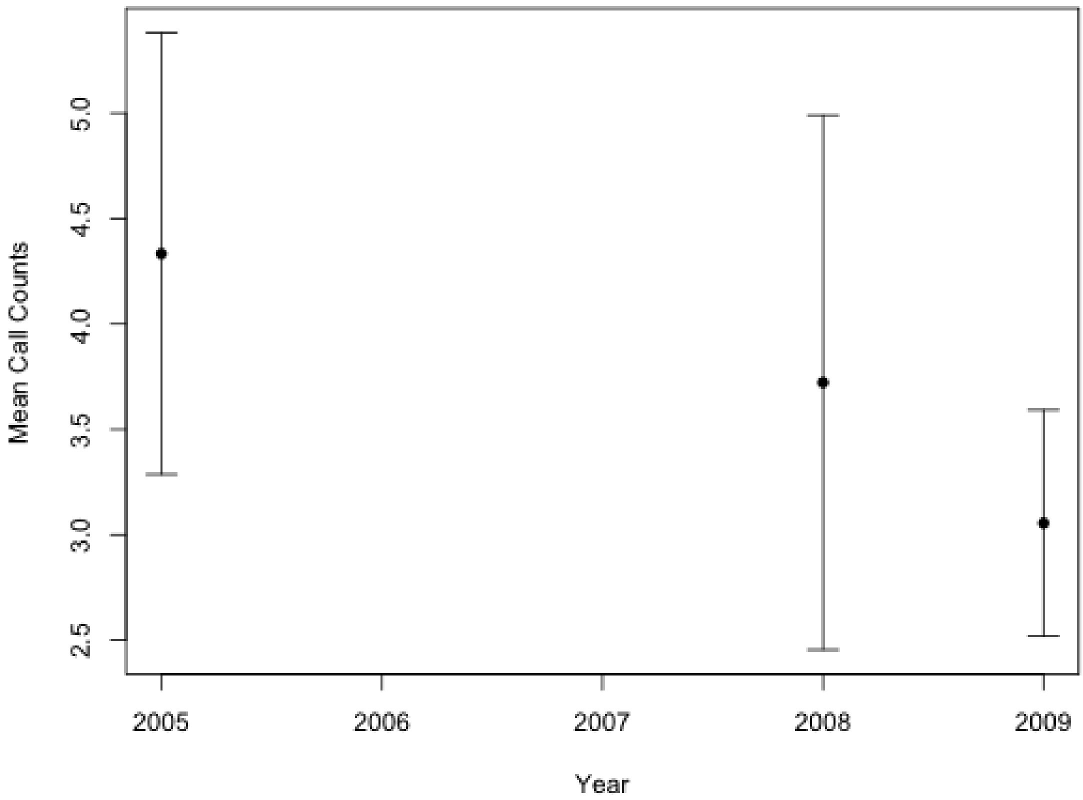

Figure 4.3 *The graph represents decline in numbers of Hill Partridge at Pipar, based on the pooled means for all counts during each year of survey. Y-error bars are fitted from calculations of Standard Errors.*

For the data collected on the Hill Partridge since 2005, although there appears to be a decline in numbers (negative parameter estimates in **Tables 4.1 – 4.3**), the GLM test revealed a non-significant result and has not detected any significant decline.

4.2 Analysis of vegetation data - evidence of change.

Multidimensional scaling of percentage cover data from vegetation plots (**Figure 4.4**) does not suggest that there is any significant shift in the composition of the plots. The closer together the points on the graph, the more similar they are. If there was evidence of 'clumping' of plots then this would suggest that there is a shift in vegetation composition between years.

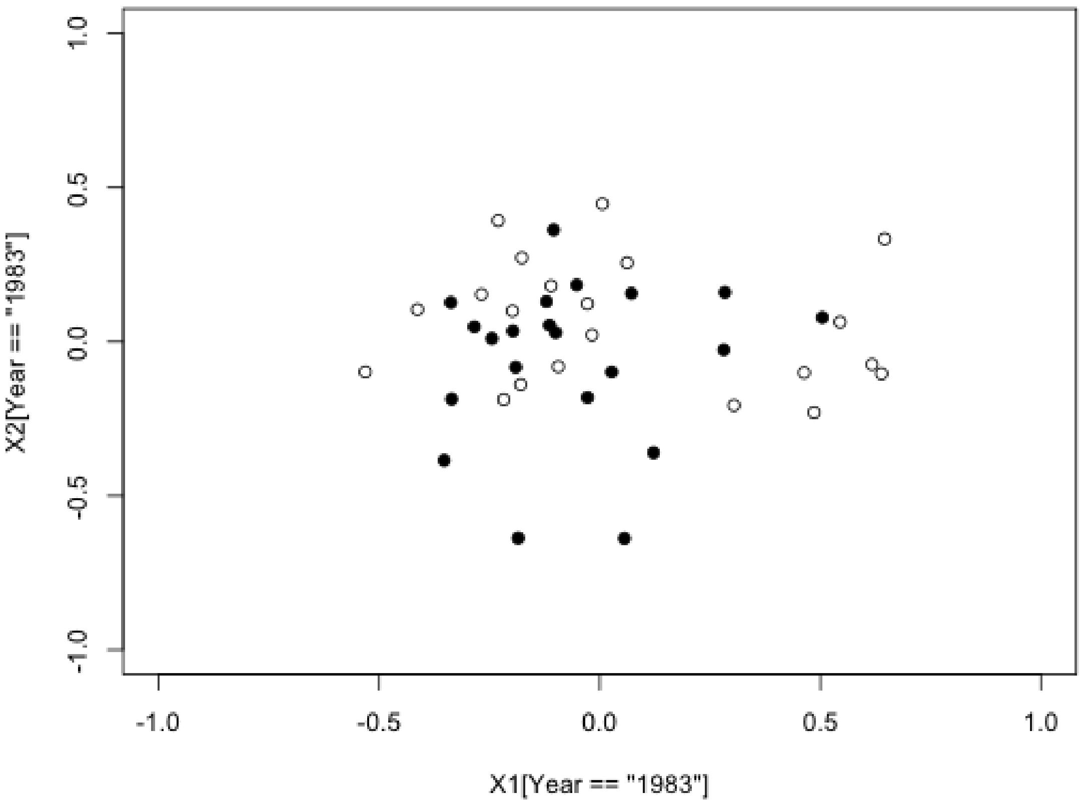

Figure 4.4 *Graph showing vegetation plots at Pipar in multivariate space using coordinates calculated by ordination based on cover types Plant, Litter, Moss, Rock and Bare Ground for data from 1983 (Black spots) and 2005 (white spots).*

The ANOVA test results are displayed in **Table 4.8**. with the mean values for each cover type per year in T**able 4.9**.

Table 4.8 *Statistical values for ANOVA tests on each vegetation cover type at Pipar against Year.*

Plant	Leaf Litter	Rock	Moss	Bare Ground
F = 0.093	F = 3.313	F = 3.359	F =0.660	F = 0.292
P = 0.762	P = 0.076	P = 0.074	P = 0.421	P = 0. 592

Table 4.9 *Mean values of each vegetation cover type at Pipar for each year.*

Year	Plant	Leaf Litter	Rock	Moss	Bare Ground
1983	37.86	34.19	2.05	14.19	11.71
2005	39.95	22.48	6.67	17.19	13.71

The ordination analysis did not highlight any significant change, and this is supported by the statistical values from the ANOVA test.

5. DISCUSSION

5.1 Change we can believe in? Strengths and weaknesses in the long-term data sets for Galliformes at Pipar.

It is clear from the literature on species surveys that count data can act as a primary source of information on population change, but that caution must be exercised when using this information to analyse the significance of changes in population numbers as counts are only indices of population size (Link & Sauer, 1998). Differences in population size over a time series can be a result of misinterpreting temporal and geographic variations in the proportion of animals counted (Link & Sauer, 1998), and that therefore should be incorporated into the analysis of population parameters (Kéry & Royle, 2008)

5.1.1 Issues of pseudoreplication & detectability

One of the difficulties of using GLM for this data set is that pseudoreplication has been introduced by surveying the same sites several times. There may be differences in vegetation or habitat between sites that affect abundance of different Galliforme species. The Galliformes at each survey site (area around the fixed listening stations) might be considered in this case as independent groups, so the sample might not be a true random representation of the overall population. Replicates should be independent and not form part of a time series (Crawley, 2007) – which is not the case for data that has been collected from the same place on successive occasions (temporal pseudoreplication). Similarly replicates grouped together in one place are not spatially independent (spatial pseudoreplication). While replicates could introduce pseudoreplication, and could reduce power of GLM statistics (or render them invalid) methods by Link et al. (1994) and Kéry et al. (2009) include uncertain detection of study species into the model and so account for this pseudoreplication; in fact they show that replicate surveys improve the power of GLM's to detect a trend. The data collected in Pipar is typical of many bird count surveys, such as those used to collect data for the North American Breeding Bird Survey (BBS), which was investigated by Link et al. (1994, 1998). These counts are collected in the same sample sites either each year or every two years, and change is modelled across the sites over time. The variability in the count indexes, introduced by variability in population size at each site, as well as variability due to uncertain detection, can be mitigated by introduction of repeat counts at survey sites (Link, 1994, Kéry & Royle, 2008). This is, then, the case for Pipar and reduced within-site variation can be assumed. However, Link et al. (1994) stress that although repeating counts at sites does provide a method for mitigating variation within-sites, replicating surveys at each site is not always as cost efficient as initiating new sites into the survey. However, If the aim is to minimise variance of counts during the surveys and if within-site variability is large, then replication of sites might be more desirable as an option, as long as the cost of doing so is far less than sampling new sites (Link et al., 2004).

Other than accounting for detectability in the modelling, another way of overcoming this pseudoreplication is to use a mixed effect model. The Mixed Effect Model treats sample sites as independent and by doing so accounts for variability between sites and mitigates the effects of pseudoreplication. A recommendation for future analysis of these data sets is therefore to carry out Mixed Effect Models for counts against year to look for trends, without having to incorporate detectability and yet avoiding the limiting factor of pseudoreplication.

Pseudoreplication could also be overcome by summarising the data for each site per year and re-running the GLM's using either the maximum count or the mean counts. This would effectively transform the data into non-replicate data mitigating the effects of pseudoreplication, but perhaps at a cost to statistical power to detect a trend (Kéry et al., 2009).

5.1.2 Trends in Galliforme abundance at Pipar.

Considering the limitations to running GLM's on this call count data, the results using maximum and mean counts per site against year are displayed in tables 4.2 and 4.3 as well as results for using all the counts (Table 4.1). Considering that detectability was not taken into account when running the GLM's the results for mean and maximum counts are arguably more accurate in describing the decline in pheasant abundance in Pipar.

For **Satyr Tragopan** a significant decline in abundance is detected only when **all** counts are used (Table 4.1). Only by excluding data from 1979 is the decline detected when using the **mean** and **maximum** counts (Table 4.4). In the absence of any indication that there were unusual factors in the data collection that affected the Satyr Tragopan data exclusively, it is not possible to justify excluding 1979 as an anomaly. However, unusual weather or other ecological conditions might have affected the amount of calling Satyr Tragopan males, to which the other species had a different response. This apparently low count in 1979 may just indicate that Satyr Tragopan numbers were down in 1979. This then could be attributed to a substantial nonlinear variation between years for this species, in which case treating year as a categorical variable rather than a linear variable may be more appropriate for this data. Further investigation into this is needed.

For **Koklass Pheasant,** the apparent decline in abundance is supported by the outcome of the GLM analysis (Tables 4.1-4.3) as the Koklass data shows a dramatic drop in numbers during the last survey in 2009. Again, it might be assumed that Koklass pheasant reacted differently from the other species to some aspect of the 2009 survey. However, testing this by excluding the 2009 survey showed that the result remained significant (Tables 4.5 – 4.7) albeit at reduced strength.

For **Hill Partridge**, Although there is a suggestion of a decline from the negative parameter estimates presented in tables 4.1 – 4.3, no significant trends were detected by all the GLM analysis performed on this data set. This is most likely to be due to the fact that only the last three years of surveys collected data for this species.
An important factor in abundance trends analysis is the amount of time taken to detect any trends present in the population. These results for Hill partridge highlight the fact that with these monitoring protocols in Pipar, a substantial time series is needed in order to detect a trend. This is supported by the GLM analysis on Koklass Pheasants excluding the last three years of data (Tables 4.5 – 4.7). These suggest that in order to detect a decline of approximately 30%, 25 years of surveying is required. The implication

of only being able to detect a significant decline over such a long period is that the monitoring protocols need to be improved. Organizations collecting abundance data over a long time period often face intermittent funding difficulties, leading to unequal intervals in the time series and these 'missing' observations and irregular sampling can compromise the detection of trends. However Humbert et al. (2009) argue that the extent to which these gaps can compromise results is not well-known and requires further investigation

The results of using maximum and mean counts are similar for all species. Mean values average out any variation in counts over the replicate surveys, but may perhaps underestimate the true abundance as a result. Link et al. (1994) warn that average counts are a biased estimate of actual numbers of the species present and that no amount of replicate or new sites will eliminate this bias. The lower counts might be as a result of an unseen factor that affected the number of birds calling. Using the maximum counts is arguably a truer representation of the number of birds in a site, simply because that number were heard on at least one replicate. This does not however take into account the fact that birds may move between sites, although care was taken not to double count individual birds as described in the methods. Maximum counts would certainly be more appropriate if it was known that the species in question were territorial and so a maximum count would represent more accurately the number of calling birds within the area, as those individuals were unlikely to be counted elsewhere. However using the mean value can be considered useful in reducing the effect of variation from observer error (Diefenbach, 2003).

5.1.3 Inferences from vegetation mapping.

Several authors have used ordination techniques to determine the vegetation composition of landscapes across the world (Beals, 2006, He et al. 2006, Cooper et al., 2010). Techniques such as two-way indicator species analysis (TWINSPAN) and Detrended Correspondance Analysis (DCA) help to identify groups or communities of species that can be considered as different, based on species information for individual plots. The differences between the ordination techniques available allow for slight variation in the statistical power but are generally similar in their capabilities (Zhang, 2010). Rolecek et al. (2009) have suggested improvements to TWINSPAN that account for heterogeneity between plots.

By collecting data on species abundance for a series of plots, as well as other variables such as soil nutrient content, geographical data and other environmental conditions that may affect the distribution of certain plant species, the relationships between these factors and the distribution of vegetation can be better understood (Cooper et al., 2010). This sort of information can then be used to inform better management plans for zoning protection and development in different areas of the habitat (He et al., 2006). With plans to open up the Pipar Pheasant Sanctuary to more tourism, (Kaul & Shakya, 2001) this type of information would be extremely valuable in determining which areas to use to minimise potentially negative impact on habitat requirements of different Galliforme species. It would also provide a means of monitoring the effects of tourism development on the Pipar environment and its wildlife, allowing managers to adjust their strategies accordingly

Although detailed information was collected by Picozzi (1983) and Poudyal (2005) on species present in plots across the Bowl, unfortunately it was compiled into a set of data that does not contain the species per individual plot, but rather the number of times species were encountered through the surveys. This unfortunately does not allow for these types of ordination analysis and therefore establishing distinctive communities of plant species for further analysis of relationships with environmental factors was not possible. If different plant communities could be identified through these techniques, then by comparing the call counts from each listening station, reasons for particular heterogeneity between plots might be explained by vegetation composition. As the Pipar Bowl is approximately 42km^2 and all on an east facing slope of the valley, there may not be enough variation in vegetation composition for comparison other than the fact that altitude may play a role in determining plant species composition and spread of Galliformes across the terrain.

Multidimensional Scaling (MDS) techniques are a set of related ordination statistical methods that are often used for visualising information and facilitating exploration of any similarities present within a complicated set of data. MDS analysis is often used for representing a community structure which is then related to environmental factors (Clarke & Ainsworth, 1993, Guo, 2010).

With the effects of a changing climate being such an important current issue due to its potential impact on environmental change, several authors have discussed the uses of ordination techniques in describing effects on vegetation composition and highlighting the important variables that should be collected in order to allow for such inferences to be made (He et al., 2007, Baselga & Araujo, 2009). Information on abiotic factors such as geography, soil composition and environmental variables could be very valuable for managers of the Pipar Reserve to predict effects from a changing climate, and the effects a change in vegetation composition could have on Galliforme numbers.

Another problem that is evident in endeavouring to find relationships between the vegetation data and Galliforme abundance, relates to the timing of the vegetation surveys. Both Picozzi (1983) and Poudyal (2005) reported that data collection took place in the month of November and, for Poudyal and his team, not long after the first snowfalls of the year, making plant identification rather difficult as many trees and shrubs were dry. They proposed carrying out the survey at a time when most plants are flowering such as during the Spring months, so that more accurate data on species can be collected. As the Galliforme surveys are also undertaken in the Spring this could pose a challenge if there are limited human resources available, but there could also be financial economies, from running both surveys successively.

With a clear distinction between plant communities described, comparative analysis would be more attainable. The spread and diversity of species in the reserve could be more accurately mapped and any changes between years clarified with more comparative statistics such as t-tests based on Shannon indices. If the raw data on species specific to individual plots had been retained, then comparative t-tests based on Shannon Indices would provide more informative results about vegetation composition and change in Pipar. Changes in percentage cover does not give a very clear indication of changes in vegetation composition that can be directly related to Galliformes. Habitat suitability would be very useful information but it requires more than just cover, particularly as one of the categories is 'plant cover' as a whole. More informative data on species (and suitable habitat) could suggest reasons for distribution of Galliformes as well as providing more statistical power for analysis of change.

5.2 How many? How often? Improving accuracy and value of data collection for Galliforme abundance and vegetation surveys

Rosenstock (2002) reported that although index counts (such as these in Pipar) do not account for variability in detection, 95% of the bird studies that they considered between 1989-1998 relied on point counts such as this. While models that incorporate detectability are not as widely considered for this type of study, Diefenbach et al. (2003) also agree that they need more attention for application to land bird studies. Farnsworth et al. (2002) support the view that using survey methods that correct for uncertain detection is recommended and should not be ignored as is often the case.

Farnsworth et al. (2002) established that time of day and observer differences also affected detectability. Kaul & Shakya (2001) describe how, as well as dawn call counts, dusk call counts were collected in Pipar, but due to inconsistent calling from the birds at this time, these were not amenable to counts and so were abandoned. As all counts therefore are conducted at dawn, this reduces potential for variability in the results. Observer heterogeneity however could have an effect on detection probability. Farnsworth et al. (2002) suggest that rotating the observers between sites would take account of this variability, and it is suggested that this approach is taken in Pipar to minimize variability. More support for this approach is given by Cunningham et al. (1999) implying that variance of counts can be mitigated by averaging counts from two observers. Double observer approaches require two observers recording data from the same site, and while they can reduce variability from observer heterogeneity (Forcey et al., 2006) the extra cost involved may not justify this approach for Pipar. Further investigation into these methods is required.

Considering changing the sampling method, one of the most important considerations for reducing error in the model is to increase the sample size. By re-sampling the data collected in Pipar and generating call counts for 10 listening stations, the power to detect a trend was increased. This can provide managers of the reserve with a more accurate understanding of what is happening to abundance of galliformes and therefore enable a more timely conservation response. In order to detect how rapidly the GLM analysis for Pipar would pick up the trend with increasing sample size, the final years data was sequentially excluded year by year until the trend was no longer detected. For the Koklass data this occurred when running GLM excluding the data from years 2009, 2008 and 2005. This might suggest that the current monitoring protocols require up to 25 years to detect a decline of about 30% in numbers. If increasing sample size led to a trend being detected over fewer years of surveys, this would be of greater value to conservation managers, who could adapt their strategies before the trend became too firmly established. Thogmartin et al. (2007) investigated bird point counts, and suggest a minimum sample size (number of sites) of 30 in order to detect a decline of magnitude 5% per year in a short time series (3-5 years). For small areas of land (such as parks), they also suggest that managers include sites in surrounding areas in order to attain a large enough sample size. Bart et al. (2004) recommend surveying two thirds of the target area to detect significant trends in abundance. Managers of Pipar have started to monitor surrounding areas (Poudyal, 2009) and this would be recommended as an improvement to the current protocols.

In summary, the power of long-term data collection at Pipar could be improved by minor adjustments to current protocols focused on increasing the sample size, adjusting the collection and analysis of replicate data to account for differences in detectability between sites, and increasing the detail collected on vegetation.

On **increasing sample size,** Poudyal (2009) mentions that several new sample sites created in the area surrounding the Pipar reserve would allow comparison of Galliforme abundance in different parts of the Annapurna Conservation Area. Also increased sample size within Pipar by including a larger area in the survey would not only increase the power of the analysis but also identify suitable habitat for Galliformes and provide insights into how they might be using it. Continuing the monitoring protocol with

replicate surveys, the potential impact of pseudoreplication issues could be reduced by and either using summarised counts (eg maximum) to run GLM or by incorporating a model that accounts for detectability - within sites, between sites and due to observer heterogeneity. Running Mixed Effect Models is an alternative approach.
Increasing the detail of vegetation collected – i.e. retaining data on all species within each plot – will allow for better understanding of the composition of vegetation in the bowl through ordination, and could provide better information on change. Having access to data at the species level that is correlated with the geographic variables (aspect, slope, altitude) could be crucial in order to monitor the impact of changed pressures on the habitat such as a changing climate and increased human use. Improvements could include the collection of soil samples and a focus on vegetation mapping around calling stations to derive inferences about any changes in Galliforme numbers related to shifting vegetation patterns.

Continuing with current collection protocols with these minor improvements has the added advantage of cost-effectiveness – observers are available, trained, experienced in this terrain and passing on their skills. Some additional costs for increasing sample size and adding elements such as soil collection/analysis would be expected.

5.3 Implications for conservation management.

Conservation management requires trade-offs between action for different species and populations, between the states of different ecosystems, between preservation or transformation, between the needs of different people and those of other species (Leader-Williams et al., 2010). Conservation management also occurs at different levels, each involving choices and each requiring evidence.

At the **international level**, several authors (e.g Collen, 2009) suggest that indicators of species population trends are some of the most sensitive and useful measures of change in biodiversity status. When combined with other indicators (such as extinction risk, habitat extent and condition, and community composition), a compelling picture of continuing biodiversity decline is demonstrated (Butchart, 2010) which should provide an alert to the international community to step up conservation action. The Living Planet Index (Collen, 2009) aggregates trends in the abundance of species for which data is available and has been adopted by the Convention on Biological Diversity to measure progress (or lack of it) towards the 2010 target (a reduction in the rate of biodiversity loss by 2010).

While investment in data on species population trends can help rally and inform action between nations, at the **local level however,** managers are seeking a clear understanding of observed changes, how they are significant and what is driving them. Responses to natural or human-induced changes might indicate different conservation responses. At Pipar, managers are having to assess whether specific conservation strategies are required for the Galliformes, and whether they have a baseline against which to assess the impact of increasing tourism or restricting use of the habitat by local communities (Poudyal, 2009).

5.3.1 Drivers of decline in Galliformes at Pipar.

Both natural and human-induced changes could be driving a decline of Galliformes at Pipar. A changing climate could have two levels of impact – directly on vegetation composition with altitude and across micro-climates, indirectly through consequent increase of human pressures. A finer lens on vegetation monitoring protocols could inform the former; separate effort into monitoring of human use in and around Pipar is necessary for the latter. Two aspects of human-use may be of concern to managers – the influence of non-timber forest product (NTFP) collection and the potential to support the reserve financially through increased tourism

Collection of Yarsaguma (caterpillar fungus) at Pipar is a seasonal activity that draws people into the reserve (Subedi, 2010) and the high value of this NTFP, together with the need to maintain the natural habitat conditions that nurture it, make this a useful mechanism to derive community benefit from, and support for, Pipar. However, if opportunistic taking of pheasants by collectors is occurring, the collection activity will require increased investment in monitoring.

Eco-tourism is a mechanism used widely to support protected areas financially but requires close monitoring to ensure that it is "Eco", i.e. does not damage the resource on which it depend*).* It is also a fragile market generally, although the specialist birder market is more persistent and reliable and might be the one for Pipar to target for greatest return. As it has not been a major mechanism for a remote area such as Pipar, the possible decline in Galliformes cannot be attributed to tourism; a careful increase in tourism could fund both increased monitoring and greater control measures for Pipar.

5.3.2 Trade- offs between statistical power and investment of resources

Balmford et al. (2003) derived high-level aggregate figures of funding available to managers of protected areas globally and estimated that they were merely 20% of what was needed globally. At the local scale of the Pipar protected area, conservation managers seeking to harness scarce resources might look to potentially more cost-effective ways of monitoring populations of key species, such as camera traps or satellite data-based indices for monitoring land use and habitat, despite their limitations.

There is clearly a trade-off between increasing statistical significance by mechanistic and complex models that endeavour to conquer the complexities of data from natural systems, and the real world constraints under which field biologists operate. Much research on avian biology is undertaken without reference to these complex models (Beissinger et al., 2006). Arguably the best conservation decisions and actions will arise from cooperation between all parties in assessment of ecological process - ,the field biologists, statisticians and managers (Beissinger et al., 2006) - supporting the development of these models and by testing and validating them.

5.3.3 Broader implications of long-term studies for policy and practice

Leader-Williams et al. (2010) point out that choices in conservation are often hidden; they describe how philosophical and cultural factors influence choices as well as a scientific evidence base, but stress that credible measuring of impact is key, whether it is measuring success or failure against the management intention. Analysis of the long-term data on Galliformes at Pipar indicates that up to a 25 year time-span is necessary to demonstrate a significant trend of approximately 30% decline in one species of pheasant. While this will vary between habitats and species, and may be reduced by changes to monitoring protocols, this finding puts even greater importance on long-term data sets in trying to understand population trends.

The value of the long-term data from Pipar also goes beyond the limits of its statistical significance in demonstrating trends. The involvement of conservation scientists and managers from the area in design and implementation of the surveys over many years generates and perpetuates a knowledgeable and enthused support base for the area and its biodiversity within the region. The broader discussion around interpretation of results set against a changing backdrop of pressures on Pipar and options to allay them informs conservation management decision-making. This work has demonstrated however, that some minor changes to survey protocols would strengthen the inferences that can be drawn from this long-term data set. They would decrease the time period within which significant changes in Galliforme populations can be reliably demonstrated, and increase the useful detail that could be derived. A finer lens could be focused on the response of different species to the impact of habitat-change induced by human use or a changing pattern of climate. Collection of this more detailed and robust data would require an increased investment in time and resources, but possibly not by a substantive order of magnitude. The return would be in more defensible data to inform management choices – providing a baseline for the impact of increased tourism, or a map of areas where protection should be intensified.

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