Validity of Biodiversity Monitoring Programmes: Boundary Stream Mainland Island Project, Department of Conservation.

Brendon Rex Christensen

2003

A thesis submitted to Canterbury University in part fulfilment of the requirements of the degree of Master of Forestry Science.

University of Canterbury

Sunrise over Boundary Stream Scenic Reserve – September 2001

Abstract

The recent move to *in situ* conservation management world-wide is supported by, and stems from the 1992 International Convention on Biological Diversity. The Department of Conservation – charged with the conservation of New Zealand's natural resources – has directed efforts towards the restoration of natural processes as an avenue to halt local biodiversity decline. Ecosystem, habitat, and nature restoration programmes such as the Boundary Stream Mainland Island Project (BSMIP) represent the forefront of conservation management, combining intensive multi-species pest control, with broad-scale hierarchical monitoring programmes.

Monitoring programmes confer information that is intended to support decision-making and management by the reduction of uncertainty, or by increasing knowledge. The validity of monitoring programmes depends on three key parts; the guiding objectives, biological relevance, and statistical reliability. Seven major long-term monitoring programmes established at the BSMIP were evaluated according to the above criteria. All monitoring programmes had appropriate guiding objectives, and were biologically relevant (outcome and result monitoring were balanced respective to each other and to the restoration intervention and efforts at BSMIP). The statistical reliability of the programmes was appraised with the use of the Computer programme MONITOR, which provided a calculated value for the statistical power of the monitoring programmes. All monitoring programmes except two (Lizard monitoring: which was initially designed as a short-term species survey, and Mustelid monitoring: which would be a good candidate for a double sampling methodology) had a robust design (evaluated using the actual initial data, and conservative criteria for the detection of population change). The monitoring programmes that did achieve a level of statistical robustness, provided a statistical power of ≥0.8 (≥80%) within appropriate timeframes for restoration of ecosystem processes (e.g. the timeframe for detection of a 10% change in the abundance, density, relative index, *etc* of the Result monitoring programmes: Rodents = three years, Possums = six years, and Outcome monitoring programmes: Wētā = five years, Ground Invertebrates = four years, Birds (species nos.) = four years, Vegetation (Species, and sapling nos.) = 15 years).

The guiding objectives for monitoring programmes must have clear, specific, measurable, and achievable goals, in-order to identify appropriate variables, in both spatial and temporal scales. The biological relevance or "linkage" between monitored groups is important and must be at least outlined, for monitoring programmes to be able to identify potential cause and effect. Statistical reliability (the balance between statistical significance, statistical power, and the timeframe for a conclusive result to be determined) is important, as it is the key method of detecting change. Statistical power can improve the design and efficiency of monitoring programmes and clarify research results. Power analysis has become readily available for researchers and managers with the development of computer programmes specifically designed for this task.

Table of Contents:

List of Figures:

Page

List of Tables:

Acknowledgements:

David Norton initiated this work, and provided enthusiasm, and direction for myself as a student, as well as a conservation manager. Jenny Brown gave positive and ongoing encouragement, and with ease dispelled the myths about statistical science being difficult and arcane. Both showed a large amount of patience, especially as this thesis always seemed to be "a couple of months away", I thank you.

The Department of Conservation provided much valued support for myself to complete my Masters. Geoff Walls, who with a bit of expert conjuring outlined and initiated an excellent monitoring programme for Boundary Stream, and also instilled an awareness of the "essentialness" for conservation monitoring in me. Geoff Walls, John Adams, and Ken Hunt whose impetus started Boundary Stream will hopefully be rewarded in knowing that they have initiated an excellent project, with benefits too numerous to count, not only for conservation, but also for the people involved. I know that I am richer within myself for being involved in the BSMIP. Steve Cranwell, and John Cheyne recognized a need both for the project, and myself to pursue this investigation, I thank you for your foresight, receptiveness, and insight.

I thank Murray Heays, Andrew Herries, The Druries, and The Maxwells for their key support of the BSMIP project as good neighbours. I thank Lisa Ward for help with the data input, Simon Smale for the help with the *Te Reo* Maori translation, and Dave Hunt for a comprehensive and useful review of this thesis.

The gentleman scholar of Ahuriri, Hans Rook whose understanding of conservation and actual effort of "bullets and staples" has conserved and will retain New Zealand's natural heritage, has given me insight and true career training. To the past, present, and future workers at Boundary Stream, this work is yours as much as it is mine, I only trust that I have done you service.

The local Hapu Ngati Tue, Ngati Kuru Mokihi, and Ngati Whakairi, H. Guthrie-Smith, Margaret Heays, and the Shine family who each in their own part helped conserve the native life in the area now called Boundary Stream. And finally the Maungaharuru range forests,

"***Kia haruru ano nga koiora rereketanga a Maungaharuru***"

So that all the different living things on Maungaharuru will roar with life again

Chapter 1 Introduction

1.1 *In – Situ* Conservation

The recent move to *in situ* conservation within conservation management (Norton 1993), as a concerted national effort stems from 1992, when the New Zealand Government became a signatory to the Convention of Biological Diversity at the "Earth Summit" in Rio de Janeiro, Brazil (Anon 1994). This convention recognized global concern at the loss of biological diversity "Biodiversity" and established a framework by which contracting parties should set about achieving the stated conservation objectives. This lead to the preparation of the New Zealand Biodiversity Strategy (Anon. 2. 2000), with its first theme the conservation of indigenous biodiversity on land. The Department of Conservation has selected the "mainland island" management approach as one of a number of avenues to stem local biodiversity decline (Anon. 2. 2000).

Mainland Island Projects are designed to achieve this goal of conservation *"In – Situ"* (in its place), for the local indigenous ecosystem. The intensity and scope of pest species management performed at Mainland Island Projects represents a large increase in effort, along with the monitoring of results and conservation outcomes (Saunders & Norton 2001). It also seeks an ideal of ecological restoration by re-establishing a completely functioning ecosystem (Stanturf *et al.* 2001). This *in-situ* approach provides an opportunity to directly apply the *scientific method* (experimental manipulation) of testing hypotheses on large and complex ecosystems (Likens 1985). Mainland Island Projects, and similar intensive restoration efforts advance Lawton's (1997) requirement that conservation action be based on, and use science. The Boundary Stream Mainland Island Project approaches what van Diggelen *et al.* (2001) terms as "true", restoration. It is the most ambitious level of restoration, a complete return to an unaltered ecosystem state. This desired condition while admirable, cannot be achieved in ecosystem restoration (Cairns 1991; Hobbs & Norton 1996) because of the lack of inerrancy, it is virtually impossible to remove the imprint of over 1000 years of human intervention and associated pest species impact on New Zealand's ecosystems.

Saunders (2000) has identified the six current Mainland Island projects as having three primary attributes;

- *They are adjacent to other areas not managed for restoration purposes,*
- *They are focused on ecosystems rather than species alone,*
- *They have restoration rather than protection goals.*

The initiation of mainland island projects represents a move away from the single species management of conservation towards a broader ecosystem management focus. It is the current culmination of the advance in conservation management, building from a (written) history of improvements and initiatives since the efforts of Richard Henry (c 1890s) who transferred kakapo and kiwi to the then predator-free Resolution Island (Saunders 2000).

Mainland Island Projects, and "mainland habitat islands" have incorporated longer-term approach to conservation management as suggested by Norton (1993). The Boundary Stream Mainland Island Project is driven from the long-term vision established in 1995;

- *Boundary Stream Scenic Reserve will be restored, by careful nurturing and enhancement, to the vibrant indigenous ecosystem it once was.*
- *The reserve will be a place where the public can visit and enjoy a flourishing fauna and flora reminiscent of a typical Hawkes Bay forest of the past.*
- *It will be a showpiece for the conservancy, providing a centre for community involvement and demonstrating what can be achieved in protecting and enhancing biodiversity given sufficient resources, enthusiasm, commitment and public support.*

(Adams 1995)

The "Mainland Island" concept was launched nationally during 1995, with the Boundary Stream Mainland Island Project initiated in the same year, with intensive management of pest species beginning May 1996. Over fifteen independent long-term monitoring programmes have been initiated within the project to date. These programmes were established to assess the ecological response to the management of pest species within the Boundary Stream Scenic Reserve (Treatment site), and the comparative Non-Treatment sites of Thomas Bush and Cashes Bush Scenic Reserves (Christensen 2000). It is these monitoring programmes that have been evaluated in terms of their validity for this study.

The measurement of the conservation efforts are as important to the core Boundary Stream Mainland Island Project management objectives as are the actual efforts themselves. This is identified in the management objectives below;

5.1 Ecosystem Recovery: The recovery of forest structure and ecosystem processes by the control to low levels and where possible the elimination of animal and plant pests and the exclusion of domestic farm animals.

5.2 Monitoring: Monitoring of key environmental factors to establish baseline information and measure changes resulting from enhancement activities.

5.3 Threatened Species Recovery: The recovery of resident threatened species e.g. kakabeak, yellow-flowered mistletoe, kiwi, kereru, etc. Determination of the status of threatened species known to be present, and the detection of other species that may be present.

5.4 (Re)Introductions: The (re)introduction of species formerly present or at risk in the region e.g. NI saddleback, blue duck, stichbird, kokako, Pittosporum obcordatum, Dactylanthus taylorii, Powelliphanta traversii 'Maungaharuru', etc.

(Adams 1995)

1.2 Monitoring of Conservation Management

The very first "key step forward" of the Department of Conservation's Strategic Business Plan 1998-2002 "Restoring the Dawn Chorus" states the necessity for;

"1. Better information

Putting in place better programmes for monitoring and reporting on the ecosystems, species, sites, and facilities we manage and measuring our effectiveness."

(Anon 1999)

This directive and its successor under the 'new' first key step of '*Expanding the biodiversity effort'* of the 'Restoring the Dawn Chorus 2002-2005' (Anon 2002) is pivotal for the Boundary Stream Mainland Island Project. It guides managers to ensure that the monitoring programmes adequately determine the Boundary Stream Mainland Island Project's effectiveness as an ecosystem restoration project. As yet there is little guidance for the design of monitoring programmes, and those that are set up are undertaken with little inquiry into how valid they are as effective monitoring programmes (Thomas 2001). Currently there exists an information "gap" in the management of the New Zealand environment (Anon 1997). This gap is centred around "*…the absence of sound information about the state of our natural heritage assets…*" (*ibid*.). Inventory and monitoring are key components in addressing this deficiency.

1.3 The Use of Statistical Power in Monitoring

Just as there exists a need for robust ecological foundations, such as clear goals and good design on which to develop and implement restoration programmes (Diamond 1987; Clewell & Rieger 1997), the methods of monitoring the ecological response to the restoration efforts also need to incorporate an appropriate level of statistical "robustness" (Norton 1996; Norton 2000). This can be approached by detailed consideration of the sampling framework "monitoring design" (Eberhardt & Thomas 1991). An effective method of assessing the level of "robustness" of the monitoring programme design is to incorporate statistical power analysis. Fairweather (1991) has stated that statistical power analysis is a relatively recent inclusion to actual research. Previously researchers (and natural resource managers) have given cursory consideration to the concept of statistical power (Kraemer & Thiemann 1987), and it is often an unfamiliar concept to them (Gerrodette 1991; Thomas & Krebs 1997), and decision-makers (Peterman 1990b).

Statistical power analysis can improve the design and efficiency of monitoring programmes (Green 1989; Faith *et al.* 1991; Fairweather 1991; Greenwood 1993; Brown & Miller 1996; Steidl *et al*. 1997; Anderson-Cook 1998), and clarify research results (Peterman 1990b; Reed & Blaustein 1995; Andrén 1996). It can be also be used to determine whether the intrinsic variance is appropriate for the population of interest (Lougheed *et al.* 1999). It has been recommended by a number of researchers McBride *et al.* (1993), Taylor and Gerrodette (1993), Osenberg *et al.* (1994), Thomas & Juanes (1996), Steidl *et al.* (1997), and Brown & Miller (1998). Statistical power analysis can reaffirm the necessity of precautionary management, such as in the conservation of coastal cetaceans (Thompson *et al.* 2000). The need and reasons for performing statistical power analysis has been outlined by Green (1994), stressing the importance of reducing error and standard deviation as a way of increasing power to detect change or signal. Statistical power analysis can thus provide benefits for overall conservation management and should be used to direct the process of conservation monitoring. The benefits of using statistical power analysis include;

1. The determination of a programme or design's ability (sensitivity) to detect a change.
2. The efficient design and planning of studies: *a priori* statistical power analysis.
3. The evaluation of any non-insignificant results.

While the risks of not using statistical power analysis include;

1. Making false inferences due to non-significant results from weak tests (Hayes 1987; Peterman 1989; Forbes 1990; Fairweather 1991; Ottenbacher 1996; Brown & Miller 1996).
2. The potential waste of time, effort, and resources on a programme or design that may not yield useful information (Gerrodette 1987; Brown & Miller 1996; Cherry 1998).
3. It is unknown what impacts or perturbations are actually occurring, regardless of whether their effects are beneficial or harmful (Bernstein & Zalinski 1983; Fairweather 1991; Peterman & McGonigle 1992) unless they are very large (Carpenter 1990; Dixon & Garrett 1993).
4. A complacency in thinking that Type I errors cost more than Type II errors, with statistical power being ignored in favour of statistical significance (Toft & Shea 1983; Peterman 1990b; Fairweather 1991).

The incorporation of statistical power analysis into conservation, and environmental monitoring programmes has occurred to far lesser degree than its addition into research studies. Statistical power analysis has been used by a number of authors (see Appendix 1) for monitoring programmes, detailing the likelihood of detecting a change in trends, and do not necessarily express that the use of statistical power analysis is the norm.

Statistical power analysis has also been applied by the United States Geographic Service (USGS) to determine sample sizes for the North American Amphibian Monitoring Program (sic), and to evaluate mushroom population monitoring using the computer programme MONITOR (Gibbs 1995). In New Zealand, statistical power analysis has been used by Brown & Miller (1998) for the design of stoat (*Mustela erminea*) monitoring programmes, Pledger (1998) in the monitoring of Hamilton's Frog (*Leiopelma hamiltoni*) populations, briefly outlined by Ogle (1999) for the determination of sample size for foliar browse scoring, and by O'Donnell & Langton (2003) in the design of long-tailed bat (*Chalinolobus tuberculatus*) monitoring programmes (see Appendix 1).

1.4 Aim of this Thesis

This thesis is a review of the justifications for, and concepts used in established conservation monitoring programmes. It seeks to build on the current use of *a priori* statistical power analysis in conservation monitoring within New Zealand, by using the Boundary Stream Mainland Island Project's monitoring programmes as a case study. The broad question that underlies this Masters thesis is;

> *'Is the monitoring performed at the Boundary Stream Mainland Island Project going to show anything*?'

The statistical reliability of the Boundary Stream Mainland Island Project monitoring programmes is evaluated (through a preliminary *a priori* "prospective power analysis") in terms of the ability to detect an actual change, the determination of timeframes required to observe a conclusive change, and a determination of effective monitoring designs. It offers recommendations such as; improvement of designs, different designs and uses for monitoring programmes that are currently not as robust as they could be, future directions for conservation management, the importance of relating goals, and objectives of projects to the monitoring designs and statistical analyses. This thesis will in some part, advance the methods and understanding in evaluating the ecological management success of the Boundary Stream Mainland Island Project, and infer important and valuable considerations in the evaluation of restoration projects generally. General guiding principles for monitoring within the Department of Conservation will be recommended, as well as considerations of statistical validity. It is considered that these would provide guidance in the design and planning of future monitoring programmes.

Chapter 2 Boundary Stream – The Place & Project

2.1 Introduction to Boundary Stream Scenic Reserve – The Place

The Boundary Stream Scenic Reserve (BSSR) contains the majority of the Boundary Stream Mainland Island Project Treatment Site. The reserve is situated on the south-eastern flanks of the northern end of the Maungaharuru range, approximately 60kms north-west of Napier (Figure 2.1.1).

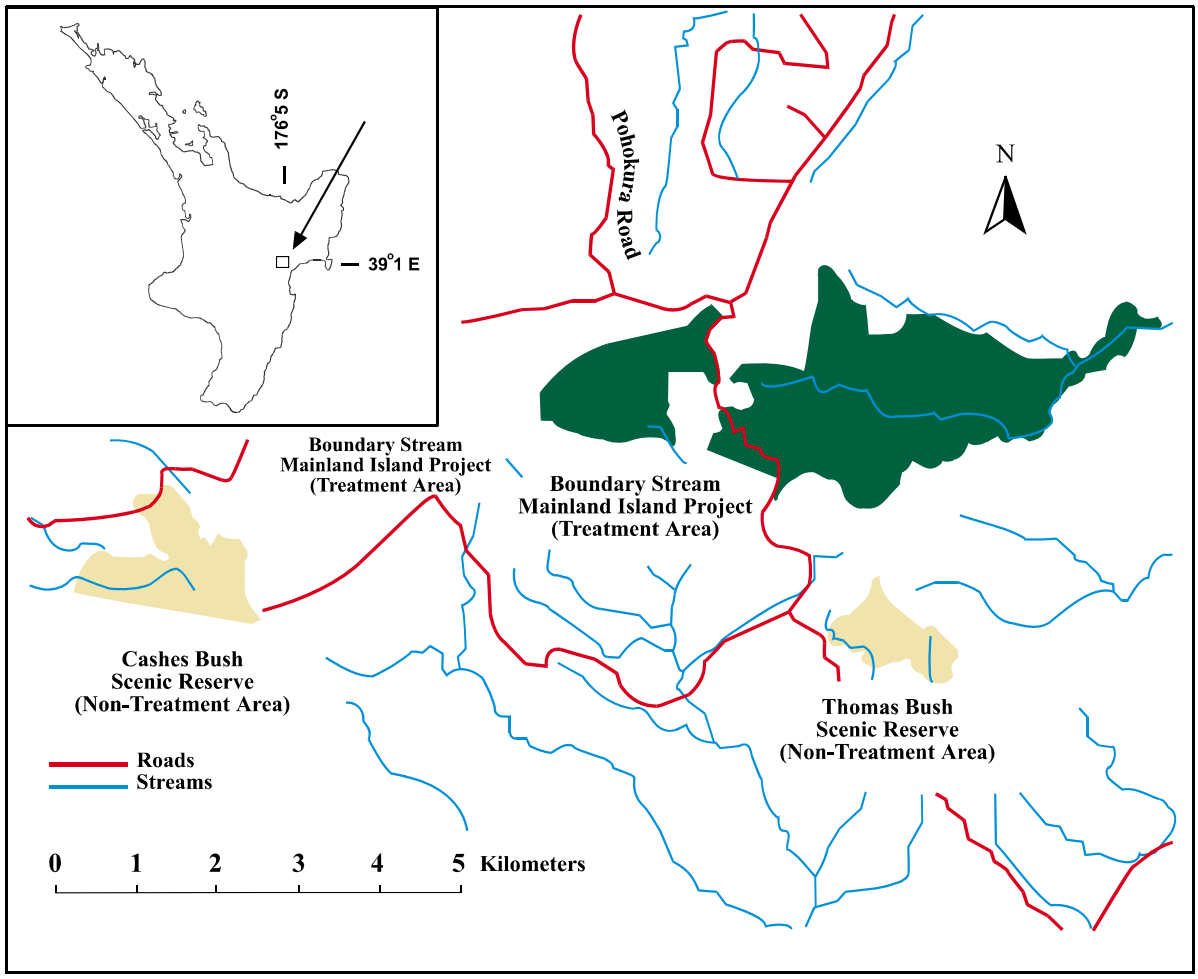

Figure 2.1.1 Location Diagram of Boundary Stream Mainland Island Project

The contiguous forest including the reserve is approx. 800 ha in size (including 100ha of private land), and extends from 300m a.s.l. to 1000m a.s.l. While previously cleared in parts, the reserve remains the largest and most intact forest left in the Maungaharuru Ecological District (Adams 1995). The reserve is the home for a relatively large range of native species, including wood pigeon, New Zealand falcon, whitehead, and the occasional north island kaka. In 1998 north island robin were the first of the planned introduced species to be brought back into the reserve. North island brown kiwi were first re-introduced during 2000, and north island kokako re-introduced during 2001. Boundary Stream Mainland Island Project has been described as a "Living Laboratory", and its accessibility to the public provide opportunities for education, research, and involvement in restoration programmes (Anon.1. 2000).

2.2 Landform and Geology

The Maungaharuru range was formed as a result of pressure from the Pacific Continental Plate against the Australian Continental Plate (Cutten 1994; Townsend 1996). Walls (1989) identifies the Maungaharuru range as a unique site among mountain ranges;

'*Like an outlier of the inland mountains, different with its own special character'*.

The Maungaharuru range is a tilted-block range, formed of late pliocene and pleistocene (up to five million years old) sedimentary material, that has been uplifted, tilted and broken (McEwen 1987; Cutten 1994; Townsend 1996; Graafhuis 2001). This has created a series of deeply incised limestone, sandstone, and siltstone gullies, one of which contains the Boundary Stream Scenic Reserve (Figure 2.2.1). The reserve has a complex topography of steep escarpments, limestone outcrops, and numerous streams – forming several waterfalls (Adams 1995; Christensen 2000).

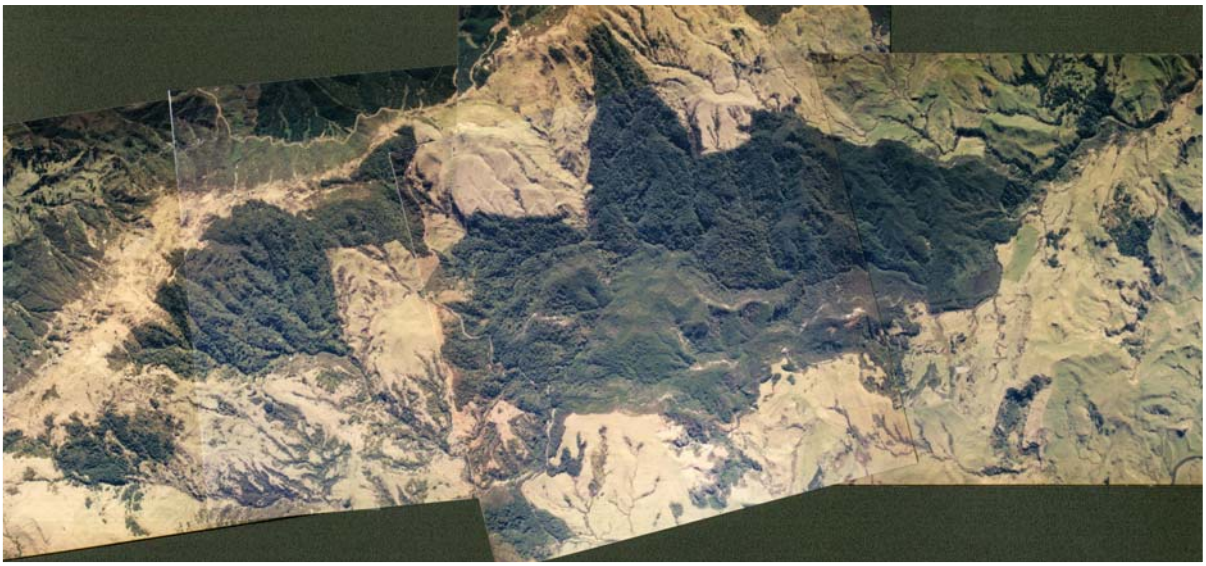

Figure 2.2.1 Composite Aerial Photograph of Boundary Stream Scenic Reserve

c 1995, Scale 1: 50 000

2.3 Vegetation

This complex topography, and large altitudinal range provides a mosaic of habitats with a range of vegetation types, from lowland scrub and broadleaf forest to montane forest and grasslands (Adams 1995; Christensen 1999). Walls (1989) performed the first evaluation of the conservation values of the Maungaharuru range, noting the range's special mountain holly (*Olearia iliciflolia*) and broadleaf forests, with no other such forests or woodlands found within Hawkes Bay. The nearest ecological equivalents of such forests are the leatherwood (*Olearia colensoi*) scrub and pink pine forests (*Halocarpus biformis*) of the southern Ureweras, and the akeake (*Dodonaea viscosa*) forests of the Chatham Island (Walls 1989). The range also holds red tussock (*Chionochloa rubra*) on the south-western end of the range, which is rare for the area, several small wetlands with a small sedge *Carex enysii*, which is known nowhere else in the North Island, and a native daphne *Pimela* spp. which is confined to the Maungaharuru range (Walls 1989). The range also holds a few mountain daisies (*Celmisia* spp.), and speargrasses (*Aciphylla* spp.) as Walls (1989) notes as;

> "*the most tangible evidence of the alpine nature of the crest of the range*".

Grant (1996) notes in his excellent and highly informative book: *Hawke's Bay forests of yesterday*, that the Boundary Stream Scenic Reserve contains some very old matai (*Prumnopitys taxifolia*) and rimu (*Dacrydium cupressinum*), remnant of the older forests once present on Maungaharuru. The Boundary Stream Scenic Reserve contains a matai with the largest diameter (225cm *c* 1996) of those in the Hawkes Bay region (Grant 1996), a veritable *Kaitiaki* (guardian) of the forest – being over several hundreds of years old.

Geoff Walls produced the first comprehensive vegetation map of the Boundary Stream Scenic Reserve. He distinguished at least 12 distinct vegetation types within the reserve. Of the 220+ species of indigenous vascular plants recorded by A. P. Druce (1985-1988), with additions by G. Walls (1995-1997) include the threatened plant species such as kowhai ngutu-kaka kakabeak (*Clianthus puniceus*) at its southern-most limit in the wild, *Pimelea* "Maungaharuru" (undescribed, aff. *P. aridula*) and yellow-flowered mistletoe (*Alepis flavida*). G. Walls (1997) noted other plants of botanical interest including; neinei (*Dracophyllum latifolium*), tawari (*Ixerba brexioides*), and tawheowheo (*Quintinia serrata*)

all at their southern-most limit in the wild, and silver beech far removed from its key domain in the axial ranges.

2.4 Human History

Maungaharuru was named the "rumbling mountain", according to local legend of the noise made by the prolific bird life that was present during Maori settlement of the area (Anon 1994; Christensen 2000). The area that now comprises of the Boundary Stream Scenic Reserve was once used seasonally as a communal area for local hapu, harvesting the abundant natural resources of bird and plant life. Hawkes Bay forests, like that on Maungaharuru were mainly affected by natural events up to european settlement (Grant 1996). These natural events, including snowstorms can still cause substantial damage to the forest on the range. Pastoral farming started in the Tutira area in the 1860s (Guthrie-Smith 1953), with large areas of forest being systematically cleared towards the range, though Boundary Stream was not affected till the 1930s (Anon. 1. 2000). This extensive clearing of the native forest carried on well into the 1970s, when large scale exotic forestry began in earnest (Anon 1994). The reserve, once part of a large vibrant forest is surrounded now by pastoral farming, a forest "island" on the mainland.

2.5 Introduction to Conservation Management Treatments – The Project

Principal threats to New Zealand's indigenous habitats on the land administered by the Department of Conservation are fire, animal pests, and plant pests (Anon 1994). The Department of Conservation is directed by legislation to control these threats, with the aim to prevent the deterioration of natural resources. The Department of Conservation uses a range of methods as well as differing magnitudes of intensity in its threat management (treatments). Within the Boundary Stream Mainland Island Project an intense broad-scale pest management is carried out leading towards the ecosystem management perspective as described by Ehrenfeld & Toth (1997) for the Treatment site. This indigenous nature and ecosystem restoration project has been coined;

"*Restoring the Past, Revitalizing the Future*",

by Chris Ward (Conservancy Advisory Scientist: East Coast/Hawkes Bay Conservancy *c* 1997).

2.6 History of Pest Management

Prior to the establishment of the Boundary Stream Mainland Island Project, the Boundary Stream Scenic Reserve was subject to the same general conservation management treatment as the rest of the land administered by DoC. Prior to 1987, pest management was the responsibility of the Hawkes Bay Pest Destruction Board. The board did little if any control work within the forests on the Maungaharuru range, though worked on the surrounding farmland on a yearly basis targeting rabbits, hares, and possums (Christensen 1996).

During the 1990s a more comprehensive pest management strategy was initiated, targeting the key conservation pests of goats and possums. Goat control comprised of annual aerial and ground shooting along the greater Maungaharuru range, targeting large mobs of goats. Possums were first managed in the area using aerial 1080 poison bait applications in 1992, and later by contract trapping. The pre-1080 application trap-catch figure of 37.2 possums per 100 trap nights is relatively high, and demonstrates the limited and sporadic focus of previous control work, where only farmland was treated (Christensen 1996).

2.7 Intensive Multi-Pest Species Control – Treatment Site

Mainland Island Projects are characterized by "Intensive multi-pest species control" (Saunders 2000). This control or more correctly management of the pests that constitute a risk or threat to indigenous biota (an ecosystem) at a particular site, is performed at a scope and intensity never attempted before on the New Zealand "Mainland". This type of management seeks to reverse the damage from over 200 years of introduced pest impacts (Duffey 2001). The results of this intensive multi-pest species management at the Boundary Stream Mainland Project Treatment Site are summarized in Table 2.7.1. The control work at BSMIP began with a "second" 1080 aerial poison application, which reduced the residual trap-catch rate (RTC) of possums to less than 5% (i.e. 5 possums per 100 Trap nights) in forested areas (Cashes Bush, Thomas Bush, and Boundary Stream Scenic Reserve) about the Maungaharuru range (Christensen 1996).

Table 2.7.1 Pest Management Results for Boundary Stream Mainland Island Project

Pest Species	1995	1996	1997	1998	1999	2000	2001
Possums (RTC)	13.0%	23.1%	0.0%	0.2%	1.3%	0.2%	0.0%
Rats (TR)	-	14.0%	1.75%	0.25%	14.5%†	1.0%	0.67%
Goats (Kills)	-	485	951	241	114	123	42
Stoats (TKills)	-	4	25	30	48	78	51
Cats (TKills)	-	-	10	13	25	27	30

TR: Tracking Rate (Annual average), †: Removal of Poison Bait, TKills: Trap-Catch

Source: Boundary Stream Mainland Island Project; Database, Annual Reports: 1996-1998, 1998-2000, 2000-2001.

With the continued use of poison in bait stations in the Treatment Site (Figure 2.7.1), the residual trap-catch rate for possums remains at below 5% for the fifth consecutive year (Christensen 2000). This continued presentation of poison baits (initially Brodifacoum) in bait stations can maintain low possum populations in small reserves (Thomas *et al.* 1995). This was used as a justification and basis for establishing Mainland Island Projects. To give comprehensive coverage of the Treatment Site, bait stations were placed on a 150m grid, enveloping both possum, and rat home ranges, and initially operated on a monthly basis.

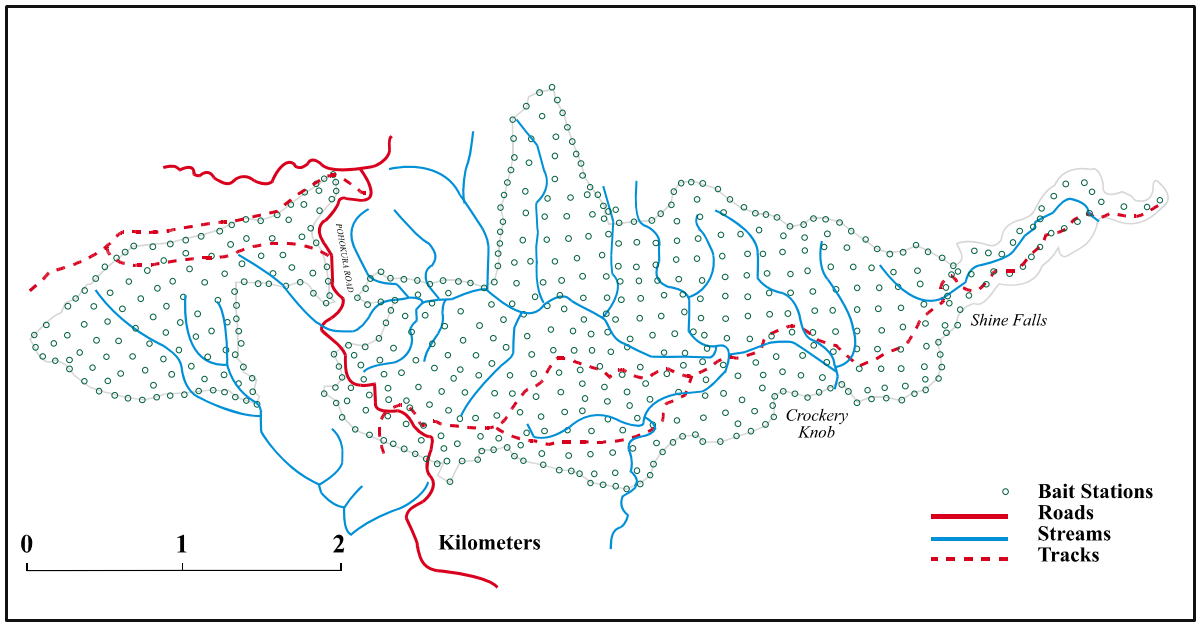

Figure 2.7.1 Possum and Rodent Management

Other pest species targeted for management in the reserve include; mustelids (ferrets, stoats, & weasels), pigs, deer, and the exclusion of stock by fencing. The management of goats occurs throughout the Treatment Site of Boundary Stream Scenic Reserve, as well as a surrounding buffer zone (Figure 2.7.2). This intensive multi-pest species management initially focused browsing pests. As they were reduced in number, greater management resources were shifted to rats, and predators such as cats, and mustelids (primarily stoats).

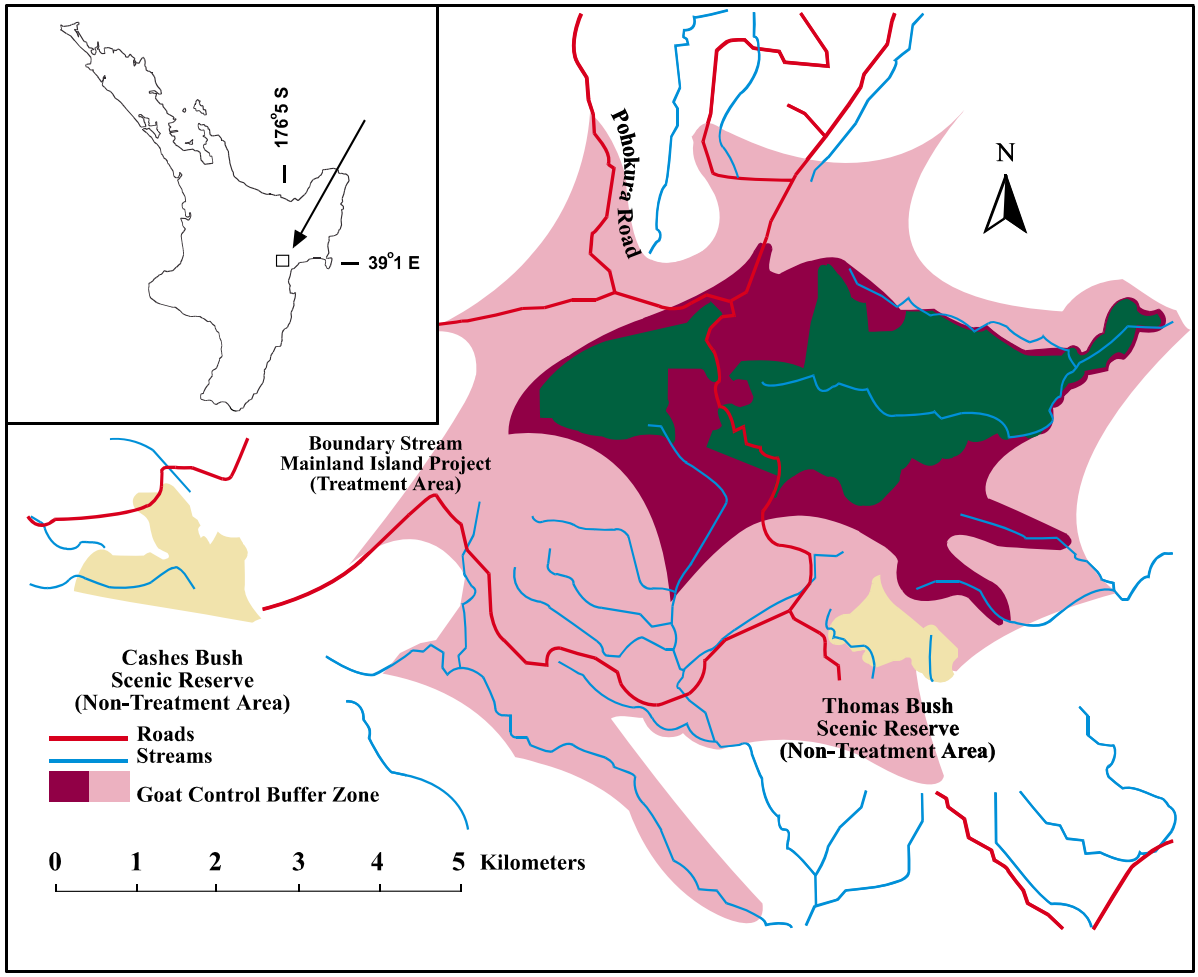

Figure 2.7.2 Goat Management Areas

2.8 General Conservation Management – Non-Treatment Sites

The conservation management that occurs at both Non-Treatment Sites (Cashes Bush Scenic Reserve, and Thomas Bush Scenic Reserve) is general in the sense that they receive no more pest species management than any other reserve in the East Coast/Hawkes Bay Conservancy. This pest species management can take the form of long-term pulsing of possums control e.g. once every seven years, and intermittent goat control e.g. once every three years or so. It is fairly apparent that the intensity of control is substantially reduced compared to that of the Boundary Stream Mainland Island Project Treatment Site. The monitored results of the "general" conservation management (pest control operations) performed at the combined Non-Treatment sites are outlined in (Table 2.8.1).

Table 2.8.1 Pest Management Results for Non-Treatment Sites (Combined)

Pest Species	1995	1996	1997	1998	1999	2000	2001
Possums (RTC)	9.3%	8.1%	4.2%	9.7%	22.7%	9.4%	2.75%†
Rats (TR)	-	35.3%	44.5%	48.8%	47.5%	37.0%	35%
Goats (Kills)	-	10	13	9	1	0	3

TR: Tracking Rate (Annual average), † Combined results of Cashes Bush & Maori Gully HBRC.

Source: Boundary Stream Mainland Island Project; Database, Annual Reports: 1996-1998, 1998-2000 (Draft), 2000-2001 (Draft).

The results of conservation pest management at the Non-Treatment Sites show limited effect on the possum and rat activity (and probably population) compared to that at Treatment Sites (Anon 2000). Rat activity remains more or less constant over the first five years since the project has been running, while possum activity has been somewhat variable due to the sporadic nature of control, as well as local environmental conditions during the time of monitoring (Anon 2000).

Chapter 3 Conservation Monitoring – A Review

3.1 Introduction to Conservation Monitoring

In his essay on conservation and ecological monitoring, Goldsmith (1991) states that monitoring has become "fashionable". Similarly Norton (1996) and Lougheed (*et al.* 1999) state that biological and ecological monitoring has now become common place in modern society as environmental issues come to the forefront of public awareness. This increase in monitoring, and auditing, highlights the progression of knowledge through science, and technology, and the evaluation of interventive management, as well as the explosion of information in the current age. While need and reasons exist for monitoring, there has been limited incorporation of practical guidance in the design of monitoring programmes.

The classic essay of T.C. Chamberlin, "The Method of Multiple Working Hypotheses" (1890 reprinted 1995), highlights the importance of objectivity and impartiality in scientific inquiry. By monitoring, or measuring more things (*i.e.* multiple working hypotheses), we gain more information and hence, more potential of association between components. Chamberlin (1890 reprinted 1995) lamented the dominance of single hypotheses, which would only confer simple explanations of complex phenomena. Michener (1997) echoes the importance of multiple working hypotheses by stating the good practice of increasing the potential of providing every rational explanation of the phenomenon in hand.

Spellerberg (1991) has stated the need for an ecological basis and understanding of ecosystems. The most basic objective of ecological monitoring is to detect trends (Krebs 1991; Spellerberg 1991). Ongoing monitoring provides the essential trend information on ecosystems; how they are changing and the rate of change. Monitoring additionally provides feedback on how intervention such as pest management affects components of the ecosystem and in the wider context the ecosystem itself. The BSMIP is attempting to approach this by relating the appropriate outcome (indigenous biota response) monitoring programmes to the corresponding result (multi-pest species management) monitoring programmes, in line with the DoC understanding of monitoring (Arand & Stephens 1998).

Goldsmith (1991) has outlined that monitoring must have clear objectives, similarly Kendel *et al.* (1989) states that objectives should determine the direction of monitoring. Objectives set a basis to record long-term environmental change and its ecological effects. In terms of management, objectives (such as determining appropriate sample sizes through pilot studies) convey information and criteria that are useful for future management, intervention, and projects. Dallimeier & Comiskey (1998) assert that good monitoring leads to good decision-making. Good decision-making should be the major reason and justification for monitoring in any management intervention.

Restoration programmes need to be well conceived and require monitoring as a critical element (Clewell & Rieger 1997). The most important aspect of ecological monitoring is in the development of appropriate statistical design (Hinds 1984). It becomes apparent that monitoring programmes require;

1. Clear well-formed goals, and objectives, and
2. Appropriate statistical design, and analysis to provide relevant and commensurate information.

The use of Statistical Power Analysis, principal component analysis, cluster analysis, and pilot studies *etc*, are suitable mechanisms to approach and determine reliable, effective, and efficient monitoring programmes.

3.2 New Zealand Conservation Monitoring Management

Monitoring has now become ubiquitous in modern society (Norton 1996), and is now part of many management programmes. Millard (1987) states that while (US) legislation has mandated the creation of numerous monitoring programmes to ensure or determine the integrity of the environment, many of the resulting monitoring programmes were ineffective, because they are based on poor experimental designs and a superficial knowledge of statistics. The need for better information on the state of the natural environment has been recognized in New Zealand (Anon 1997), and the Department of Conservation has identified this need as its very first key step in its strategic business plan; 'Restoring the Dawn Chorus 1998-2002'. Addressing this issue will continue to be vital to the Department as it moves to a more outcome-oriented focus (de Bres 2002), and has subsequently been identified under the 'new' first key step of 'Expanding the biodiversity effort' of the 'Restoring the Dawn

Chorus 2002-2005' (Anon 2002). Within New Zealand, environmental and ecosystem change has often been poorly documented (Allen 1993). Early monitoring methods concentrated on the presence of species, and later started to include relative measurements of the numbers and abundance of individuals of certain species, and groups such as vascular plants. Techniques such as permanent plots or quadrats e.g. the permanent vegetation plot (RECCE) method, have provided a quantitative record of the environment (Austin 1981; Allen 1993).

Such application of this and other methodologies e.g. animal population analysis, mark-recapture studies have provided key information for the identification of the ecological and environmental processes involved, especially in terms of conservation intervention. For example; using the permanent plot method in the evaluation of the Breaksea island restoration programme;

> "*Population age structures and recruitment of major tree and shrub species on Breaksea Island, Fiordland, New Zealand were assessed at the time of eradication of Norway rats in 1988 and over the following 5 years…. Seedling numbers of many tree and shrub species increased substantially over the period 1988-93 after rat eradication*." (Allen *et al.* 1994).

Bellingham *et al.* (2000) performed a review of the permanent plot record for the long-term monitoring of New Zealand's indigenous forests, and outlined the need for a plot network as a key part of developing well-formed decisions in conservation management. Non-forest ecosystems are also important components of New Zealand's landscape, and Dickinson *et al.* (1992) utilized a height-frequency method of sampling (tussocklands, shrublands, and wetlands) to demonstrate the need for a more standardized procedure of obtaining relevant information for conservation management.

The determination of the success of management programmes is reliant on the detection capabilities of the monitoring method used. The method has to incorporate the monitoring design of the programme or study. Towns' (1994) publication on the Whitaker's skink (*Cyclodina whitakeri*) conservation as part of the Korapuki Island (Mercury Islands) ecosystem restoration programme, showed that the;

> " *removal of rats increased lizard numbers within 12 months and rose 30 fold over 5 years, but measurable increases of lizard numbers in forest areas took up to six years*" (Towns 1994).

This study had an excellent survey design, with (before and after control) monitoring. Both monitoring occasions were temporally replicated (seven x two samples (taken yearly)). This methodology (survey design) would most likely have a moderate to high statistical power. Towns (1994) further states that previous measurement of the effects on the lizards would have been based on circumstantial comparisons.

The Department of Conservation invests considerable effort and resources in monitoring, which must address the issues of what factors influence the populations. Monitoring programmes must be designed according to the specifics of the response to be measured, and done appropriately (Arand & Stephens 1998: Brown & Miller 1998), including the comparability of Treatment and Non-Treatment sites (Brown & Norton 2001). Current and future studies in monitoring for conservation management in New Zealand will be focused on the finer determination of change, over a range of scales. For example; Sherley (1996) identified the importance of biodiversity assessment and mapping methodologies and techniques e.g. (rapid biodiversity assessment); Hutcheson *et al.* (1999) evaluation of invertebrate indicator species for monitoring change in terrestrial habitat quality; and Park (2000) proposed future conservation management at the ecosystem-landscape scale.

Restoration experiments have become an important component in the Department of Conservation's actions. They represent the most important tests of ecological theory (Ewel 1987; Michener 1997). Restoration projects are however by their very nature multivariate and messy (Clewell & Rieger 1997). They should be evaluated as fully as resources allow, and should be undertaken with a robust level of statistical reliability. Power analysis can determine this "robustness" of an experiment or monitoring programme, and provide methods for improving that design (Fairweather 1991).

3.3 Boundary Stream Mainland Island Project's Monitoring Programmes

Boundary Stream MIP is described as an ecosystem restoration project (Christensen 2000). The focus of Boundary Stream's monitoring programme should then be on community and ecosystem level processes. Boundary Stream approaches this by considering strong guild-guild relationships (Figure 3.3.1), such as rodent (rats, and mice) and mustelid activity with invertebrate abundance (terrestrial, and aboreal insects). Of the fifteen long-term monitoring programmes established at the Boundary Stream Mainland Island Project, seven major programmes (comprising 22 individual levels) have been analysed for this thesis.

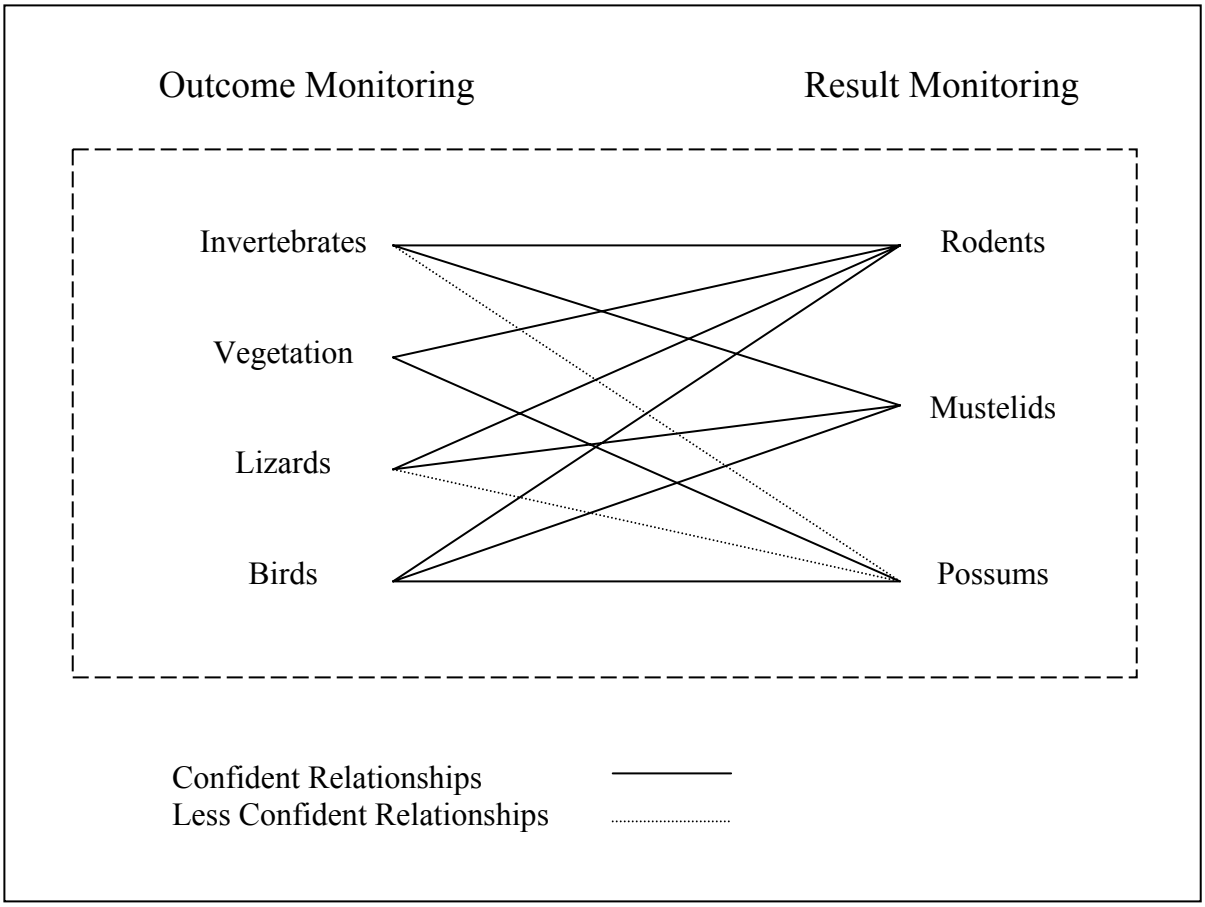

Figure 3.3.1 Monitoring Programmes within BSMIP, and their considered guild-guild relationships.

Adapted from Christensen (2000). Source Boundary Stream Mainland Island Project Annual Report 1996-1998.

The above figure shows the seven broad monitoring programmes (wētā monitoring is incorporated into the invertebrate monitoring programme) divided into the two broad descriptions of conservation monitoring as outlined by Arand & Stephens (1998), that of result and outcome monitoring. While these complex trophic relationships are difficult to determine responses to pest management (Choquenot & Parkes 2001), they represent the future of conservation and ecological monitoring and management. Inferences made from monitoring programmes must be carefully considered. Results from monitoring programmes may be influenced by unknown factors that are not necessarily controlled for. Inferences that are made may not be essentially wrong, they may however have little guarantee in being right (Hairston 1989).

3.4 Conservation Monitoring Principles – Guiding Objectives

Restoration projects require adequate measurements of the change in condition of the environment. Monitoring is the procedure, and collection of such measurements, and necessitates an approach to determine what and how much effect the management is having. Trenkel (2001) states that wildlife management is generally carried out under conditions of uncertainty; with questions such as the exact population size being unknown, its future dynamics and condition being uncertain, and with often an absence of specific management objectives. Conditions of uncertainty will be present in all restoration projects, and as Montalvo (1997) states, every restoration attempt can be seen as a field experiment. It is imperative that objectives are established which guide, and link monitoring to management (results) and its effects (outcome).

Monitoring programmes for restoration efforts need to conform to a hierarchy of importance primarily to address the main questions of cause and effect (which if monitored equate in general as Result and Outcome components). These result and outcome components can be identified for monitoring programmes by biological linkages, such as trophic-connections, over large to small spatial (geographical area) scale, depending on the interest and direction of the restoration efforts. Secondarily, the temporal aspect of discrete monitoring units needs to be considered, initially with a "Before and After" approach, and then ongoing trend monitoring (Figure 3.4.1). This also requires a control (Non-Treatment) site or state to be

monitored as a comparison over time. Burnham & Anderson (2001) state that the basis for monitoring, is to register any changes due to management, and stress the importance of developing an '*a priori*' determination of what to look for, measure or interpret. This section outlines a protocol for the monitoring of restoration efforts, and evaluation of the BSMIP's monitoring programmes hierarchy.

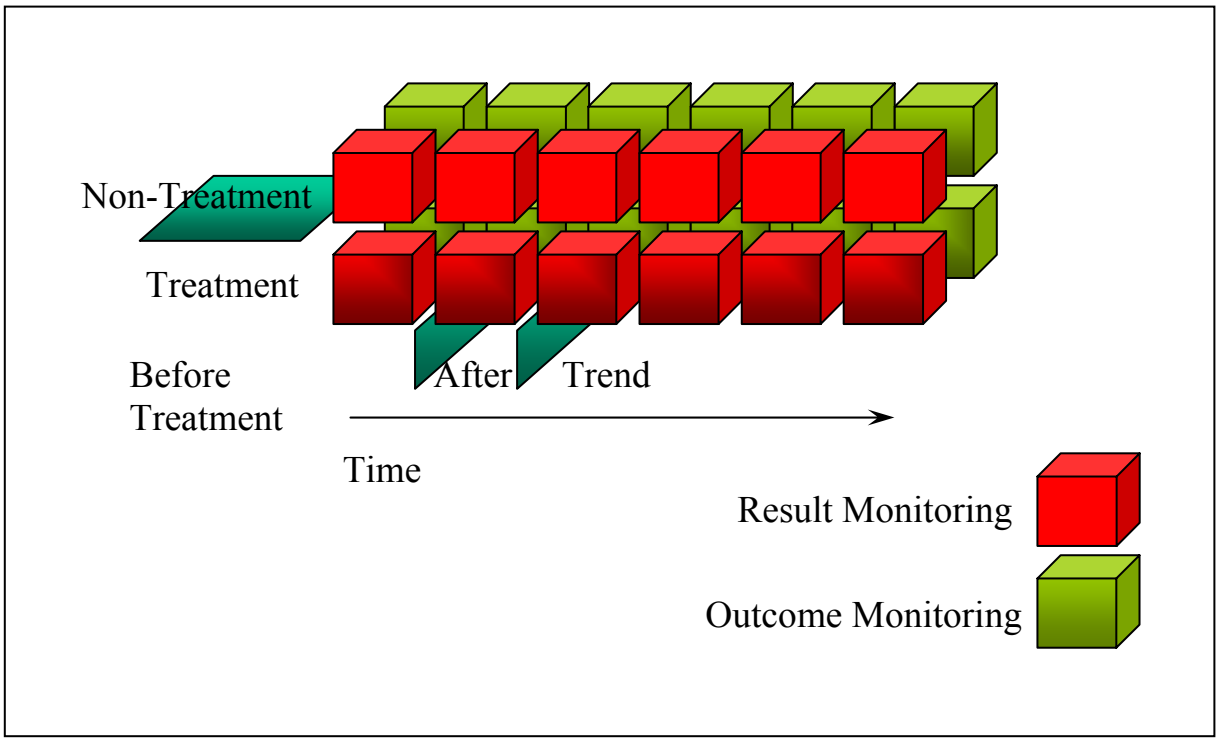

Figure 3.4.1 Diagram of Monitoring Over Time

3.4.1 Purpose of Measurement

The establishment of a monitoring programme, should start with a question;

Why are we measuring? and to what purpose?

In the conservation context, of primary importance is the measurement of ecosystem components and processes in relation to changes due to management (Arand & Stephens 1998). These components and ecological processes, are ones in which we place value in the New Zealand conservation, i.e.; benefits to indigenous species, habitats, and processes such as the honeydew cycle in beech forests, and the reduction of "pest" species such as possums, rats, goats, *etc*, and their impacts. Such components and processes, are most often determined by key research (pure science), on the biology and ecology of the biota such as

dietary studies (predation effects), population studies, and life histories, which are a necessity for any monitoring programme (Finlayson & Eliot 2001). Research makes apparent the direct interrelationships between predator and prey species, and between trophic guilds, which would be appropriate for monitoring restoration efforts. The complex interactions between species and guilds should be used as the basis for environmental policy and management and not on keystone species (Mills *et al.* 1993). Lawton (1999) states that generalizations exist within trophic web theory, and while the complexity of trophic relationships makes the ability to set pest threshold parameters difficult (Choquenot & Parkes 2001), trophic webs (and its basic unit Predator-Prey relationships) are still the best theories on which to base monitoring.

3.4.2 Conservation Project Type, & Information Collection

The BSMIP is has an adaptive management (also termed research by management) approach, using current known 'best-practice' conservation management. As a key part of this adaptive management approach, the Mainland Island Projects including the BSMIP, develop and trial new techniques and methods in order to advance conservation management. This thesis is one such example of adaptive management, reviewing what I term as the validity of the monitoring programmes. BSMIP also has broader goals, as a focus for conservation awareness and community relations, a 'showcase' for conservation, though is not the subject of this thesis. BSMIP has attempted to encompass a comparison of the state of the ecosystem (species, and guild abundance) before management intervention in both Treatment and Non-treatment sites, to that of after, conferring an understanding of the changes brought by an intensive multi-pest species management. While BSMIP incorporates an adaptive management approach, including Research by Management (RbM), the BSMIP is not wholly an experimental research project, and exists on a continuum of management.

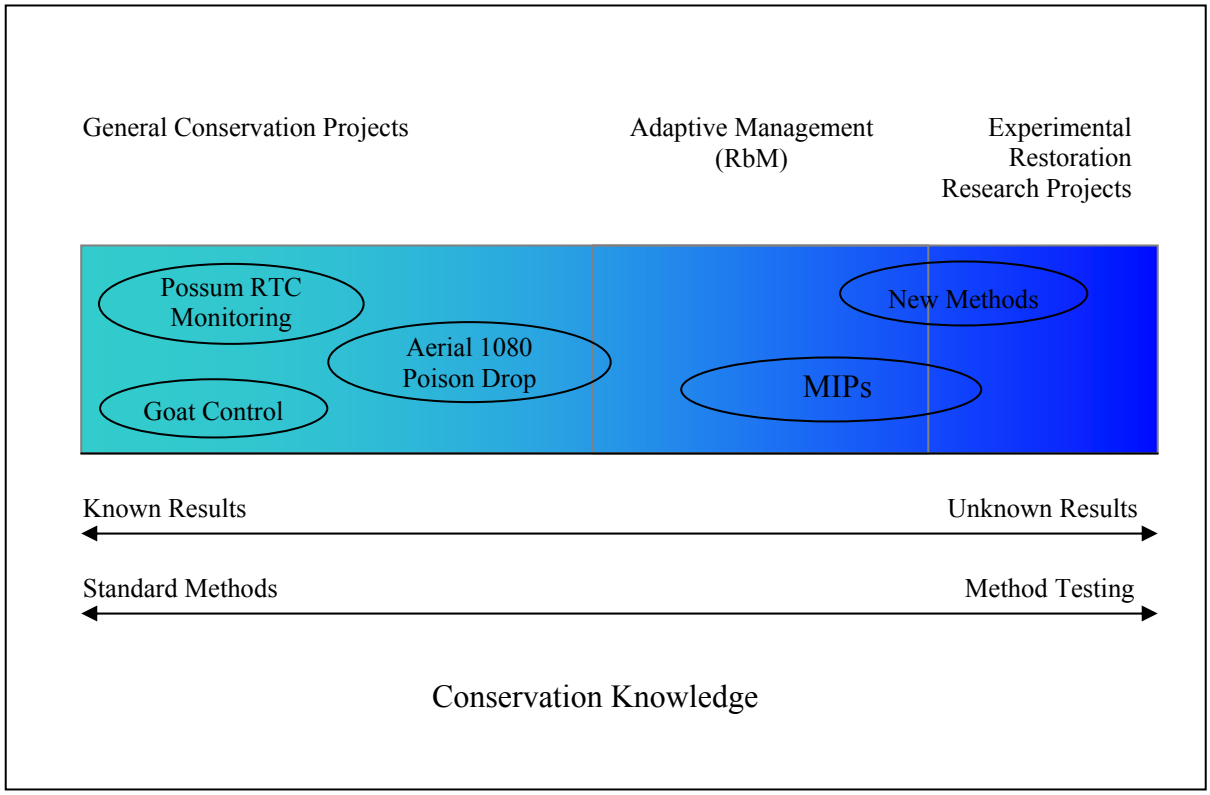

Figure 3.4.2.1 Continuum of Conservation Management Knowledge

Ongoing trend analysis (divergence of ecosystem state and condition) of the Treatment and Non-Treatment sites will confer the most beneficial information for the efficacy of the BMSIP. It would be useful to test the efficacy of the BSMIP in terms of conservation merit by placing it in a wider scope of analysis (such as meta-analysis), including other intensively managed projects.

Research, monitoring, survey, and census (inventory) as information gathering processes, all exist on a continuum of intensity and scope of measurement. Because they have different core objectives relating to the levels of certainty required in the conclusions, the design of research, monitoring, projects *etc* must often be balanced against their respective costs (Wiens & Parker 1995). Monitoring differs from the others above in that it usually measures a sample of the population of interest, with this measurement conferring an approximate statistic of interest directly attributable to that population, in an ongoing (serial) fashion. Monitoring has a broad scope, moderate intensity, and serial occasions of

measurement, whereas research often has a narrower scope, a greater intensity, and fewer occasions of measurement (Figure 3.4.2.1).

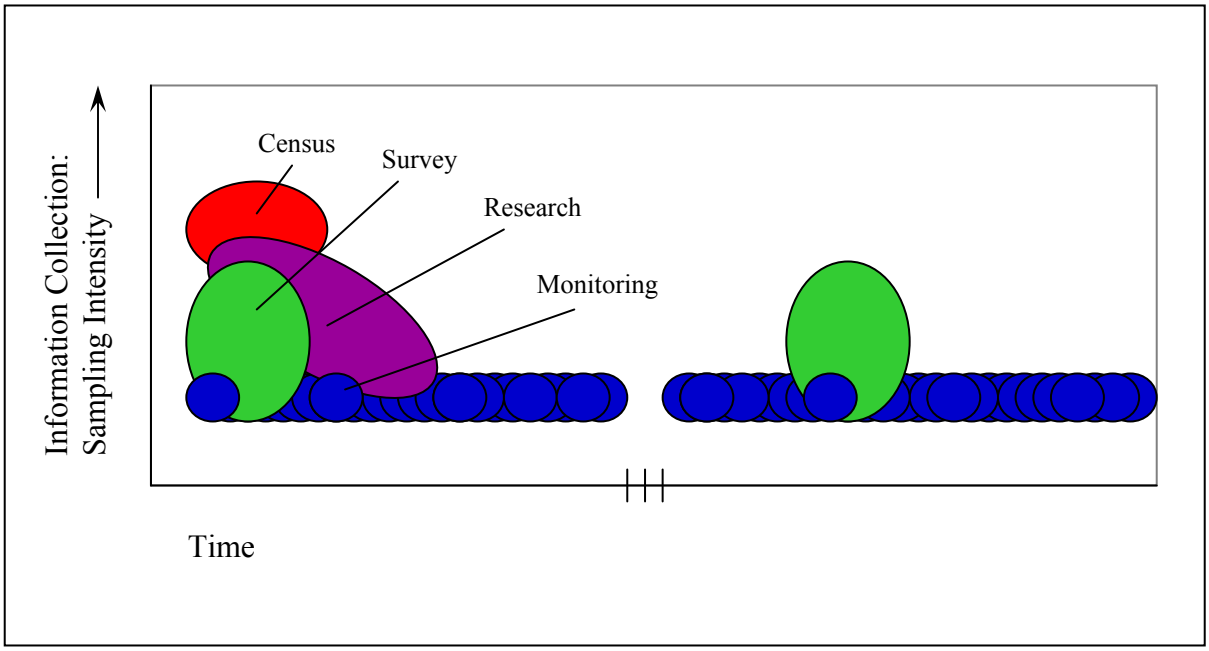

Figure 3.4.2.2 Sampling Intensity of Different Information Collection Methods

With adequate monitoring, the extent to which management objectives have been achieved is more apparent (Gerber *et al.* 2000). Thus 'robust' monitoring is more directly related to conservation management at such ongoing projects.

3.4.3 Monitoring Hierarchy

Benedetti-Cecchi (2001) states that the detection of anthropogenic disturbances requires an appropriate sampling design and statistically powerful tests. The identification of the specific level or scale of inquiry is also necessary, and will influence the sampling design, and statistical analysis. Such a clear exposition of the ecological hierarchy of organization will help clarify goals and methodologies (Aronson & le Floc'h 1996). The incorporation of a monitoring hierarchy, in which each level of inquiry e.g. whether; individuals, populations, guilds (indigenous, exotic), and trophic levels (producer, consumer, decomposer) is identified prior to the development of a sampling design, would be highly useful in

addressing a comprehensive monitoring programme. Each level should be considered as 'sub'-monitoring programme designs, with each 'sub' design measured for their individual statistical robustness.

Biological units seem to occur in geographic space in patterns of complex nested hierarchical sets, and are generally not discretely spatial located (Lawless & Stephens 1996). Lawton (1999) states that subtle processes and patterns emerge at the mesoscale level (between regional and the local community levels), and that such large scale hierarchical filters (monitoring) are the key to finding general patterns in ecology. A key assumption of hierarchy theory is that smaller sub-systems change more rapidly than larger systems (Towns & Williams 1991; Holling 1992; Norton & Ulanowicz 1992; Lawless & Stephens 1996; Norton 1996). This hierarchy, and specifically the disparate rate of change between hierarchical levels produces discontinuities in spatial and temporal scales, which will influence conservation (Lord & Norton 1990), and the design of monitoring programmes (Wiens 1989; Noss 1990; Norton 1996). An appropriate monitoring pattern (and hence design) should mirror that of the actual biological units distribution at a specific level, e.g. generally monitoring programme designs for biological conservation would have a spatially clustered design.

3.4.4 Temporal Monitoring Parameters

It is important to consider the temporal objective of any monitoring study, in order to determine whether a change in an ecological system over time has occurred (Magnuson 1990). The BACI – Before/After, Control/Impact monitoring design is the most utilitarian (Green 1979), and an appropriate 'quasi-experiment' suitable for the inductive and inferential nature of ecological studies (Hargrove & Pickering 1992), and vital at the initiation of any interventive management such as restoration efforts. BACI designs in general, are however not as appropriate for determining changes from continuous or serial conservation management as designs that have incorporated ongoing monitoring. The item of key interest is the trend of populations, or health of the ecosystem, which is the objective of conservation, and outlined within the Department of Conservation's Mission;

To conserve New Zealand's natural and historic heritage
for all to enjoy now and in the future.

He āta whakaute, manaaki, me te tiaki ia Papatuanuku ki Aotearoa kia ū tonu ai tōna whakawaiūtanga hei oranga ngakau mō te tini te mano ināianei, āke tonu ake.

(Anon 2002).

This conservation both "now, and in the future" requires the understanding of temporal scales, and the incorporation of a trend design to monitoring. Hobbs & Harris (2001) state that the majority of ecosystems are dynamic, and hence restoration project goals cannot be based on static attributes. Likewise a monitoring programme needs to capture this dynamism over time. Long-term monitoring is necessary for ecology, or else susceptible to serious misinterpretation (Magnuson 1990). Magnuson (1990) elaborates on the core ecological explanation, that often a lag period will occur before an effect is apparent in ecological systems.

It is important to link the statistical approach (monitoring design) to the objectives of a study, or project. Eberhardt (1976) states that the use of long-term post-operational monitoring has limited relevance to the baseline, over time, and instead reflects the long-term trends. In conclusion, for any long-term project that is going to be established, it must have a temporal monitoring hierarchy, that moves from a BACI design initially, which then moves to a trend monitoring design (which increases the importance of habitat replicates).

3.4.5 Review Conclusion – Guiding Objectives

Guiding objectives are paramount for an effective monitoring programme. Hinds (1984) suggests a stepwise and systematic development of long-term methodologies akin to a hierarchy for the monitoring of long-term trends in terrestrial ecosystems. I would similarly suggest such a task for the BSMIP; comprehensively mapping out the result and outcome monitoring programmes over time. This would provide an overall planning methodology to determine the dynamics of these relationships as well as the actual monitoring occasions, and would depict the transition from a BACI design to trend monitoring for each monitoring programme, and how each level of inquiry of each programme exists within the likely monitoring hierarchy. This would identify potential information 'gaps' and research needs that may require a monitoring programme to (temporarily) increase the level of sampling intensity to answer specific questions; e.g. the relative population abundance (artificial wētā roost occupancy) of Tree Wētā due to intensive mammal control (Christensen 2003).

The clear exposition of the BSMIP monitoring programmes guiding objectives will provide a clarification of what BSMIP is as a project. It is a restoration experiment, and also a 'showcase' for conservation. It has the potential to develop its vision to encompass the landscape-scale conservation management (which is already being initiated with goat management: see figure 2.7.2); conserving the whole of the Maungaharuru Range: with Boundary Stream Scenic Reserve as its heart; and/or conserving the water catchment from the range to the sea, again with Boundary Stream Scenic Reserve as its source. For this to happen both now and in the future, restoration efforts (including monitoring) will require careful consideration and planning to determine the restoration objectives and criteria by which it will be measured.

3.5 Conservation Monitoring Principles – Biological Relevance

The ideal indicators to be used for monitoring ecosystem health, are those that are biologically relevant (Cairns *et al.* 1993). This biological relevance, is based on indicators that are representative of a 'pristine' environment, and/or the values that we as resource managers wish to promote, which for conservation in New Zealand would be indigenous biological diversity or indigenous 'biodiversity'. Conservation monitoring requires more than that, 'biological relevance' needs to depict a relationship whether direct or indirect between cause and effect. It needs to determine whether the management performed has a benefit to those ecological processes, and biological entities in which we (DoC, the New Zealander public, conservation managers, *etc*, i.e. stakeholders) place a conservation value. As Krebs (1991) states, monitoring is politically attractive although 'ecologically banal' if no research or experimentation is included to comprehend the ecological mechanisms behind the system changes. This section outlines the importance of a balanced monitoring programme that incorporates both result and outcome monitoring components.

3.5.1 Biological & Statistical Significance

Statistical significance testing and its relevance to biological significance has been in question since early on from the development of statistical significance testing, and from a number of authors (Berkson 1942; Carver 1978; Jones & Matloff 1986; Perry 1986; McBride *et al.* 1993; Johnson 1999; Anderson *et al.* 2000). Green (1984) identifies the

important though perhaps obvious need that biologically defined objectives should determine the statistics rather than the reverse. Yoccoz (1991) states that only biologically-based considerations can be used to determine the amount of difference we are (or should be) focusing on for any changes within natural resources. Yoccoz (1991) further highlights the need to look at the robustness of the theoretical models in order to decide how big a difference 'effect size' must be to be considered 'biologically' significant. Whether the effect size obtained from the monitoring or study results is of appropriate size (a biologically significant change) for using a specific test is one if not the main use of statistical power analysis (Fairweather 1991; Norton 1996; Cherry 1998). Hayes & Steidl's (1997) statistical power analysis of amphibian populations has outlined the danger of suggesting inappropriate management actions based on weak inferences, and promotes the focus on the rate at which a population is changing with time, the size of change, and how much change is important. These three parameters are the key biological parameters that researchers, and resource managers should be investigating.

Mills *et al.* (1993) states that environmental policy and management should be based on complex interactions among species. The complexity of ecosystems makes the finding of key indicators extremely difficult (Hellawell 1991), likewise determining how particular wildlife populations will respond to pest control is often difficult (Choquenot & Parkes 2001). One of the best methodologies we have for such a determination is the use of trophic relationships (those that exist within a trophic web). Such identification of trophic web components stems from research such as dietary studies, and resource-use studies, to which projects such as the BSMIP can contribute too, with specifically long-term information.

3.5.2 Biological Linkages – Result & Outcome Monitoring

Arand & Stephens (1998) divides conservation monitoring into the major components of result and outcome monitoring. Result and Outcome monitoring programmes within a conservation project should have established biological linkages between monitored groups, and can be either deleterious or beneficial in terms of conservation value, though all add to the collective knowledge of conservation management *e.g.*;

Bird Counts & Mistletoe Monitoring: Nectivorous, or 'brush-tongue' birds (Tuis, Bellbirds, and Silvereyes) and linkage to dispersal of mistletoes (Ladley *et al.* 1996; Ladley *et al.* 1997).

Possum Monitoring & Vegetation Monitoring: Possums impacts on vegetation, and change of forest composition (Allen *et al.* 1997; Norton 2000; Payton 2000).

Ungulate Monitoring & Vegetation Monitoring: Deer changing structural composition and regeneration of coastal forests of Stewart Island by preferential feeding (Stewart & Burrows 1989).

Invertebrate Monitoring & Ecosystem Processes: Ants as indicators of restoration success following disturbance (Andersen & Sparling 1997).

The BSMIP monitoring programme utilizes well-known biological connections between pest species, and indigenous species (Christensen 2000). The identification of potential research e.g.: the effect of intensive pest control on tree wētā artificial wētā roost occupancy (Christensen 2003), should advance such knowledge by providing finer scale detail, than can done by monitoring *per se*. Sessions *et al.* (2001) has shown that possum control operations can improve mistletoe (*Alepis flavida*, *Peraxilla* spp.) health, and it is likely that the possum management performed within the BSMIP Treatment site has had a beneficial effect on mistletoe abundance (Christensen 2002). The possum-vegetation interaction is not a simple one, with possums targeting certain species, or even certain individuals, within a forest (Norton 2000). A need exists to better understand the relationship between possum density after control, and the vegetation response (Norton 2000). Such needs are ubiquitous in conservation management, and the BSMIP is well placed to contribute to such investigations.

3.5.3 Review Conclusion – Biological Relevance

Monitoring programmes need to be balanced with both result and outcome components that have either a direct or indirect relationship. This balance should be based on the established and known significance of the biological linkages between components of the monitoring programmes. A need exists for targeted research on such relationships, with further investigations into indicator species, and groups. New BSMIP monitoring programmes

should be able to additionally focus on finer scale parameters such as; abiotic interactions, and relative difference in breeding success, *etc*, as well as the larger coarse landscape-scale. The biological relevance of each BSMIP monitoring programme is further elaborated within section 6.4.

3.6 Conservation Monitoring Principles – Statistical Reliability

If resource management agencies are to become effective resource stewards, they must be able to evaluate restoration and conservation efforts and more comprehensively (Berger 1991). As biological resources appear to occur in geographic space and time in patterns of complex nested hierarchical sets (Holling 1992; Norton & Ulanowicz 1992), we must view the actions and processes on the appropriate scale (Wiens 1989), so as to more reliably assess distribution and abundance. This necessitates a systems, and hierarchical approach to monitoring to develop such an understanding. A monitoring programme design in its basic form should be concerned about focus and scale; numbers and distribution of sites (scale), and numbers of sampling units at individual sites (focus). The hierarchical system approach to monitoring addresses ecological scale issues, by focusing monitoring at different levels; e.g. species, guilds, and trophic levels, providing information for conservation management on the interrelationships of species through trophic guilds, and predator-prey dynamics.

3.6.1 Sampling Focus

A primary focus of any monitoring framework must be to minimize the consequences of inevitable inaccuracies and uncertainties involved in ecosystem management (Cairns *et al.* 1993). Similarly, Caughley & Sinclair (1994) note that; the precision of (animal) counts, and indices is of the utmost importance. A key criterion for precise values, is to ensure that an index of variation (Coefficient of Variation *CV*) is obtained (Engeman *et al.* 2002). This can be best encompassed by having a number of sampling units (plots: by estimating spatial variation, repeated sampling occasion: by estimating temporal variation) of the area or unit of interest.

The type of measurement design will affect the robustness of data, with survey, or presence-absence designs often subject to bias, and inadequate power (Strayer 1999), this is an

important case for such low density indices such as mustelid monitoring, or even ongoing trend monitoring for rats, and possums. Taking too many samples will waste time and resources, though taking too few samples will make the study meaningless, and possibly lead to errors in interpretation (Eckblad 1991). While repeated measurements of an area or unit of interest is of prime importance, the methodology needs to balance the use of resources, to the precision of the mean estimates. It is vital for monitoring to focus on the item of interest and replicate the measures, through a number of samples taken within a site, and also over time – so that a Coefficient of Variation (CV) is gained.

3.6.2 Sampling Scale

The scale of monitoring programmes should have a direct relationship to its focus, and must be appropriate to the individual species of concern, as pattern-process interactions involving organisms are scale-dependent and require an organism-based view (Turner *et al.* 1995). While this is vital for effective monitoring programmes, I see that a design should not only encompass an 'organism-based' or species-based focus, though should be able to feed up into higher biological level focus (e.g. trophic guilds, communities, and populations, *etc*). Hence a hierarchical monitoring programme adds a far greater value, especially to 'ecosystem restoration' projects, and indeed for environmental management (Mills *et al.*1993; Lawton 1999).

Magnuson (1990) states that a lag period will exist before any intervention effect can become apparent in ecological systems. It is thus important that a consideration of temporal scale is incorporated into monitoring. Ecosystems vary in complex ways at several spatial and temporal scales, and all reference information such as monitoring will be implicitly time- and space-based (White & Walker 1997). Green (1984) argues for long-term studies to obtain reliable estimates of baseline variation on the among-years time scale and to determine the long-term effects of impacts on complex systems. Magnuson (1990) promotes the necessity of long-term monitoring, else monitoring programmes are susceptible to serious misinterpretation, as the likelihood of reaching a statistical reliable timeframe, e.g. vegetation 10 + years, is greatly diminished. Whether and when a statistical robust level was reached, was a key part of the evaluation of statistical reliability for the BSMIP monitoring programmes.

3.6.3 Sub-sampling

Turner *et al*. (1990) states that scale is important for detection of ecological patterns, and the respective statistical interpretation. It is thus important to target monitoring to account for changes throughout the biological organizational level; at the local level; individuals, species, guilds, and at the wider regional level; communities, and populations, to address the breath of ecosystem responses due to management. The levels of interest need to be determined so that they can be adequately addressed. For example it is of interest to determine changes due to predator control in species abundance such as Bellbirds, though also nectivorous birds, indigenous birds, and total bird numbers.

While sub-sampling can substantially reduce variation (Gibbs & Melvin 1997), it is often mistakenly substituted for true replication, with sample sizes too small for adequate statistical power (Eberhardt & Thomas (1991). If monitoring for a range of biological levels, then monitoring at each level should be considered as an independent monitoring programme for the purpose of determining statistical 'robustness'. Thus each sample site should be independent. If the focus of monitoring is concentrated on single-species, then it would make better sense to target such species and dedicate time and resources, such as a mark-recapture study; e.g. for the recommended BSMIP periodic Tree Wētā mark-recapture study (Christensen 2003).

3.6.4 Effect Size & Coefficient of Variation

The magnitude of effect size (ES) in ecology-based studies is a relative measure of biological change, or 'true effect' (Peterman 1989; Goodman & Berlin 1994). ES is the difference between the results predicted by the null hypothesis and the actual state of the sample universe (population) being tested (Thomas & Juanes 1996; Thomas 1997). The sample effect size necessary for effective statistical significance testing is notably larger than the actual population effect size, this can express estimated effect sizes with a substantial positive bias (Cohen 1994). Cohen (1994) makes the statement that there is; *'No magic alternative to null hypothesis statistical testing',* and that before generalizations are made, we must first seek to understand and improve the data, following John Tukey's lead, in data '*detective'* work rather than data '*sanctification*'.

As the effect size approaches zero the realized power of a monitoring design or test becomes vanishingly small (Fairweather 1991). This makes the reliable detection of very subtle differences almost impossible without massive replication and hence great cost (Fairweather 1991; Mapstone 1995). Likewise Johnson (1999) states that the focus should be on whether an effect size of a magnitude that has been considered to be important has been consistently obtained across valid replications. This thesis focuses on changes within the differing forest habitats (and sites) between the different treatment sites, that could be measured by the individual monitoring programmes.

The estimation of a mean value should be accurate to within +/– 25% (Eckblad 1991). The expression the variability of the data, or Coefficient of Variation (CV), is highly important, as it provides an indication of how precise is the measurement of the estimate. The smaller the CV, the fewer survey units would be necessary to detect a change with the desired power and confidence (Ribic & Ganio 1996). Such summary information from preliminary studies, provides direction no only on what, where, and when to focus our attention, and monitoring, though also how much effort is required. This gives a guide to what is a biologically significant change due to management, and directs the need to postulate a biologically significant effect size (Hayes & Steidl 1997).

Cohen (1988) notes the importance of specifying the effect size, in the form of confidence intervals (Cohen 1994). It also important to report the criteria other than the significance value, such as alpha levels, and effect sizes, *etc*. (Steidl *et al.* 1997). I consider it both useful and critical for the comprehension of the effect of management intervention to state the other statistics such as CVs, ES, statistical power, and *N* (number of certain values in the population), *etc*. Osenberg *et al*. (1994) states that the key consideration is the likely effect size in the detection of environmental impacts, and by consequence the responsible scientist must also be concerned with factors which determine effect size (Ottenbacher 1996). The design of a monitoring programme needs to limit the CV, and so maximize the precision of determining the magnitude of effect size. The BSMIP approaches this by locating independent monitoring sites within specific habitat-types, and having replicated sampling sites within the habitat-types. Double sampling as suggested for tracking tunnels by (Blackwell *et al.* 2002) offers the chance of calibration of effect sizes, and increases the

confidence of any observed trends. The BSMIP has a number of covariate monitoring projects, that would support and qualify the observed changes, and trends obtained from the main "core" monitoring programmes.

3.6.5 BACI (Before-After-Control-Impact) Designs

The design of an impact/monitoring study, the statistical analysis through to the interpretation of results are inseparable (Green & Montagna 1996). The design should reflect and be directed by the kind of inferences a researcher is attempting to make. Green (1979) has stated that BACI (Before-After-Control-Impact) designs would generally be the most utilitarian for detecting environmental change e.g.;

> "*A design where the same sites are sampled both before and after impact is more efficient than a design where sites are reallocated for the after-impact sampling*."
>
> (Green 1989).

Repeated measures designs (based on resampling replicates (e.g. sites) at a series of times) are appropriate for environmental impact and monitoring studies (Green 1993). Thus the BACI design, including repeated pre- and post-treatment monitoring would be among the more reliable determinants of the effect of conservation intervention management. The BSMIP is an ongoing combination of determining the effect of the different management regimes, utilizing the 'before' characteristics (of the Treatment & Non-Treatment Site/s) prior to the incorporation of intensive management, and the current 'control' characteristics of the Non-Treatment Sites over time.

Faith *et al.* (1991) compared the statistical power of two different monitoring designs and associated statistical tests to outline the importance a minimal number of years of baseline pre-disturbance (management or intervention) data to achieve the required statistical power. This characteristic of temporal replication (especially for baseline 'pre-operational' data), and is usually missing from most of the Department of Conservation's monitoring programmes. Faith *et al.* (1991) states that pilot studies provide critical data not only for evaluating alternative community summaries (e.g. differences based on different taxonomic levels, different dissimilarity measures, and different data standardizations) and also for determining the number of replicate time periods needed in the actual development. This is

a point echoed in vegetation studies (Austin 1981; Dickinson *et al.* 1992). It becomes apparent that temporal dependency is as common as spatial dependency, hence the importance to compare a comprehensive set of differences before the disturbance and at least a similar set of differences after the disturbance.

In difference to Green (1979; 1984), Underwood (1992) notes that the use of replicated treatment and non-treatment sites is an ideal only for statistical purposes, and also that there is no reason for more than one non-treatment sites, apart from cost, and possibly subject comparability. However, Underwood (1994) later suggests that including several control (non-treatment) sites in a (asymmetrical) monitoring design leads to greater reliability of detecting a variety of environmental impacts. This represents a progression from BACI to which BSMIP approaches, e.g. the inclusion of an additional (i.e. third) Non-Treatment site; Waitere Kiwi Conservation Area as a kereru monitoring site.

Hargrove & Pickering (1992) question whether classical experimentation is adequate for real progress in landscape or regional ecology, which are like other scientific disciplines; astronomy, geology, medicine, & psychology in that they have a limited ability to replicate experiments, and that ecology as a science is inferential and inductive. The BACI design becomes as (Hargrove & Pickering 1992) states a 'legitimate quasi-experiment'. A research or monitoring design's 'robustness' comes from its ability to obtain repeated, reliable, and precise data from multiple sample sites, to which the BSMIP monitoring programme approaches. Hargrove & Pickering (1992) however make the point that quasi-experiments are no panacea, as stochastic processes (nondemonic intrusions) have greater impact on quasi-experiments than other designs. It thus becomes important to have strong 'statistically reliable and robust', hierarchical monitoring designs if possible.

3.6.6 Long-Term Monitoring

Eberhardt (1976) states that long-term post-operational monitoring has limited relevance to the baseline, over time, and instead reflects long-term trends (refer Figure 3.4.1). Time series analyses (long-term monitoring) may have greater statistical power to detect trends, and only for long-term (>10-12 year) data sets and only when statistical assumptions are met (Beier & Cunningham 1996). While these elements are correct, both Green (1984), and

(Hinds 1984) argue for long-term monitoring to obtain reliable estimates of baseline variation on the among-years time scale and to determine the long-term effects of impacts on complex systems. At BSMIP the core management focus of 'intensive multi-pest species' control is expected in some form to continue over at least the next couple of decades. The long-term evaluation of whether BSMIP is a success will be driven by the comparison results of the difference management regimes, *i.e*. intensive multi-pest species management, and that of general conservation management.

Michener (1997) argues for a number of appropriate research approaches including long-term studies, large-scale comparative studies, and modelling, to lead effective restoration projects. Wolfe *et al.* (1987) and Pechmann *et al.* (1991) state that long-term records of biological data are extremely valuable for documenting ecosystem changes, the differentiation of natural changes from those caused by humans, and for generating and analyzing testable hypotheses. Likewise Pelton & van Manen (1996) outline benefits of long-term monitoring including; hypothesis testing, the development of research techniques, long-term observations (population studies), & technology transfer and practical applications. The BSMIP has incorporated such elements in its approach, so that it is at the forefront of conservation management; e.g. trophic-level interrelationships and providing key monitoring programmes such as Tree Wētā house occupancy as a relationship to stoat and rat reduction.

Monitoring long-term population change is an integral part of effective conservation research and management as it provides the information necessary to identify conservation problems at an early stage and to suggest possible solutions (Goldsmith 1991; Thomas 1996). In order to be effective, population monitoring programmes must provide efficient and reliable estimates of population change. A successful design for long-term monitoring will be due to specific technical considerations (Skalski 1990), such as reliable estimates of abundance or population change. These and other technical considerations, relate directly to the appropriate allocation of resources in both time and space. The monitoring programmes at BSMIP differ in their design, and as such have varying characteristics; *i.e*. depth of inquiry (individual species, groups of individuals, habitat types, *etc*), or their expected the length of time till statistically robust levels are achieved, *etc*.

3.6.7 Analysis

Ribic & Ganio (1996) considers that the statistical model chosen is of critical importance for the data analysis. This primarily reflects the (monitoring) study design, thus in essence; the design of the monitoring programme should determine the *apropos* analysis. It is important to consider the results of an analysis, as negative (non-significant) results from statistical tests that have poor statistical power can only be considered to be inconclusive (Hayes 1987), as is the case of many biological studies with small sample sizes. Berger & Berry (1988) states that the acknowledgement of the role of subjectivity in the interpretation of data could open the way for more accurate and flexible statistical judgements, this would include the identification of a hypothesis. Inferences based on *a priori* considerations, should be clearly separated from those resulting from some form of data dredging (Anderson 2000; Burnham & Anderson 2001). It is thus important to establish a question, or hypotheses at the outset (Green 1994), such as the consideration of what is under investigation?, what are the objectives?; change due to management?, change over time?, and what would be the appropriate analysis, such as a repeated measures model.

The Type I error is the conclusion that an impact has occurred when, in fact one hasn't, whereas a Type II error is the conclusion that there is no impact when there is one. There exists a traditional premise that Type I errors cost more than Type II errors, and thus power is ignored in favour of significance (Toft & Shea 1983; Peterman 1990b; Fairweather 1991), with Type II errors being possibly more insidious, as the incentive for detecting Type II error is lower (Toft & Shea 1983). Power calculations are as important as significance calculations, though as with significance calculations, their value depends on how they are used (Greenwood 1993). McBride *et al.* (1993) states that significance tests do not extract the maximum information from (environmental) data, and can lead to misleading conclusions, mainly due to the fact that a significant result can be reached by collecting enough samples. Likewise, Johnson (1999) states that statistical hypothesis (significance) tests can add limited value to the products of research, and that whether any or all of the results are statistically significant is irrelevant. McBride *et al.* (1993) further states that a statistically significant result does not necessarily imply a biologically or practically significant result, and recommends managers and scientists pay more attention to statistical

power, and decide on what is a practical difference. It is necessary to establish *a priori* parameters; an in the case of the alpha level, it is the prior expression of the alpha range to that we are willing to allow, that identifies the trade-off between Type I (α) & Type II (β) errors. These errors are mutually dependent (Mapstone 1995), which should be addressed (Hinds 1984) and balanced (Fairweather 1991) in any restoration effort, and monitoring design. For monitoring it is a clarified combination of significance and power that is important.

Underwood (1997) states that either Type I or Type II errors can happen because samples in the affected treatment are not perfect measures of what is really happening. So using the precautionary principle inherent in conservation management, we should require that Type II errors should be prevented, as it is rational not to miss real impacts (Underwood 1997; Brown & Miller 1998). Cohen (1962; 1994); Sedlmeier & Gigerenzer (1989) take the analytical view that; given that the null hypothesis is always false, the rate of type I errors is 0%, not 5%, and that only Type II errors can be made, which run typically at about 50%. Balancing the relative costs of the real-life consequences of these two errors; (a key basis for monitoring), Type II errors are more paramount, when a decision would result in the loss of unique habitats or species.

A practical consideration in monitoring, is the use of repeated measurements (Eckblad 1991). Link *et al.* (1994) states that most large-scale surveys of animal populations are based on counts of individuals observed during a sampling period, which are used as indices, which not only reflects variability in population sizes among sites but also variability due to the in-exactness of the counts. Link *et al.* (1994) goes further to state that repeated measures (repeated counts at survey sites) can be used to document this additional source of variability and, in some cases mitigate its effects. Repeated measures are appropriate for environmental impact and monitoring studies (Green 1993 Norton 1996; Ottenbacher 1996), and are useful for hard to capture animals such as lizards (Moseby & Reed 2001). Underwood (1993) states that asymmetrical analyses of variance derived from repeated measures models can be used to detect many types of impact that are not identifiable using widely recommended BACI sampling, such asymmetrical, beyond BACI designs are also more logical because of spatial replication (Underwood 1993).

Ribic & Ganio (1996) state that where the same site is measured over time, the simple analysis of variance model is not appropriate, because of autocorrelation between time points". For a given sample size, using the correct model (the repeated measures model) results in greater power than the incorrect analysis of variance model (Ribic & Ganio 1996). Repeated measures analysis is the most appropriate general statistical analysis for the BSMIP data, as the main monitoring programmes have spatially established plot, point, and transects, which are monitored at approximately the same time each year.

Carver (1978) suggests a return to the scientific method of examining data and replicating results rather than a reliance on statistical significance testing to provide equivalent information. The expression of a statistical result needs to be explained as best as it possibly can, in order that a valid, and biological relevant conclusion can be supported. This can be approached by the declaration of all the determined statistics of an analysis, such as statistical power, statistical significance, the effect size, and the variation inherent in the sampled population of interest. Likewise Johnson (1999), states that statistical significance of results from hypothesis tests are irrelevant, and that the key consideration is whether the effect size of a magnitude judged to be important has been consistently obtained across valid replications. McBride *et al.* (1993) goes further to state that significance tests do not extract the maximum information from environmental data, and can lead to misleading conclusions through gaining a significant result gained simply by collecting enough samples, without conferring a biologically or practically significant result, and recommends that environmental managers and scientists pay more attention to statistical power, and decide on what is a practical difference. Bernstein & Zalinski (1983) state that the determination of statistical significance will correspond more closely to biological significance, with the incorporation of variability between sampling locations over time, and the instantaneous replicate variability into the error term of the analysis. Cohen (1994) reminds researchers (and environmental managers, *etc*) that the null hypothesis is always false, the rate of type I errors is 0%, not 5%, and that only Type II errors can be made. Which run typically at about 50%. To confirm a difference or otherwise, which is biologically relevant to the question of concern, it would be only professional to make explicit all the pertinent statistics that may convey knowledge. Statistical analysis of monitoring results should include the expression of the *a priori* statistical power (of the monitoring design), the statistical significance and/or

CI (Confidence Interval), the effect size or change in values, and the CV (Coefficient of Variation). The determination of the BSMIP monitoring programmes' statistical reliability was a key focus of this thesis, based in part on the preliminary 'prospective power analysis' of the monitoring programmes.

Chapter 4 Case Study of BSMIP Monitoring Programmes

4.1 Use of Computation Power, and MONITOR Computer Programme

The rise of new statistical theory and methods during the 1980s was founded on the growth in the power of electronic computation (Efron & Tibshirani 1991). Such methods include bootstrap methods, generalized additive models, and the large-scale simulation of data to determine how models perform. The computer estimation of statistical power can be performed utilizing such simulation of data to determine how power varies in response to a variety of criteria, such as sample size, repetitions of counts, *etc*. Calculations of statistical power have mainly concentrated on fish stocks and fishery-affected species (Edwards & Perkins 1992; Taylor & Gerrodette 1993; Gryska *et al.* 1997; Maxwell 1999; Wilson *et al.*1999; Lindley *et al.* 2000), bird populations (Taylor & Gerrodette 1993; Benn *et al.* 1996; Morrison *et al*. 1996; Gibbs & Melvin 1996; Bishop *et al.* 2000), and track surveys of predators (Beier & Cunningham 1996; Hatfield 1996; Zielinski & Stauffer 1996; Steidl *et al.* 1997; Brown & Miller 1998) see Appendix 1. This thesis expands on these studies as it investigates a range of monitoring programmes and methodologies currently used by the Department of Conservation, and specifically for an ecosystem restoration project.

The approximate power of the monitoring programmes was calculated using the freeware computer program MONITOR (monitor.exe: Gibbs 1995; Gibbs & Melvin 1997: available from the United States Geological Survey website (http://www.mp1-pwrc.usgs.gov/powcase/monitor.html.). MONITOR is a specialized power analysis computer package (software) that calculates statistical power using Monte Carlo simulations (Thomas & Krebs 1997) on linear regression analyses (n = up to 10000 simulations for each power estimate). Gibbs & Melvin (1997) used MONITOR to evaluate variously configured monitoring programmes for waterbird call-response surveys. This same programme was used by the United States Geographic Service (USGS) to determine sample sizes for the North America Amphibian survey, and is currently being used to design a monitoring programme for North American mushroom populations (Droege 1999). MONITOR was also used by Bishop *et al.* (2000) to determine the number of years of survey effort needed to detect a 10% change in the numbers of migrant shorebirds of Western Sandpiper (*Calidris*

mauri), and Dunlin (*Calidris alpina pacifica*) on the Copper River Delta, Alaska. MONITOR had internal limits of operation of; up to 10000 repetitions for each simulation, and a (0-10%) change in population/abundance/activity *etc.* The possible Effect Size (ES) or level of changes (+/– 10%) registered were considered to be a conservative level of change, in line with Steidl *et al.* (1997) recommendation that minimum biologically significant effect sizes be used for all Power analyses. This conservative ES used evaluated in this thesis encompasses what is needed for a reliable monitoring programme.

4.2 Method of Statistical Reliability Analysis

The statistical power analyses were conducted using MONITOR (Gibbs 1995) computer programme. The individual sampling methodology (monitoring design) is specific for each monitoring programme, and are given in Appendix 2, and 3. MONITOR required the monitoring programme design parameters, of both the monitoring design (Plots) including the initial monitoring values (generally from the initial 1996 values), mean, and standard deviation (Table 4.2.1), the sampling effort (Surveys), and the simulation of trend variation and linear regression analysis (trend type) for the calculation of statistical power.

Table 4.2.1 MONITOR Computer Programme: Field Parameters

Plots	Surveys	Trends
Number Monitored	Number Conducted	Type
Counts/Plot/Survey	Occasion	Significance Level
Initial Values		No. of Tails
		Constant Added
		Trend Variation
		Rounding
		Trend Coverage

Simulations were run for each of the different levels of inquiry; Boundary Stream Scenic Reserve (Treatment site), Cashes and Thomas Bush Scenic Reserves (Combined Non-Treatment sites), individual Non-Treatment sites, and vegetation types according to each monitoring programme. The simulations were performed containing the following criteria;

alpha levels (0.05, 0.1, 0.2, 0.25), and monitoring occasions (3, 4, 5, 6, 7, 8, 9, 10, 15, 20, 50, 100). These monitoring occasions were specific to each monitoring programme, and did not necessarily have the same temporal scale (Appendix 2, & 3). The combination of the monitoring occasion's data produces a power curve as below; for the wētā monitoring programme within the Treatment site (Boundary Stream), with positive change, an alpha level of 0.10.

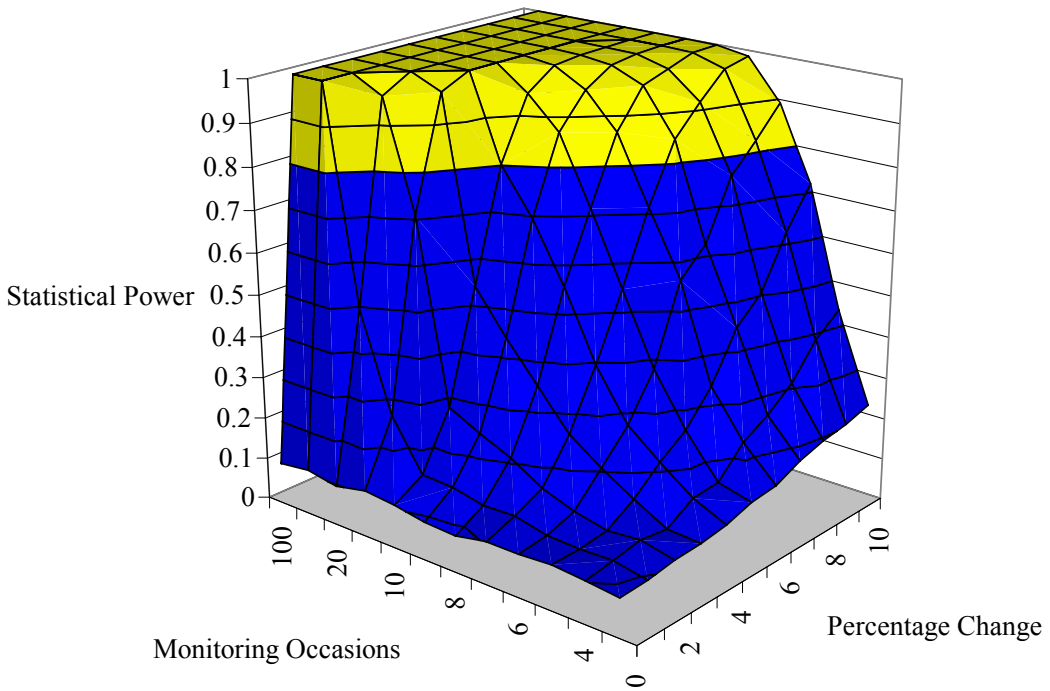

Figure 4.2.1 Statistical Power Curve for Boundary Stream Wētā Monitoring Programme

Three dimensional graph showing the calculated statistical power of the monitoring programme over time (monitoring occasions), and the positive percentage change of the wētā numbers (assuming the previous conditions; normality of distribution, *etc*).

For the wētā monitoring programme, from year three (monitoring occasion: six) onwards, the programme is robust (statistically powerful) enough to detect at least a 10% increase in wētā roost occupancy numbers. For the analyses, I assumed that a power >0.80 was

sufficient to detect a population (index) change (Rotenberry & Wiens 1985; Cohen 1988). This can be seen for the wētā, with the yellow area of the graph representing data and their respective criteria that has a statistical power of 0.80 or above. The (blue) area indicates that little confidence should be placed in a conclusion based on a failure to reject a hypothesis of no change; *Ho* (Peterman 1990b) should the parameters of an analysis fall into this area.

The analysis of the monitoring programmes produced over 1200 power curves (with over 140 separate statistical power calculations for each curve: Table 4.2.2), each one distinct according to a range of criteria including; the monitoring programme design, level of inquiry (Treatment Regime, Scenic Reserve, habitat or vegetation type), and sampling effort. The specific trend characteristics were kept constant e.g.; no. of tails (two), constant added (1), trend variation (0), and rounding (decimal) for each simulation. The expected timeframes for each monitoring programme to reach a robust level are given in Appendix 3, and 4.

Table 4.2.2 Power Curves for Monitoring Programmes

Monitoring Programme	Number of Power Curves
Wētā	44
Invertebrates	162
Vegetation	90
Lizards	60
Birds	360
Rodents	216
Mustelids	180
Possums	88

4.3 Monitoring Programmes

4.3.1 Wētā

4.3.1.1 Wētā Monitoring – General

Tree wētā (*Hemideina* spp.) are an important component of New Zealand forest ecosystem, and are used as invertebrate indicator species in many restoration programmes (Gibbs 1998; Spurr & Drew 1999; Trewick & Morgan-Richards 2000; Christensen 2001). Due to the fact that tree wētā live in aggregations (Ordish 1992), and roost in holes, tunnels, or similar confined dark recesses such as artificial shelters (Ordish 1992; Trewick & Morgan-Richards 1995; Townsend *et al.* 1997), the use of wētā roosts appear to be appropriate devices for monitoring wētā (Trewick & Morgan-Richards 2000, Christensen 2003). The single cavity wētā roosts at the BSMIP, have already determined new and the most northern records of species such as the Hawkes Bay tree wētā (*Hemideina trewicki*) (Christensen 2002a).

4.3.1.2 Wētā Monitoring Programme – Guiding Objectives

Monitoring of the tree wētā within the BSMIP, acts as a direct outcome measure for the management of arboreal predators, such as rats, and stoats. Tree wētā are largely vegetarians (Gibbs 1998), so the tree wētā monitoring programme could additionally act as a secondary or covariate outcome monitoring programme with the vegetation monitoring (alongside phenology, and FBI (Foliar Browse Index) monitoring). It additionally acts as a comparative measure of the response of the arboreal invertebrate assemblage compared with the terrestrial invertebrate response to the intensive multi-pest species management. This arboreal/terrestrial invertebrate predation interaction was suggested by Moller (1985), though as of yet few studies have attempted to detail such a response.

Tree wētā monitoring also provides response data to the intensive multi-pest species management for consumers such as birds. Tree wētā, being large nocturnally active invertebrates, are an important prey item for kiwi, and morepork (Heather & Robertson 1996). Stephenson (1998), Haw *et al.* (2001) both found over 98% of morepork pellets contained insect material, and that wētā formed a major component of this material. Other birds such as tomtits, and robins feed on wētā, and it is known that blackbirds feed mainly

on the forest floor (Heather & Robertson 1996), though it is unknown to what effect blackbirds are having on wētā. The abundance of such prey items such as tree wētā for birds, is an important criterion that the tree wētā monitoring programme can provide, to determine the overall interrelationship between managed pests and the response of the indigenous biota.

4.3.1.3 Wētā Monitoring Programme – Biological Relevance

The tree wētā monitoring programme at the BSMIP has clear interrelationships with other component programmes of the overall BSMIP Monitoring Programme, built on as well as adding too established biological knowledge. Wētā are preyed on by range of mammalian animals; Rats: Daniel (1973); Innes (1979; 1990; 2001); Clout (1980); Rickard (1996); Blackwell (2000); Cats: Fitzgerald (1990); mice: Fitzgerald (1996); Ruscoe (2001); stoats: King & Moody (1982); Murphy & Dowding (1995); Hedgehogs: Berry (1999); Hendra (1999). Moller (1985) states that the number and size of natural wētā galleries and the presence or absence of predators may be important determinants of tree wētā ecology such as; population density, sex ratio, harem formation, and ground activity. Rufaut (1995) found that tree wētā were vulnerable to rodent predation, existed in significantly lower densities, and occupied refuges with significantly smaller entrance holes in comparison to tree weta on rodent-free islands. Miller & Miller (1995) found that both rats and mice predominantly preyed on wētā (mainly *Hemideina thoracica*: the Auckland Tree wētā) on Rangitoto island. King (1990), states that stoats take insects (mostly wētā of *Hemideina*, *Hemiandrus*, and *Gymnoplectron* spp.). Miles *et al.* (1997) found that tree wētā were the highest percentage (24%) prey item for stoats, in Tongariro National Park. Clearly rodents (Gibbs 1998), and mustelids (stoats) have had a great impact on wētā affecting not only their abundance, but also their behavioural ecology.

4.3.1.4 Wētā Monitoring Programme – Statistical Reliability

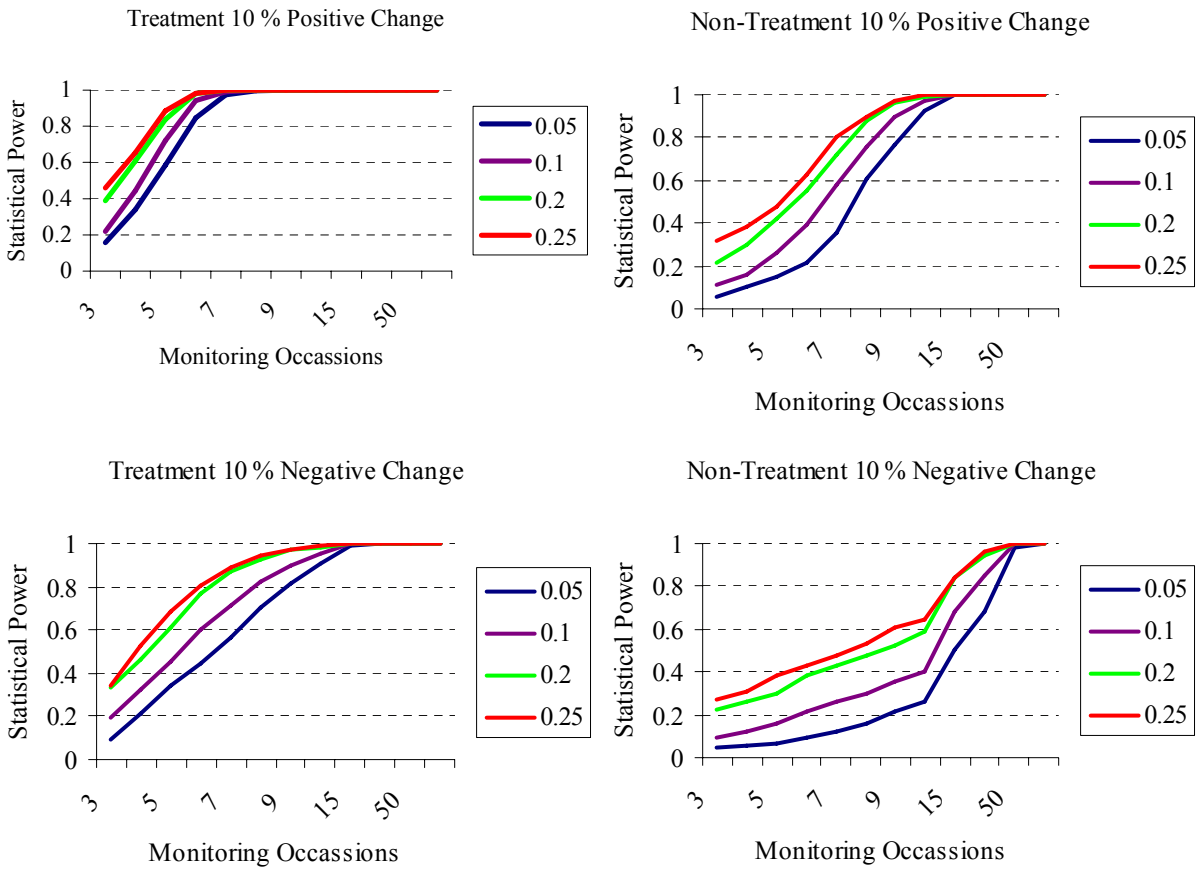

Figure 4.3.1.4.1 Statistical Power for Wētā Monitoring Programme: All Wētā

The calculated statistical power values for the Tree Wētā monitoring programme show a robust design (from initial figures) for a 10% positive change, with both the Treatment site, and (combined) Non-Treatment sites reaching a calculated statistical power of 1.00 for all alpha levels. The expected timeframe for the wētā monitoring programme (with houses monitored twice each summer) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.1) level equates in this case to two and ½ years for the Treatment site, and four and ½ years for the (combined) Non-Treatment sites, for the determination of a positive change.

The tree wētā monitoring programme design was found to be robust for wētā overall (tree, ground, and cave wētā numbers collectively), and tree wētā numbers specifically, though decreased in its statistical reliability (power), as the level of enquiry became more specific to tree wētā species (Appendix: 3-6). Simulations for each significance criterion or alpha level, α (0.05, 0.1, 0.2, 0.25) reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence.

The Non-Treatment sites had almost twice as long a timeframe to reach a robust' level than the Treatment site for the determination of a positive change. This is due primarily to the smaller number of artificial wētā roosts in the Non-Treatment sites (4 lines of twenty roosts) relative to the Treatment site (5 lines of twenty roosts); and secondarily to the smaller initial index values of the Non-Treatment sites relative to that of the Treatment site. In the monitoring design sense, the numbers of lines relates directly to the sample size for power analysis. Cohen (1988; 1992) states that the sample size (N) increases with an increase in the desired statistical power for research planning, and additionally reduces the confidence interval (Cohen 1994). As there would be a saturation point of tree wētā numbers occupying the wētā roosts (Christensen 2003), it would be important to increase the sample size of the monitoring design (i.e. the wētā roosts), and specifically within the Non-Treatment sites if the sample size was intended to be increased. This would increase the statistical power of the tree wētā monitoring programme, and additionally reduce the timeframe to determine changes. Repeat surveys on either side of the two current summer monitoring surveys would also increase the statistical power.

The tree wētā roosts are fixed spatially because they are nailed to trees, and are not moved between monitoring occasions. The model selection for the analysis (e.g. Analysis of Variance) over time of the occupancy of the wētā houses requires careful consideration so as to avoid confounding issues such as the over or under estimate of the time or spatial factors (Buckland *et al*. 1997). Repeated measures analysis is the correct model for use when measurements are made at the same site at different times, though still requires the parameter estimates to be reliable (Ribic & Ganio 1996). Millard et al. (1985), and Zar (1999) state that MANOVA *multivariate analysis of variance*, or multivariate time series

analysis procedures would be appropriate (repeated-measures) statistical techniques in this case. Repeated measures analysis is appropriate for monitoring studies, as they have the practical benefit of being more efficient (fewer samples are used to achieve the same statistical power) than a (randomized) design where sites are reallocated after the treatment or impact (Green 1984; Green 1989; Green 1993; Underwood 1993; Norton 1996; Ribic & Ganio 1996).

4.3.1.5 Wētā Monitoring Programme – Conclusion

The BSMIP tree wētā monitoring programme is effective, balanced, and will fulfil the important need for long-term trend monitoring of this group of invertebrates. It encompasses comparative Non-Treatment sites, and has a valid cause ("results" of management; such as rodent, mustelid, and possum monitoring) and effect ("outcome" of tree wētā roost occupancy) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects. The tree wētā monitoring programme adds important supporting or covariate information to the broader invertebrate monitoring programme (relative changes in the arboreal and terrestrial invertebrate assemblages), vegetation monitoring programme (composition, and phenology health of the forests), and bird monitoring programme (changes in the relative abundance of insectivorous birds, such as robins).

The tree wētā monitoring programme design was found to be robust, and reliable to detect a trend (+/- 10%) for a moderate level of confidence (α: 0.25, or 75% CI) upon six years of its inception. I consider this to be a biologically responsive timeframe for tree wētā, as it is approximately similar to an adult wētā lifespan (Gibbs 1998; Jamieson *et al*. 2000). While unfortunate that no relative "Before" measure was made on the tree wētā roost occupancy prior to the initial management in 1996, since the management is ongoing the comparative difference between the Treatment and Non-treatment sites would provide a valid and effective measure of outcome change due to management. With the appropriate statistical analysis, the relative changes over time can be measured between the different management (Treatment) regimes, and habitat-types, of the different wētā roost occupants (Appendix: 3-6). The statistical analysis on the tree wētā data requires consideration of space-time

correlation (Millard *et al*. 1985; Buckland *et al*. 1997), which is best addressed by repeated measures analysis (Green 1984; Green 1989; Green 1993; Underwood 1993; Norton 1996; Ribic & Ganio 1996).

4.3.1.6 Wētā Monitoring Programme - Recommendations

Recommendations include an extra line in each of the Non-Treatment sites, or other local Non-Treatment reserves, and an increase of at least one house per group (i.e. 5 roosts per group), which will be useful as replacement of wētā roosts will need to occur in time. This would increase the statistical power of the monitoring programme, increase the precision of measurements per habitat-type, and enable finer scale analysis such as other species numbers and individual occupancy rates of the wētā roosts. If it is possible, these new lines should be placed in similar vegetation habitats to that of the Treatment site, so as to enable direct habitat comparisons (as recommended by Brown & Norton (2001)) to be made for the occupancy of artificial roost occupancy by tree wētā. These roosts will require an establishment period (Trewick & Morgan-Richards 2000), as well as a weathering time period (esp. if macrocarpa wood is used as it contains a natural insecticide (Steve Trewick *Pers Comm* 2001)). While the addition of extra wētā roosts is not a necessity for the overall design, it will strengthen investigation of records such as the relatively rare (recently located) Hawkes Bay tree wētā in the area (Christensen 2002). Repeated measures analysis should be used for the BSMIP tree wētā monitoring programme, as the artificial wētā roosts are fixed to trees, and so the measurements are 'repeated' spatially in any analysis over time.

4.3.2 Ground Invertebrates

4.3.2.1 Ground Invertebrate Monitoring Programme – General

There is a growing awareness of the importance of insects as collective "keystone" groups of species, as well as indicators of environmental change and biodiversity (Williams 1993; Saunders 1994; Norton 1996; Fisher 1998). While pitfall traps were not designed to accurately estimate population densities (Hansen 1988) and specifically target the more active ground-based species (Green 1996), they are a convenient method of detecting relative abundance, and species diversity of ground insects (Greenslade 1964; Oliver & Beattie 1996; Watts & Gibbs 2000). Pitfall trapping has been used extensively for invertebrate diversity studies in New Zealand (Crosby 1992; Townsend 1996; Crisp *et al.* 1998; Watts & Gibbs 2000). This combination of the importance of invertebrates within the ecosystem, the convenience of the method, and the ability to detect relative abundance, makes pitfall trapping excellent value for monitoring specific biodiversity changes in relation to conservation management. This includes the comparison and definition of habitats for a complete range of parameters using species assemblages (Hutcheson *et al.* 1999), or for determining the relationship between indigenous beetles and indigenous plants (Crisp *et al.* 1998).

4.3.2.2 Ground Invertebrate Monitoring Programme – Guiding Objectives

Monitoring of the ground invertebrates within the BSMIP, acts as a direct outcome measure for the management of the key terrestrial invertebrate predators, such as rats, and stoats. As mentioned in above in the tree wētā monitoring conclusion, it additionally acts as a comparative measure of the response of the terrestrial invertebrate assemblage compared with the arboreal response to the intensive multi-pest species management. The ground invertebrate monitoring programme also provides response data from the changes of abundance of the indigenous (and non-managed) consumers such as birds, and lizards. Numerous bird species feed on ground invertebrates, such as morepork, robins, tomtits, bellbirds, tui, kiwi, and it is known that blackbirds feed mainly on the forest floor (Heather & Robertson 1996), though it is unknown to what effect blackbirds are having on ground invertebrates. The availability, and abundance of such food items for birds such as ground

invertebrates, are important criteria that the ground invertebrate monitoring programme can provide, to determine the overall interrelationship between managed pests and the response of the indigenous biota.

4.3.2.3 Ground Invertebrate Monitoring Programme – Biological Relevance

As with the tree wētā monitoring programme, the ground invertebrate monitoring programme at the BSMIP has strong interrelationships with other component programmes of the overall BSMIP. Fisher (1998) states that invertebrates (insects) provide essential ecosystem processes for the long-term survival of populations and species assemblages in preserved landscapes, acting as "keystone" species, and assemblages. Ground invertebrates in New Zealand are preyed upon directly by a wide range of pest animals; mice (Murphy & Pickard 1990; Ruscoe 2001), rats (Innes 1990; Moors 1990), hedgehogs (Brockie 1990), possums (Cowan 1990; Sadlier 2000), weasels (King 1990a; King *et al*. 2001), stoats (King 1990; King *et al*. 2001), ferrets (Lavers & Clapperton 1990), cats (Fitzgerald 1990; Gillies 2001), and pigs (McIlroy 1990; McIlroy 2001). Vertebrate mammals have had a large impact on ground invertebrates, affecting species composition, and abundance. Hutcheson (1999) found that beetle diversity was higher after animal control management occurred in the Mapara forest, and that insect biodiversity is associated with resource availability and habitat processes. Alley (*et al.* 2001) found that house mouse population "eruptions" in southern beech (*Nothofagus* spp.) forests after mast seedlings were triggered by increases in the populations of some arthropods, especially Lepidoptera larvae and spiders. Wardle *et al.* (2001) found that all micro-arthropod and macro-faunal groups were consistently adversely affected by browsing on vegetation by mammals in indigenous New Zealand forests, irrespective of trophic position. The diversity and scale of interactions between the ground invertebrate fauna of New Zealand's indigenous forests and introduced mammals is vast, and hugely complex, though the outlining of key parameters (such as predator-prey relationships) are discernible.

4.3.2.4 Ground Invertebrate Monitoring Programme – Statistical Reliability

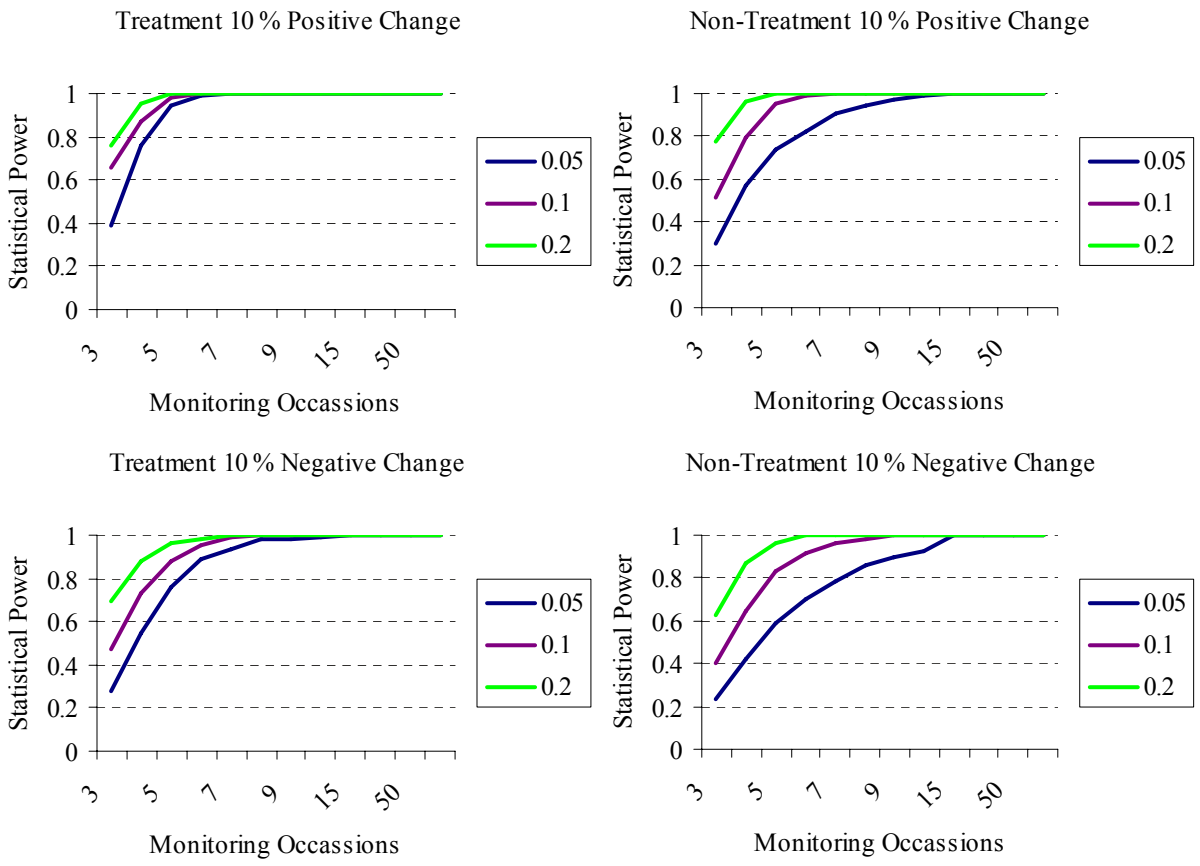

Figure 4.3.2.4.1 Statistical Power for Invertebrate Monitoring Programme: Total Numbers

The calculated statistical power values for the Invertebrate monitoring programme (Total Numbers) show a very robust design (from initial figures) for a 10% positive change, with both the Treatment site, and (combined) Non-Treatment sites reaching a calculated statistical power of 1.00 for all alpha levels. The expected timeframe for the invertebrate monitoring programme (total numbers: with pitfall traps monitored once each summer) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to four years for both the Treatment site, and the (combined) Non-Treatment sites for the determination of a positive change.

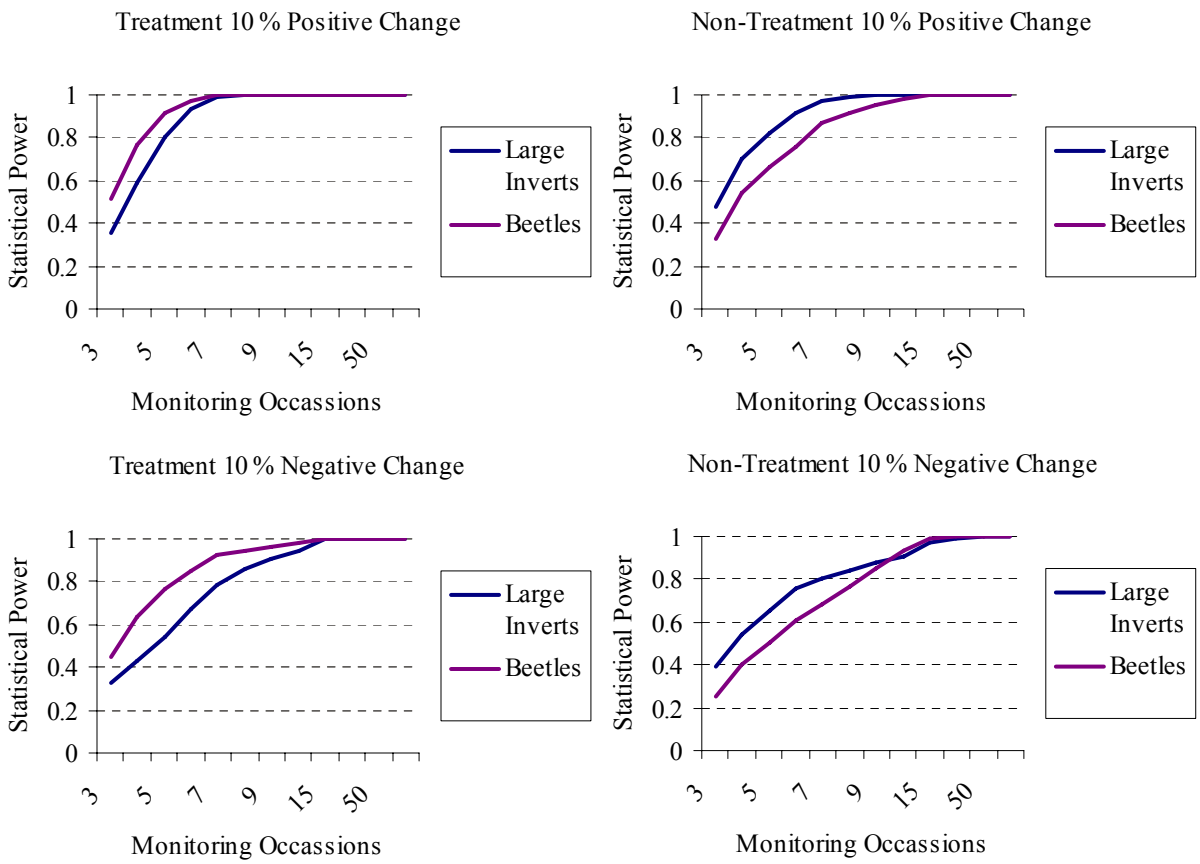

Figure 4.3.2.4.2 Statistical Power for Invertebrate Monitoring Programme: Species Assemblages

The calculated statistical power values for the Invertebrate monitoring programme (for two selected Species Assemblages; 'Large' (3cm+) Invertebrates, and Beetles) show a very robust design (from initial figures) for a 10% change, with both the Treatment site, and (combined) Non-Treatment sites reaching a calculated statistical power of 1.00 for all alpha levels. The expected timeframe for the invertebrate monitoring programme (with pitfall traps monitored once each summer) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to five years for 'Large' invertebrates for both the Treatment site, and the (combined) Non-Treatment sites, for the determination of a positive change. For 'beetles' the expected timeframe for each monitoring programme to reach a robust level equates to and five years for the Treatment site, and seven years for the (combined) Non-Treatment sites, for the determination of a positive change.

The ground invertebrate monitoring programme design was found to be highly robust for the overall invertebrate numbers, and had limited decrease in its statistical reliability (power), as the level of enquiry became more specific to the species assemblages of the 'Large invertebrates', and 'Beetles' (Appendix: 3-6). Simulations for the three significance criterion or alpha levels, α (0.05, 0.1, 0.2) reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence.

A "Before" measure was made on the total numbers of ground invertebrates prior to the initial management in 1996, though since the management is ongoing, and the current monitoring design is now more precise (e.g. looking at size classes) the comparative difference in trends between the Treatment and Non-treatment sites would provide a more useful measure of outcome change due to management. Similar to the tree wētā monitoring technique, the pitfall traps are not moved between monitoring occasions (i.e. are fixed spatially). Thus the statistical analysis on the ground invertebrate data requires due consideration of space-time correlation (Millard *et al*. 1985; Buckland *et al*. 1997), which is best addressed by repeated measures analysis as recommended by a number of authors, e.g. Green (1984; 1989; 1993); Underwood (1993); Norton (1996); Ribic & Ganio (1996).

4.3.2.5 Ground Invertebrate Monitoring Programme – Conclusion

Conservation management of the indigenous invertebrate element is essential for enhancing landscape biodiversity, given that invertebrates provide the bulk of the biodiversity in New Zealand (Keesing & Wratten 1998). Like the tree wētā monitoring programme, the BSMIP ground invertebrate monitoring programme is effective, balanced, and will fulfil the important need for long-term trend monitoring of this group of invertebrates. It encompasses comparative Non-Treatment sites, and has a valid cause ("results" of management; such as rodent, mustelid, and possum monitoring) and effect ("outcome" of pitfall trap sample composition, and abundance) basis monitoring of management. This fulfils Arand & Stephens (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects.

The ground invertebrate monitoring programme design was found to be robust, and reliable to detect a trend (+/- 10%) for a moderate level of confidence (α: 0.25, or 75% CI) upon ten years of its inception. I consider this to encompass a biologically responsive timeframe for ground invertebrates, as previous studies on ground invertebrate response to animal control (e.g. Hutcheson (1999)) have found significant differences in similar timeframes with less monitoring occasions. The monitoring design while robust, is only taking a "snap-shot" albeit a destructive one, of the ground invertebrate's summer activity.

The ground invertebrate monitoring programme, along with the tree wētā monitoring programme adds important supporting or covariate information to the broader invertebrate monitoring programme (relative changes in the arboreal and terrestrial invertebrate assemblages), and bird monitoring programme (changes in the relative abundance of insectivorous birds, such as robins). While limited "Before" or pre-management monitoring has occurred, the appropriate statistical analysis can measure relative changes over time between the different management (Treatment) regimes, and habitat-types, of the pitfall trap contents.

4.3.2.6 Ground Invertebrate Monitoring Programme – Recommendations

It is recommended that the ground invertebrate monitoring programme is kept in its current form. A method of analysis including a "repeated measures" protocol should be used to statistically examine the ground invertebrate data.

4.3.3 Lizards

4.3.3.1 Lizard Monitoring – General

New Zealand has a great ecological diversity of geckos (Gekkonidae), and skinks (Scincidae) (Towns 1994; Daugherty 1990; Towns *et al.* 2002). The majority of *Cyclodina* skinks (Towns 1994), and at least half of the North Island *Oligosoma* skink species appear to be particularly susceptible to predation by pest mammals (Towns *et al.* 2002). Moseby & Read (2001) state that pitfall trapping of small terrestrial vertebrates is widely used for surveys, research projects, and impact assessments. They found that the number of trap sites (1, 3, 5, 10) trapped 30%, 55%, 65%, & 73% of the reptiles in that habitat (arid South Australia), clearly indicating the greater reliability of a more comprehensive sampling design. Pitfall trapping has been used in New Zealand for the capturing of Whitaker's skinks (*Cyclodina whitakeri*) for translocation to a rodent-free island in the Mercury islands (Towns 1994), the comparative study of two *Oligosoma* spp. skinks in Canterbury (Freeman 1997), and for surveying Big Bay skink (*Oligosoma* sp.) in the west coast (Tocher 1999).

To date, the BSMIP lizard (skink) pitfall trap monitoring programme has only captured common skink (*Oligosoma nigriplantare polychroma*) though they may be one of the more higher altitude records 1000m+ asl. found thus far. Enge (2001) found that lizards were captured more frequently than expected in funnel traps than in pitfall traps. It may be that another method would be more suitable, or a blend of one or two methods, such as tin shelters over a number of pitfall traps. The shelter acts as a funnel for skinks, and the pitfall traps as reliable capture equipment would provide a coefficient of variance (CV) for that sample unit (i.e. the shelter).

4.3.3.2 Lizard Monitoring Programme – Guiding Objectives

Cyclodina spp skinks predominantly inhabit forests (Gill & Whitaker 1996) and are not known in the BSMIP area, whereas *Oligosoma* spp. have a broader habitat range, and often favour grassy habitats. Benn (1995) states that broad surveys are often not statistically robust. The BSMIP lizard (skink) monitoring programme initially acted as a survey tool, to look for both *Oligosoma* spp., and *Cyclodina* spp. and while useful in that regard, has limited design use as an ongoing monitoring programme. Ecosystems are dynamic and in

constant change, as variability can increase with surveys over time (Pimm & Redfearn 1988; Kareiva & Bergelson 1997), it becomes very important to establish a solid robust targeted monitoring programme. A key method of doing this would be to reduce the variability inherent in surveys, and target the ecology of the species investigated. In the BSMIP Lizard (skink) monitoring programme, this would be to target the common skink (*Oligosoma nigriplantare polychroma*) known in the grassy areas about the Maungaharuru range. Consideration should be given to a cluster (grouped traps) sampling methodology and comparative plots in Non-Treatment sites, so as to measure the response of lizards to management. Ryan *et al.* (2002) tested three different census techniques (time-constrained searches, coverboards "shelters', and drift fences with pitfall and funnel traps), and found that while pitfall and funnel traps were highly effective, the coverboards contributed to measures of abundance and revealed species not detected by the other techniques. The use of shelters may be a more reliable and valid method for 'monitoring' skinks over the long-term, as they would be more closely linked to the ecology of the species than traps. Freeman (1997) used baited pitfall traps throughout the summer months of November to March, when lizards were most active. Lengthening the monitoring period from one month (when the pitfalls were open) of the BSMIP lizard (skink) monitoring programme would gain a more reliable estimate of the skink activity.

4.3.3.3 Lizard Monitoring Programme – Biological Relevance

The lizard (skink) monitoring programme at the BSMIP has clear interrelationships with other component programmes of the overall BSMIP Monitoring Programme, built on as well as adding too established biological knowledge. As lizards (skinks) are taken as food by a variety of pest mammals: rats: Innes (1990); cats: Fitzgerald (1990); hedgehogs: Moss & Sanders (2001); stoats: King (1990); King *et al.* (2001), they act as an excellent indicator of the response of an indigenous ecosystem to restoration management. Towns (1994) found that removal of rats (kiore: *Rattus exulans*) increased Whitaker's skink numbers within 12 months and rose 30 fold over 5 years at coastal sites, though measurable increases of lizard numbers in forest areas took up to six years. Demonstrating that predation rather than habitat deficiencies were responsible for the depleted lizard stocks. This was an excellent monitoring design with both "Before" and "After" rodent control monitoring, along with

trend monitoring. Towns (1994) notes, that previous measurement of effects on the lizards would be based on circumstantial comparisons. Further analysis of the Mercury Island data on shore skinks (*Oligosoma smithii*), showed large increases in the capture frequency due to the removal of rats (Towns 1996). Adams (1997) noted a dramatic increase in skinks after the eradication of Norway rats (*Rattus norvegicus*) from Motu-o-kura. Similarly, Owen (1997) states that Speckled skink (*Oligosoma infrapunctatum*) were discovered on Mokoia island after the removal of rats. One of two weasels caught at Pukerua Bay, Wellington had a stomach crammed with pieces of skink, being over 70% of the stoat's stomach content (Miskelly 1997). Monitoring of the lizards (skinks) within the BSMIP, acts as a direct outcome measure for the management of predators, such as rats, cats, and stoats.

4.3.3.4 Lizard Monitoring Programme – Statistical Reliability

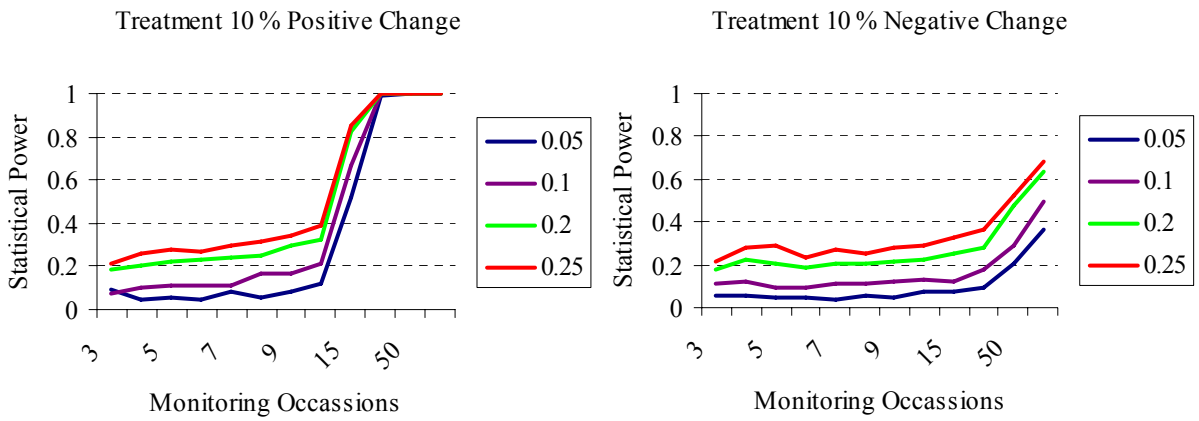

Figure 4.3.3.4.1 Statistical Power for Lizard Monitoring Programme

The calculated statistical power values for the Lizard monitoring programme (pooled lines X 1) show a low design strength (from initial figures) for a 10% positive and negative change in the Treatment site. The monitoring design reaches a calculated statistical power of 1.00 for all alpha levels for a 10% positive change, though does not reach 0.8 (The level of "robustness") for a negative change within 100 monitoring occasions (in this case 100 years). The expected timeframe for the lizard monitoring programme (with pitfall traps monitored once each summer) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.05, 0.1, 0.2, 0.25) level equates in this case to 14, 15, 17, and 18 years respectively for the determination of a positive change.

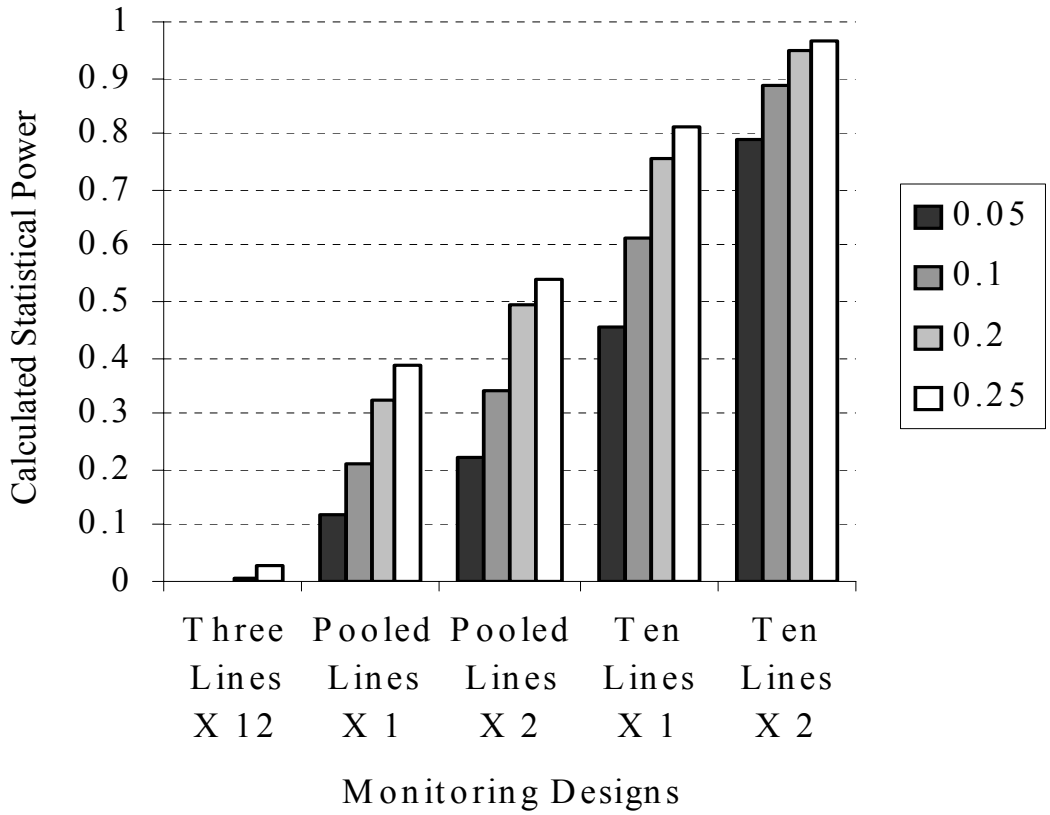

Figure 4.3.3.4.2 Calculated Statistical Power for a Range of Lizard Monitoring Designs

The calculated statistical power values for a range of monitoring designs modelling the initial data with alpha levels (0.05, 0.1, 0.2, 0.25) for a 10% positive change at year ten. Details of each "simulated" monitoring programme are given below;

- Three lines X 12 (each sampling occasion with assumed independence – for each line) equating to 36 sample units, monitored once.
- Pooled Lines X 1 (combined data from the three lines) equating to one sample unit, monitored once.
- Pooled Lines X 2 (combined data from the three lines) equating to one sample unit, monitored twice.

- Ten Lines X 1 (Figures taken from high trap-catch sites – groups of ten traps), monitored once.
- Ten Lines X 2 (Figures taken from high trap-catch sites) monitored twice.

Using the initial data, and modelling for areas of expected/known skinks (High catch): the Ten lines X 2 design, for the lizard monitoring programme approximately doubles the statistical power of the current monitoring design. This design would reduce the number of lizard pitfall traps from 150 traps (Three lines of 50 traps) to 100 traps (ten lines of ten traps).

The lizard monitoring programme design was found to have a low strength to determine lizard numbers. Simulations for the four significance criterion or alpha levels, α (0.05, 0.1, 0.2, 0.25) reached the 'robust' (0.8 Statistical Power) level for a positive 10% change over time (up to 100 monitoring occasions), though only after the 15th monitor (i.e. year 15). The monitoring programme design did not however reach this level for a negative 10% change over time (up to 100 monitoring occasions). This indicates that the monitoring design in its current form is unreliable to determine changes over time. This is due to the small number of transect lines (three) at the Treatment site, the fact there was a very high variability in the initial counts (18, 0, and 1) of lizards between the lines, and that no comparative Non-Treatment transect lines have been established.

4.3.3.5 Lizard Monitoring Programme – Conclusion

The current methodology (pitfall trapping using transect lines) utilized at BSMIP does not approach a robust design relative to the objectives of the Mainland Island Project. It does not have comparative Non-Treatment sites relative to the Treatment sites, and thus limited comparison of the lizard (skink) response to the management actions can be made. With the established biological linkage (rodents and mustelids preying upon lizards), the lizard (skink) monitoring programme this fulfils Arand & Stephens (1999) conservation monitoring guidelines, with a direct linkage between "results" (rodent and mustelid tracking tunnel activity) and "outcomes" (lizard pitfall trap captures) in the measurement of conservation projects. It would be useful to continue with lizard (skink) monitoring,

although changes should be made to the data capture methodology (skink capture), and the monitoring programme's design.

The lizard (skink) monitoring programme design was found to have low "robustness", and unreliable to detect a negative trend (– 10%) for even a moderate level of confidence (α: 0.25, or 75% CI). While it could detect a positive trend (+ 10%) for a range of levels of confidence, it could do this only after twenty years since its inception. While this is most likely a biologically unresponsive timeframe for lizards (skinks), there is limited information on skink longevity in wild, though can live for several years (13+) in captivity ((striped skink) Whitaker 1993; (common skink) Towns Pers comm. 2002). No relative "Before" measure was made on the skink capture frequency prior to the initial management in 1996, and no comparative measure within Non-treatment has occurred. The lack of these two components in the monitoring programme seriously limit the determination of change for lizards (skinks) due to the restoration management actions occurring at the BSMIP. The design of the initial monitoring programme is similar to a survey to detect presence/absence of lizards, and it is this author's contention that this was the original objective established in 1996. Broad surveys are often not statistically robust (Benn 1995), and the investigation of the current BSMIP lizard monitoring programme supports this statement. The "simulated" monitoring programme targeting known skink habitats was shown to be at least twice as robust as the current design, and were reliable for a high level of confidence.

Monitoring programmes need to properly address sampling variance when measuring population variability and to present a means of doing so (Link & Nichols 1994). Statistical power is constrained by the inherent variability of the data gathered (Peterman 1990a; Osenberg *et al*. 1994). Eberhardt (1978), and Carpenter (1990) outlined that the variability of (community and ecosystem) measures may be so great that experiments may not detect perturbation effects unless they are very large. Hayes & Steidl (1997) states that monitoring should be focused on the rate at which population is changing over time. The BSMIP lizard (skink) monitoring programme in its current form cannot accomplish this with such high variability as has been obtained. If the lizard monitoring programme is to continue (targeting skinks), then the reduction of this variability is necessary. Two options for this are; focusing monitoring (i.e. repeated surveys) at skink habitats, and increasing the number

of sample sites. The effect of low number of sampling units has been discussed above in the wētā part of the statistical reliability section. The "simulated" lizard monitoring programmes, targeting high trap-catch sites, showed a greatly improved statistical power, though the determination of a decline (Appendix: 3-4) would still take several years. Link *et al*. (1994) states that for survey-type designs, it is frequently better to initiate new sites than to attempt to replicate existing sites.

Michener (1997) stressed a need for comparative analysis in the quantitative evaluation of restoration experiments. It is unfortunate that no Non-Treatment lizard monitoring is currently in place, though comparative trend monitoring would still be very useful if Non-Treatment lizard monitoring were established. With the continual RbM (Adaptive, or research management) approach being a key principle of the BSMIP, the potential for example the introduction of new management controlling rodents, specifically mice would make a BACI design still a viable option. The BACI monitoring model is very good at detecting changes due to impact (Eberhardt 1976), the most utilitarian for determining environmental change (Green 1979), and are practical examples of what Hargrove & Pickering (1992) term 'quasi-experiments' that do progress landscape ecology. The BACI design has become widely recommended (Underwood 1993), and the most appropriate for anthropogenic disturbances (Benedetti-Cecchi 2001). This is especially useful for conservation management considering operations and monitoring are (hopefully) planned and targeted at specific threats (i.e. pest animals), and assets (i.e. indigenous animals).

I suggest the current design be changed; with the establishment of more sample sites (that have fewer pitfall traps i.e. < 50) within the Treatment site, and the survey of likely habitats at the Non-Treatment sites, with the intention of establishing (Non-Treatment) monitoring sites. This has direct bearing to the guiding objectives, and biological relevance of the monitoring programme. I suggest that repeated measures analysis (MANOVA *multivariate analysis of variance*) be used as the analysis model for the BSMIP lizard monitoring programme (as with the tree wētā, and ground invertebrate monitoring programmes).

4.3.3.6 Lizard Monitoring Programme – Recommendations

I recommend that the current BSMIP lizard (skink) monitoring programme is terminated, and be reviewed. The data that has been gathered should be used as a preliminary study to determine capture variability over time. The design that is present at the moment could be still helpful as a long-term presence absence survey (i.e. done once every ten years or so), though a smaller more focused systematic sampling methodology would probably be more reliable, and certainly be more comprehensive in this regard. I suggest that a different annual monitoring design be instated, targeting areas of known skink habitat, and increasing timeframe of monitoring to cover the summer months. A change to using ten lines of ten sites, (in this thesis, the example was simulated high catch traps) monitored twice, with comparative Non-Treatment sites would be reliable in determining the response of lizards (skinks) to the restoration management. A change to the basic capture methodology and monitoring design, that is more closely linked to the skink's ecology should be also considered. I suggest lines or sites of clustered shelters, with four or more pitfalls under each shelter, incorporated at open grass sites as the annual lizard (skink) monitoring programme. A method of analysis including a "repeated measures" protocol should be used to statistically examine the lizard data.

4.3.4 Birds

4.3.4.1 Bird Monitoring – General

The New Zealand avifauna is substantially different from that of the rest of the world, due to the breakaway of "New Zealand" from Gondwanaland about 80 million years ago, allowing the formation of a highly endemic bird fauna (Gaze 1994). Bird-call counts have long been a popular method of surveying bird at a landscape scale (Rangen *et al.* 2000), and have been the basis for measuring an approximate density of birds in New Zealand (Dawson & Bull 1975; McKinlay 2001). The technique of bird counts has incorporated bird conservation into the decision-making process of indigenous forest management (Dawson & Bull 1975; McKinlay 2001).

4.3.4.2 Bird Monitoring Programme – Guiding Objectives

Hole-nesting birds such as stitchbird, and mohua, are extremely susceptible to predation by rats (Gaze 1994), and larger hole nesting birds such as kaka are susceptible to stoat predation. Predator control (including data from BSMIP) results in a substantial increase in robin breeding success, in a number of sites about New Zealand (Armstrong *et al.* 2002). Pierce (2002) found that breeding success approximately doubled for both pycroft's petrel, and little shearwaters following kiore removal on Marotere (Chickens islands). The monitoring of birds within the BSMIP, acts as a direct outcome measure for the management of predators, such as cats, rats, and stoats.

The bird monitoring also provides a secondary level of outcome data to the intensive multi-pest species management for such initial responses as vegetation change, and invertebrate numbers. Numerous bird species feed on invertebrates, such as morepork, robins, fantails, rifleman, tomtits, bellbirds, tuis, kiwi, and blackbirds (Heather & Robertson 1996; Stephenson (1998); Haw et al. (2001). Birds such as bellbirds and tuis are largely reliant on nectar, though also take fruit in autumn when flowers are not common (Heather & Robertson 1996). The response of birds to the multi-pest species management has multiple levels of interrelationships. The removal of insect predators and vegetation browsers at the Treatment site, is likely to provide birds with a greater abundance of food, than at the Non-

Treatment sites. This would increase the relative carrying capacity of birds within the Treatment site, than that of the Non-Treatment sites.

4.3.4.3 Bird Monitoring Programme – Biological Relevance

The bird monitoring programme at the BSMIP has clear interrelationships with other component programmes of the overall BSMIP Monitoring Programme, built on established biological knowledge, and results from published restoration projects. Birds, and bird eggs are preyed on by range of mammalian animals in New Zealand; rats: Innes (1990); Innes (2001); cats: Fitzgerald (1990); Gillies (2001); hedgehogs: (Moss & Sanders 2001); stoats: (King 1990; King *et al.* 2001); weasels (King 1990a; King *et al.* 2001): ferrets (Lavers & Clapperton 1990; Clapperton 2001); possums: (Sadlier 2000; McLeod & Thompson 2002); pigs: (McIlroy 1990; McIlroy 2001), and also dogs. Girardet (2001) states that after the cat eradication programme on Little Barrier island robins, parakeets, and warblers increased on some transects, and fantails and blackbirds decreased on one.

4.3.4.4 Bird Monitoring Programme – Statistical Reliability

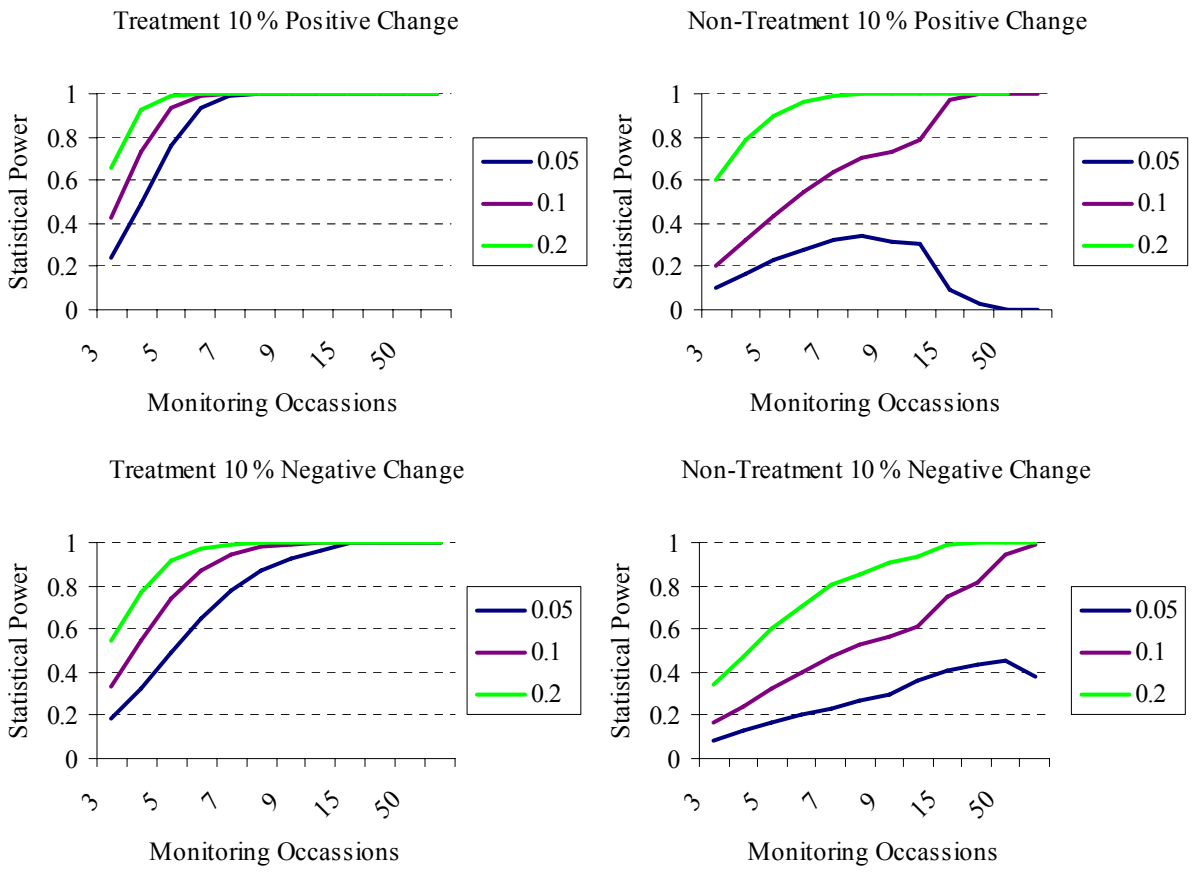

Figure 4.3.4.4.1 Statistical Power for Bird Monitoring Programme: Overall Nos.

The calculated statistical power values for the initial Bird monitoring programme (Overall Bird Numbers) show in general a robust design (from initial figures) for a 10% change. While the Treatment site has a robust design (reaching a calculated statistical power of 1.00 for all alpha levels), the Non-Treatment (combined sites) shows a robust design for only the higher (0.1, 0.2) alpha levels. The expected timeframe for the bird monitoring programme (overall numbers: with lines monitored once each season) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to ¾ years for the Treatment site, and one and ½ years for the (combined) Non-Treatment sites, for the determination of a 10% positive change.

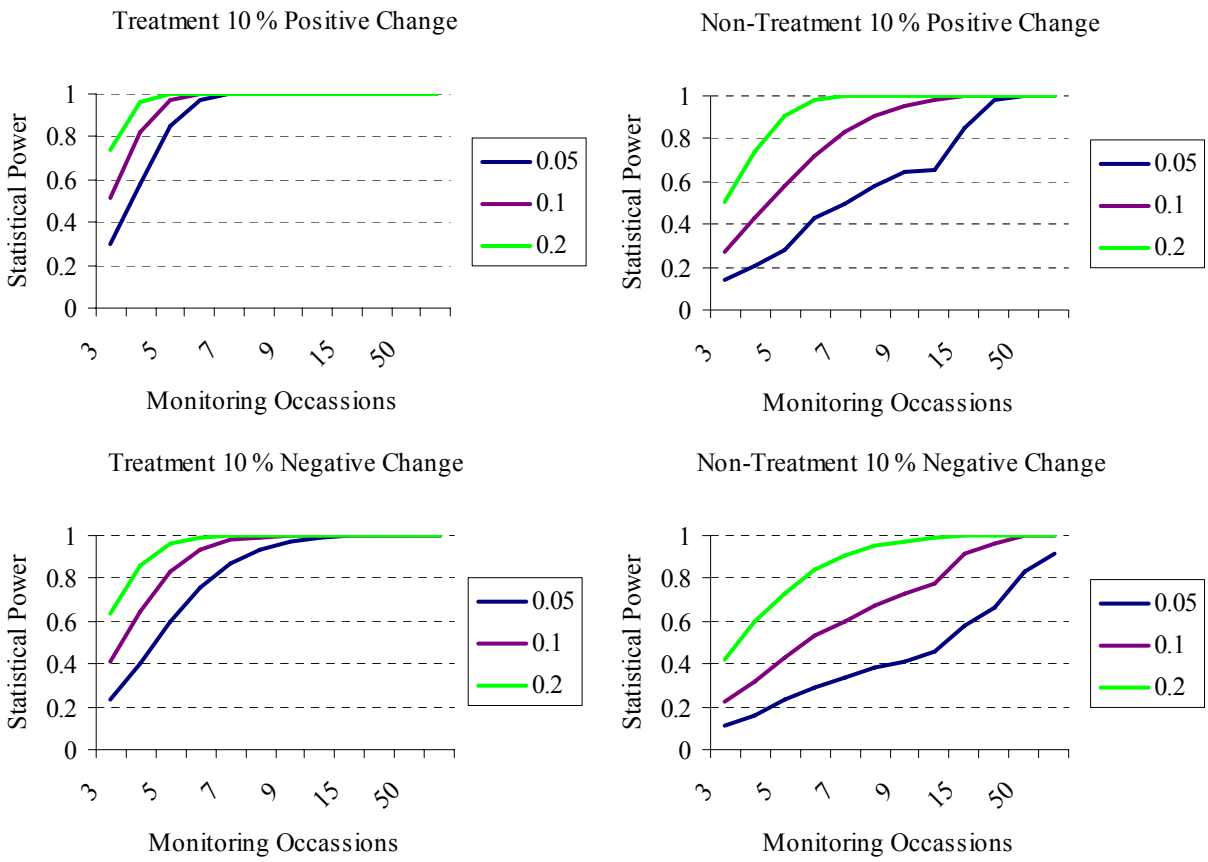

Figure 4.3.4.4.2 Statistical Power for Bird Monitoring Programme: No. of Species

The calculated statistical power values for the Bird monitoring programme (Total number of species) show in general a robust design (from initial figures) for a 10% change. While the Treatment site has a robust design (reaching a calculated statistical power of 1.00 for all alpha levels), the Non-Treatment (combined sites) shows a robust design for the 0.1, and 0.2 alpha levels with only a moderately robust design for the 0.05 alpha level. The expected timeframe for the bird monitoring programme (species numbers: with lines monitored once each season) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to one year for the Treatment site, and one and ¼ years for the (combined) Non-Treatment sites, for the determination of a 10% positive change.

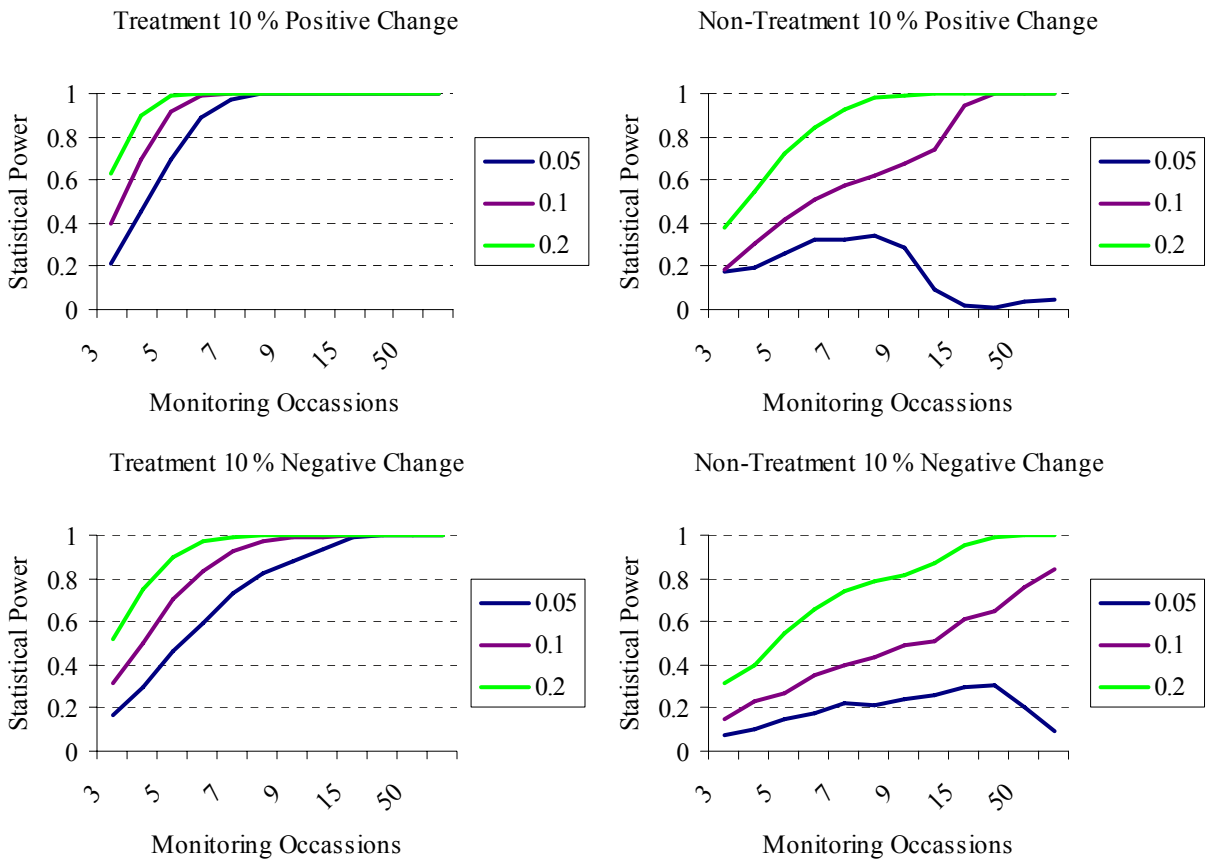

Figure 4.3.4.4.3 Statistical Power for Bird Monitoring Programme: Indigenous Bird Nos.

The calculated statistical power values for the Bird monitoring programme (Indigenous bird numbers) show in general a robust design (from initial figures) for a 10% change. While the Treatment site has a robust design (reaching a calculated statistical power of 1.00 for all alpha levels), the Non-Treatment (combined sites) shows a robust design for only the 0.2 alpha level with only a moderately robust design for the 0.1 alpha level. The expected timeframe for the bird monitoring programme (indigenous bird numbers: with lines monitored once each season) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to one year for the Treatment site, and one and ½ years for the (combined) Non-Treatment sites, for the determination of a 10% positive change.

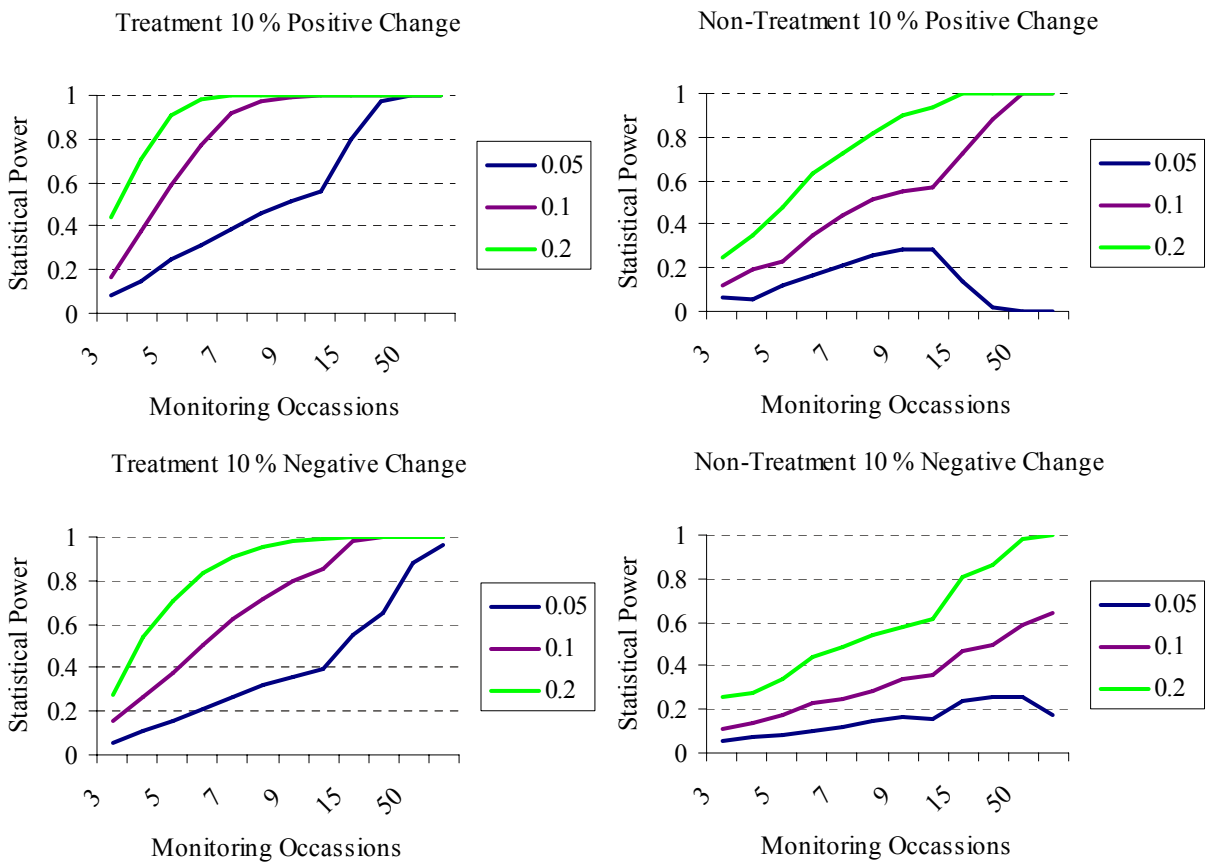

Figure 4.3.4.4.4 Statistical Power for Bird Monitoring Programme: Bellbird Nos.

The calculated statistical power values for the Bird monitoring programme (Bellbird numbers) show a moderately robust design (from initial figures) for a 10% positive change. While the Treatment site has a robust design (reaching a calculated statistical power of 1.00) for all alpha levels, the Non-Treatment (combined sites) shows a limited robust design for only the 0.2 alpha level. The expected timeframe for the bird monitoring programme (Bellbird numbers: with lines monitored once each season) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to one and ¼ years for the Treatment site, and two years for the (combined) Non-Treatment sites, for the determination of a 10% positive change.

The bird (five minute bird call counts) monitoring programme design was found to be generally robust for overall bird numbers, species numbers, indigenous bird numbers, and bellbirds numbers, though decreased in its statistical reliability (power), as the level of enquiry became more specific through to individual bird species numbers. Simulations for each significance criterion or alpha level, α (0.05, 0.1, 0.2,) reached the 'robust' (0.8 Statistical Power) level for a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence for the Treatment site. The Non-Treatment monitoring design simulations only achieved a robust level for the highest level of confidence (significance criterion: α 0.05) for the number of species, though not for the overall bird numbers, indigenous bird numbers, and Bellbird numbers.

This lack of strength for the highest level of confidence (significance criterion: α 0.05) in the Non-Treatment monitoring design is due to the low number of bird transect lines. The lack of enough sample sites (units) has been discussed in the tree wētā statistical reliability section 4.3.1.4. I suggest increasing the numbers of lines in the Non-Treatment sites, with one extra in each of Cashes Bush, and Thomas Bush Scenic Reserves, or other local Non-Treatment sites. This would increase the statistical power of the monitoring programme, and enable finer scale analysis, especially for some of the less abundant individual bird species (from initial numbers), e.g. shining cuckoo, tomtits, *etc*. If it is possible, these new lines should be placed in similar vegetation habitats to that of the Treatment site, so as to enable direct habitat comparisons to be made. I suggest that repeated measures analysis be used as the data analysis model for the BSMIP bird monitoring programme, because as with the tree wētā, ground invertebrate, and lizard monitoring programmes, the collection of data (bird counts) is 'repeated' spatially over time, as recommended by Green (1984; 1989; 1993); Underwood (1993); Norton (1996); Ribic & Ganio (1996).

This analysis concerns the statistical power of the bird monitoring design, though not necessarily the adequateness of the monitoring method. Bird-call counts have long been the basis for measuring an approximate density of birds in New Zealand, *circa* Dawson & Bull (1975), and now are a widely accepted measure of conspicuousness (McKinlay 2001). Dawson & Bull (1975) identified the distance detectability of birds within New Zealand's

forests as an issue, and since Buckland *et al*. (1993) improvement in estimate of absolute density, or abundance as related to distance from an observer, more adequate measures of abundance can be determined. Cassey (1999) suggests that distance sampling be incorporated into conservation management. This would improve the reliability of conservation monitoring programmes.

4.3.4.5 Bird Monitoring Programme – Conclusion

The BSMIP bird monitoring programme is robust, balanced, and will fulfil the important need for long-term trend monitoring of this group of animals. It encompasses comparative Non-Treatment sites, and has a valid cause ("results" of management; such as rodent, mustelid, and possum monitoring) and effect ("outcome" of bird abundance) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects. The bird monitoring programme links to other outcome monitoring programmes such as the invertebrate monitoring programme (numbers and composition of invertebrates), and the vegetation monitoring programme (composition, and phenology health of the forests). McKinlay (2001) states that current bird counting methods are often inadequate to provide confidence that all birds that are present are being detected. The adequacy of the relative level of detection is most likely different for each species, and thus the introduction of distance sampling methodology as suggested by Cassey (1999) to the current five minute bird count methodology would prove to be have more reliability (Thompson 2002).

The initial bird monitoring programme design was found to be robust, and reliable to detect a trend (+/- 10%) for a moderate to high level of confidence (α: 0.1, or 90%) upon four and a half years of its inception. I consider this to encompass a biologically responsive timeframe for bird species, indigenous bird numbers, species numbers, and the overall bird numbers. This timeframe would encompass the lifespan of the majority of birds, and certainly a number of breeding seasons. An initial "Before" measure was performed prior to the start of the intensive multi-pest species management in 1996, providing a comparative measure of outcome change due to management. The statistical analysis on the bird count data requires consideration of space-time correlation (Millard *et al*. 1985; Buckland *et al*. 1997), which is

best addressed by repeated measures analysis (Green 1984; Green 1989; Green 1993; Underwood 1993; Norton 1996; Ribic & Ganio 1996).

4.3.4.6 Bird Monitoring Programme – Recommendations

I recommend that the current methodology is continued, with the core monitoring programme retained, and if possible incorporate distance sampling into the methodology, with additional systematic distance sampling of new areas. Repeated measures analysis should be used to analyze the bird count data.

4.3.5 Vegetation

4.3.5.1 Vegetation Monitoring – General

New Zealand's indigenous conifer, broadleaf, and beech forests are an important landscape feature (Dawson 1988), covering approximately 23% of the land surface (Allen 1993; Newell *et al.* 2002). Treshow & Allen (1985) state that ecosystems are dynamic and are in constant of change, which promotes a level of uncertainty in the assessment of vegetation. Permanent vegetation plots are recognized as an effective method for detailing forest dynamics, encompassing this inherent variability (Austin 1981; Newell *et al.* 2002). The 20m x 20m vegetation plot is extensively used for the purpose of research and monitoring, by the examination of forest structure, species composition, and species distribution (Newell *et al.* 2002).

4.3.5.2 Vegetation Monitoring Programme – Guiding Objectives

The monitoring of the vegetation within the BSMIP, acts as a direct outcome measure for the management of browsing mammals such as goats, deer, and possums. Allen *et al.* (1984) found that despite a reduction in deer numbers (introduced browsing animals), the structure and composition of most forest types are still affected. Within a few years of ungulate exclusion, palatable plant species numbers increase (Allen *et al.* 1984; Stewart & Burrows 1989; Smale *et al.* 1995; Nugent *et al.* 2001). Sweetapple & Burns (2002) found that vegetation recovery occurs following the control of goats, though understorey condition improvements following possum control were negligible in the absence of effective goat control. Bellingham & Allan (2002) found that the frequency of whitetail deer (*Odocoileus virginianus*) on Stewart Island was a significant predictor of increased seedling density of unpalatable species and decreases of palatable species at a plot scale. Such studies lead to the determination that the relative intensity and scope of pest mammal control was highly important if restoration objectives were to be met.

The vegetation monitoring also provides a secondary level of result data to the intensive multi-pest species management for such responses as changes in wētā, and bird abundance. As tree wētā are largely vegetarians (Gibbs 1998), the vegetation monitoring (alongside phenology, and FBI (Foliar Browse Index) could additionally act as a secondary result

monitoring programme. Such interrelationships are based on trophic linkages (Odum 1983), and a benefit-cascade assumption. That being the case, the removal of mammalian browsers of vegetation at the Treatment site, is likely to provide birds with a greater abundance of food, than at the Non-Treatment sites. This would increase the relative carrying capacity of birds, and wētā within the Treatment site, compared to that of the Non-Treatment sites.

4.3.5.3 Vegetation Monitoring Programme – Biological Relevance

The vegetation monitoring programme at the BSMIP has clear interrelationships with other component programmes of the overall BSMIP Monitoring Programme, built on established biological knowledge, and results from published forest conservation management projects. Canopy, and sub-canopy vegetation (foliage, buds, flowers, bark) is browsed on by Possums (Cowan 1990; Allen *et al.* 1997; Nugent *et al.* 2000; Cowan 2001; Cochrane *et al.* 2003), and shrub-tier and ground foliage is browsed on by Goats (Rudge 1990 Parkes 2001), and Deer (Nugent *et al.* 2001), seed and seedlings by rats (Allen *et al.* 1994), and below ground vegetation is rooted up by Pigs (McIlroy 1990; McIlroy 2001). Introduced vertebrate pest mammals have had a large though often varying impact on New Zealand's indigenous forest communities (Wardle *et al.* 2001), preferentially affecting plant species composition, and abundance; possums: Payton (2000); Sweetapple & Burns (2002); goats: Parkes (2001); Sweetapple & Burns (2002); deer: Nugent *et al.* (2001)).

4.3.5.4 Vegetation Monitoring Programme – Statistical Reliability

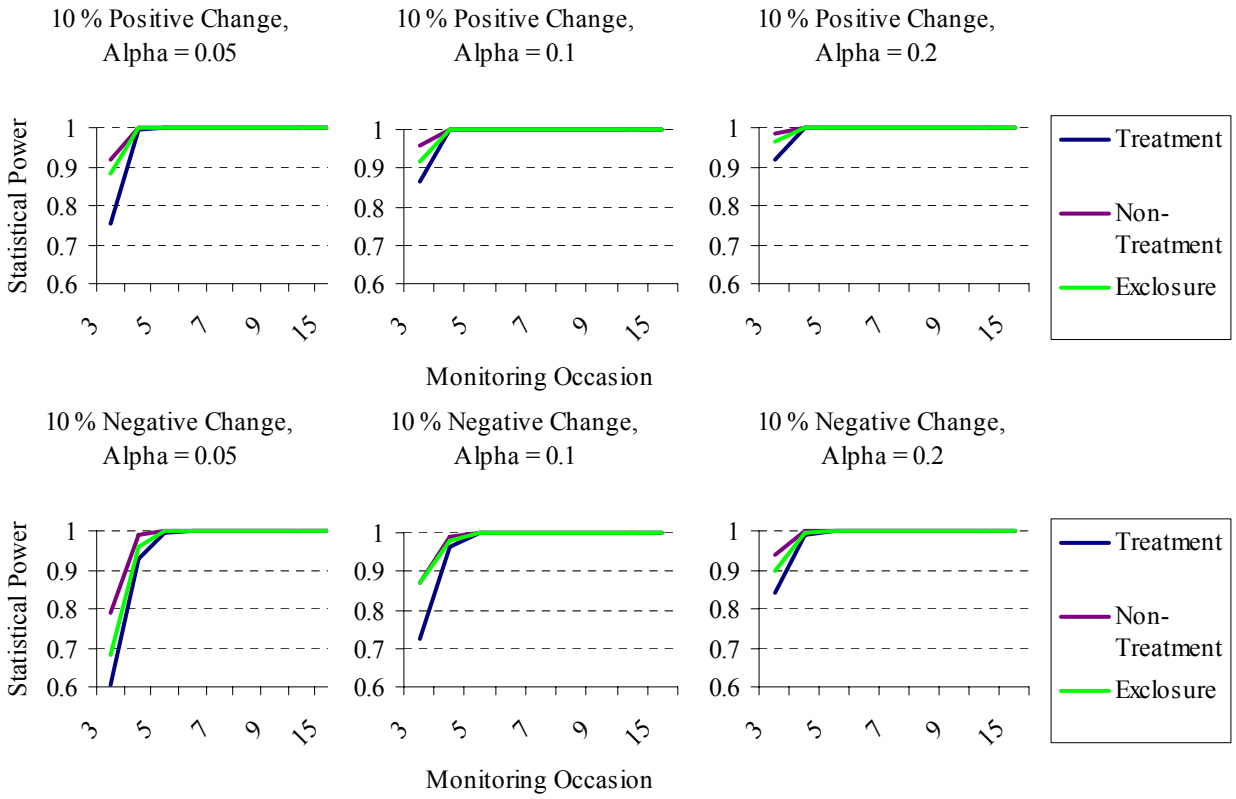

Figure 4.3.5.4.1 Statistical Power for Vegetation Monitoring Programme: Species Nos.

The calculated statistical power values for the Vegetation monitoring programme (Species Numbers: Seedling, and all Sapling tiers) show an extremely robust design (from initial figures) for a 10% positive change. All three management regimes (Treatment, Non-Treatment, and Exclosure sites) have a very robust design (reaching a calculated statistical power of 1.00 within three monitoring occasions) for all alpha levels, within five monitoring occasions. The expected timeframe for the vegetation monitoring programme (Sapling numbers: with plots monitored once every five years) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.1) level equates in this case to five years (at the completion of the second monitoring occasion) for both the combined Non-Treatment Sites, and Exclosure sites, and ten years for the Treatment Site (for the determination of a positive change).

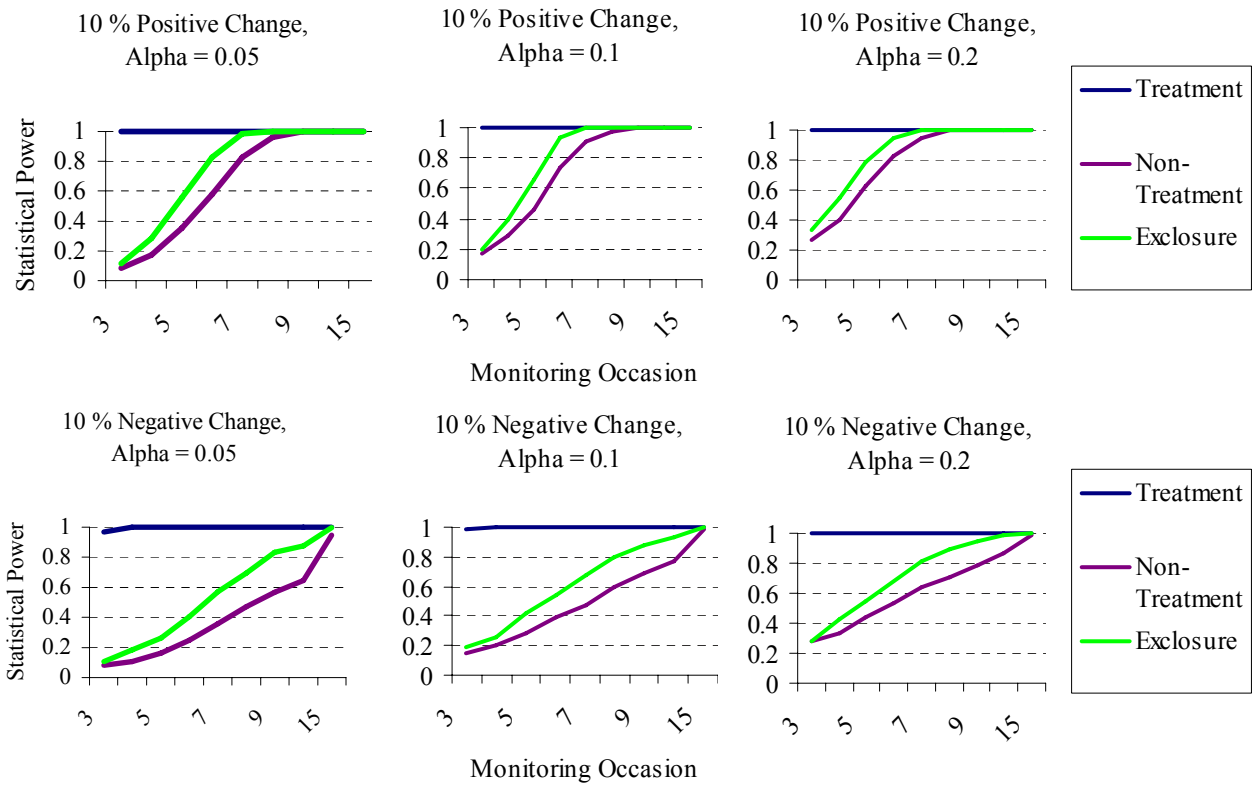

Figure 4.3.5.4.2 Statistical Power for Vegetation Monitoring Programme: Sapling Nos.

The calculated statistical power values for the Vegetation monitoring programme (Sapling Numbers: Seedling and first four Sapling tiers) show a very robust design (from initial figures) for a 10% positive change. While the Treatment site has an extremely very robust design (reaching a calculated statistical power of 1.00 within three monitoring occasions) for all alpha levels, the Non-Treatment (combined sites) and Exclosure site regimes show a moderate to strong robust design. The expected timeframe for the vegetation monitoring programme (Sapling numbers: with plots monitored once every five years) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.1) level equates in this case to five years for the Treatment site, 30 years for the (combined) Non-Treatment sites, and 25 years for the exclosure sites. (for the determination of a positive change).

The vegetation (20x20m vegetation plots) monitoring programme design was found to be highly robust for species numbers, sapling numbers, though the Non-Treatment took up to six times as long as the Treatment site to reach a similar level of reliability for the Sapling numbers (seedling, and first four sapling tiers) (Appendix: 1,2). Simulations for the three significance criterion or alpha levels, α (0.05, 0.1, 0.2) reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence.

This difference in statistical reliability timeframes is due to the lower relative number of vegetation plots between the Treatment (19 initial; 16 ongoing), the Non-Treatment (8 initial; 4 ongoing), and the Exclosures (7). The lack of enough sample sites (units) has been discussed in the tree wētā statistical reliability section 6.4.1. I suggest adding an extra vegetation plot to each of the Non-Treatment sites, Cashes Bush, and Thomas Bush Scenic Reserves, or other local Non-Treatment sites. This would increase the statistical power of the monitoring programme, an enable finer scale analysis, for changes in individual plant species, e.g. kamahi (*Weinmannia racemosa*), rewarewa (*Knightia excelsa*), *etc*. If it is possible, these new plots should be placed in similar vegetation habitats to that of the Treatment site, so as to enable direct habitat comparisons to be made. I suggest that repeated measures analysis (MANOVA *multivariate analysis of variance*) be used as the data analysis model for the BSMIP vegetation monitoring programme, because as with the tree wētā, ground invertebrate, lizard, and bird monitoring programmes, the collection of data (vegetation plot) is 'repeated' spatially over time.

4.3.5.5 Vegetation Monitoring Programme – Conclusion

The vegetation monitoring programme design was found to be highly robust for species numbers, and moderately robust for sapling numbers. It is reliable to detect a trend (+/-10%) for a moderate to high level of confidence (α: 0.1, or 90% CI) upon thirty years of its inception. I consider this to be a biologically responsive timeframe to determine the comparative vegetation response due to restoration management, as it is well within the timeframes for temporal fluctuations influencing vegetation composition (Austin 1981), an addresses forest successional and cyclic hypotheses (Dawson 1988). The initial

measurements of the majority of vegetation plots were made prior to the spring flush of vegetation in 1996, effectively establishing a baseline "Before or Pre" measurement. This provides a comparative measure of outcome change due to the intensive multi-pest species management. Similar to the tree wētā, ground invertebrate, lizard, and bird data, the statistical analysis on the vegetation data requires consideration of space-time correlation (Millard *et al*. 1985; Buckland *et al*. 1997), which is best addressed by repeated measures analysis (Green 1984; Green 1989; Green 1993; Underwood 1993; Norton 1996; Ribic & Ganio 1996).

Nugent *et al.* (2001); Wardle *et al.* (2001) state that the conservation of indigenous biodiversity in New Zealand's remaining indigenous forests is threatened by the presence of introduced wild mammalian herbivores, as they have the potential to radically change the vegetative structure and composition. The BSMIP vegetation monitoring programme will add important findings to the conservation of New Zealand's forests, as it records the response of the Boundary Stream Scenic Reserve forest to a new (multi-all species) level of pest management. The BSMIP vegetation monitoring programme design is effective, balanced, and will fulfil the important need for long-term trend vegetation monitoring within the BSMIP. It encompasses comparative Non-Treatment sites, and has a valid cause ("results" of management; such as rodent, and possum monitoring) and effect ("outcome" of plant species composition, and abundance) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects.

4.3.5.6 Vegetation Monitoring Programme – Recommendations

Recommendations include an extra permanent vegetation plot in each of the Non-Treatment sites, or other local Non-Treatment reserves. If possible, these new plots should be placed in ideally comparative, or at least similar vegetation habitats to those of the Treatment site, so as to enable direct habitat comparisons to be made. The monitoring of the plots should all occur within the same season of the same year, and be spaced at least five years apart. The inclusion of a repeated measures protocol should be used to analyse the vegetation plot data.

4.3.6 Mustelids

4.3.6.1 Mustelid Monitoring – General

Conservation (and certainly preservation) of many of New Zealand's endemic species will rely on the removal of mustelids, and other mammalian predators (Gillies & Murphy 1997). Footprint tracking tunnels are used widely to provide an index of density and abundance of small introduced mammals (King & Edgar 1977; Brown *et al.* 1996; Blackwell 2002). Effective methods of monitoring abundance are important tools for the management of predators (Wilson & Delahay 2001). Research indicates that mustelids have large home ranges, and while these are often overlapping (Clapperton 2001; King et al. 2001), the size of the home ranges would confer a relative low density of mustelids, compared with other mammals. At low animal densities, index values and indeed monitoring in general has a lack of precision and large confidence intervals (CI), this makes the determination of changes, or comparisons between management and/or sites difficult (Thomas & Brown 2000).

4.3.6.2 Mustelid Monitoring Programme – Guiding Objectives

The BSMIP mustelid monitoring programme, acts as an operational or result measure of the intensive management (i.e. the extensive Fenn trap network ringing the reserve). The three mustelids species in New Zealand; ferrets, stoats, and weasels, are all broadly flexible in their diet, and being opportunistic predators can rapidly shift to alternative though available resources (Clapperton 2001; King *et al.* 2001). Such studies lead to the determination that the relative intensity and scope of the pest mammal control (whether a suite, or single-pest species focus) are highly important if restoration objectives are to be met. Research on predators including mustelids has found that they generally have a low density in the wild, with large dispersal as juveniles, and a large home range (Jedrzejewski *et al.* 1995; Caley & Morriss 2001; Miller *et al.* 2001) in relation to other similar sized mammals (Innes & Skipworth 1983; Hooker & Innes 1995). As these animals exist and act on a larger spatial scale, monitoring lines or sites need to become more spatially separated to ensure independence of counts. A larger, if not landscape-sized scale approach for monitoring predators must occur.

4.3.6.3 Mustelid Monitoring Programme – Biological Relevance

Ferrets feed predominantly on rabbits (Clapperton 2001), stoats feed predominantly on rats (King *et al.* 2001), and weasels feed predominantly on mice (King *et al.* 2001), though birds, lizards, and invertebrates are important food for all mustelids depending on the relative availability and amount of prey. Stoat impacts in New Zealand have attracted considerable research compared to that on weasels and ferrets, with clear impacts known on birds (King 1984; Wilson *et al.* 1993; McLennan *et al.* 1996; McLennan 1997; Cuthbert *et al.* 2000), lizards (King 1990; Miskelly 1997; King *et al.* 2001), and invertebrates (King & Moody 1982; Rickard 1996). Such a breadth of direct impact on the indigenous fauna, means that net outcomes of mustelid management must be measured at the community level. Though King (1984) does state, the most dominant effect on New Zealand's avifauna biota has been human influenced effects such as habitat destruction, and not the predation pressure from mustelids.

4.3.6.4 Mustelid Monitoring Programme – Statistical Reliability

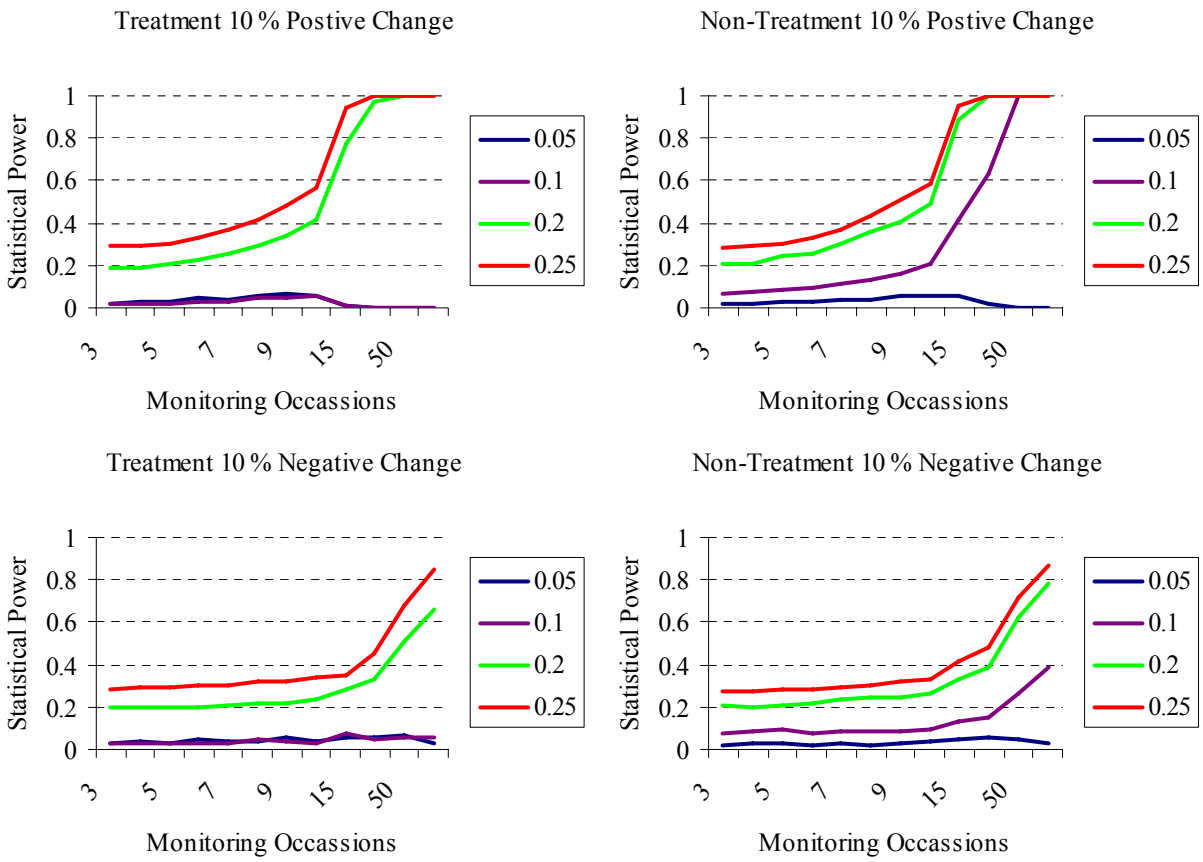

Figure 4.3.6.4.1 Statistical Power for Mustelid Monitoring Programme

The calculated statistical power values for the Mustelid monitoring programme (Total Numbers) show a low robust design (from initial figures) for a 10% positive and negative change, with only a moderate strength at the higher alpha levels. Both management regimes (Treatment, Non-Treatment sites) have a moderately robust design (reaching a calculated statistical power of 1.00) for the higher alpha levels. The expected timeframe for the mustelid monitoring programme (Total numbers: per run, four times per year) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.25) level equates in this case to approximately twenty years for both management regimes (for the determination of a negative change).

The mustelid (tracking tunnel) monitoring programme design was found to have a low level of robustness for detecting a change in the relative abundance (tracking rate) of mustelids.

Only alpha level α 0.25 reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time for both the Treatment and Non-Treatment sites, indicating that the monitoring design was un-reliable at a high level of confidence.

At the time of the initial part of the BSMIP establishment, a smaller than expected encounter rate (tracking tunnel rates, and trap-catch rate) of mustelids was found about the Boundary Stream (Maungaharuru Range) area compared to other indigenous forest sites. The effect that this small initial index value has on a monitoring programme has been discussed in the wētā statistical reliability section 4.3.1.4. Brown & Miller (1998) outlined the degree to which low numbers, and the pre-control density (initial index values) had on a mustelid (stoat) monitoring programme design, and determined that the tracking tunnel methodology can be statistically robust for large reductions (e.g. 50%, 70% control) in stoat populations.

The current BSMIP mustelid monitoring programme was not established prior to the initial management (aerial 1080 drop, trap lines, and bait station network), and so missed the opportunity for a BACI design. Because of the possible low density of mustelids about the Boundary Stream area, and the low tracking indices found, the level to which statistical analysis of the data can be performed is limited. While trend monitoring in comparison to Treatment and Non-Treatment sites, may be useful, I suggest that the monitoring programme be used as a presence-absence design as a "trigger" for targeted intensive management (e.g. additional traps in these areas, and/or the use of stoat dogs). Strayer (1999) states that because of potential problems with bias and inadequate power, presence-absence designs should be used and interpreted cautiously.

An additional approach that could indicate large spatial changes in mustelid distribution would be presence/absence surveys over a greater area about the Maungaharuru range. Mustelids have a large home range in comparison to other mammal species of similar or even significantly larger in size (King *et al.* 2001; Murphy & Dowding 1994; 1995). The methodology may be the; use of index-oriented surveys (i.e. counting track rates) in localized areas of high relative abundance (spread over a larger area). This landscape-scale approach (due to ranging behaviour of mustelids and especially stoats, in New Zealand) would similar in principle to the goat management as outlined in section 2.7

I suggest that the data from the mustelid monitoring programme, be used as descriptive rather than for interpretative statistics, and as covariate information with other monitoring and management programmes (e.g. bird counts, mustelid trapping results, cat management results, rabbit abundance within and about the BSMIP, *etc*). Until a more reliable measure of detecting low level abundance of mustelids is developed for management, interpretative statistical analysis of BSMIP mustelid monitoring data would have limited value for management.

4.3.6.5 Mustelid Monitoring Programme – Conclusion

The BSMIP mustelid monitoring programme design is effective in the sense of detecting the presence of mustelids, and is balanced between comparative sites of the two different management regimes, although it will not alone fulfil the important need for long-term trend monitoring of mustelids within the BSMIP. It has a valid ("outcome" of tree wētā and ground invertebrate abundance, and plant species composition and abundance) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects. The apparent low densities of mustelids raises the question of what is our confidence of determining the trend of rat numbers due to the ongoing management? Blackwell *et al.* (2002) advocates using a second density estimate independent from tracking rates for rodents, and I suggest such an approach be used to better determine mustelid density, and would give a comparative measure of tracking tunnel effectiveness and precision. Ruscoe *et al.* (2001) states that the use of such tracking tunnel indices to establish animal (mice) population trends assumes that a systematic relationship between the index and actual animal (mice) density. A double sampling methodology as mentioned by Blackwell *et al.* (2002) would approach the determination of a systematic relationship between the tracking tunnel index, and the actual mustelid density. Choquenot *et al.* (2001) found that for simulated detection (0.7 probability of a stoat entering a tunnel) of low numbers of stoats (<5), by their colonization into new areas, an exponential increase in numbers of tracking tunnels was required. Though from their graph of required tracking tunnels per numbers of stoats, approx. 100 tunnels per treatment site were still necessary for the detection of a stoat (n = at least five animals, at 90% CI). Mustelids have large home

ranges, and disperse large distances, and therefore any monitoring programme should operate on a similarly large spatial scale. Engeman *et al.* (2002) used a passive tracking index (dirt roads) to simultaneously monitor multiple species of animals which was found to be useful, and could also determine some of the biology of the animals. Checking of such areas (dirt and sand roads), or placing scavenger boards at a wider spatial scale about the BSMIP, could additionally be useful to target such mustelid monitoring, and management.

The mustelid monitoring programme design was found to have a low level of "robustness" for the detection of a change in population numbers. It was found to only be reliable to detect a trend (+/- 10%) for a low level of confidence (α: 0.25, or 75% CI) only after twenty years since its inception (using the initial data). I do not consider this to be a biologically responsive timeframe to determine the effectiveness of mustelid control, as it is substantially larger than the mating season of one year, with female stoats reproductively mature while unweaned nestlings, and young males mature within one year (King 1990; King *et al*. 2001). No relative "Before" measure was made on the mustelid tracking rates prior to the initial management in 1996, though since the management is ongoing, the comparative difference between the Treatment and Non-treatment sites would provide a indicative measure of activity at each site. Repeated surveys may offer some increase in the statistical power of the monitoring design although it is likely that trends would still be hard to determine. Limited statistical analysis can be made for trends with this data, though it may well be useful as a "trigger" for mustelid management in particular areas.

4.3.6.6 Mustelid Monitoring Programme - Recommendations

The current BSMIP mustelid monitoring programme should be continued, though its primary use should be as a presence/absence response tool, inorder to trigger targeted or focused management (e.g. increased traps at a particular site or area). Currently, limited statistical analysis can be performed to determine mustelid population trends over time, though with an additional method of measuring an index of density (Blackwell 2002), a better determination of relative mustelid abundance can be gained. An investigation to determine such a double sampling design should be performed.

The consideration of the ecology of mustelids, specifically their home range, and activity patterns (e.g. use of roads, stream sides, *etc*) would necessitate a more extensive monitoring programme (and management) buffer be instated. I suggest that lines or sampling sites at distances at least up to 5 km from the Treatment site be established. The methodology could be focused on presence/absence of mustelids, by using index-oriented surveys counting the track rates among each tracking tunnel group (i.e. cluster, or line), and so identify localized areas of high relative abundance. This would make available the opportunity to target additional (temporary and site focused) management in a wider spatial scale for the BSMIP.

For new projects, extensive and intensive monitoring (many small lines of tracking tunnels – monitored weekly) in a BACI design to determine change in numbers, similar to Brown & Miller (1998) recommendations for stoat monitoring with tracking tunnels should be established. After the initial control, and monitoring has been performed, it may be then possible to modify the monitoring programme to focus on trend monitoring, eg. tracking tunnels used seasonally, and used as a trigger for targeted management.

4.3.7 Rodents

4.3.7.1 Rodent Monitoring – General

Both rats and mice have had dramatic effect on the New Zealand environment, and are still important pests (Innes 2001; Ruscoe 2001). Footprint tracking tunnels are used widely to provide an index of density and abundance of small introduced mammals (King & Edgar 1977; Brown *et al.* 1996; Blackwell 2002). Monitoring of rodents in general has a lack of precision and large confidence intervals (CI) with animals at low densities, and this makes the determination of changes, or comparisons between management and/or sites difficult (Thomas & Brown 2000). Given that we can effectively control rodents on islands (Atkinson 2001), and rats on the mainland (within intensively managed projects) to low densities, it would be timely to develop a more reliable methodology to determine the abundance of rodents at low density.

4.3.7.2 Rodent Monitoring Programme – Guiding Objectives

The BSMIP rodent monitoring, acts as an operational or result measure of the intensive management (i.e. the initial 1080 aerial poison drop, and the subsequent presentation of toxin through the 150m x 150m bait station network, as well as the extensive Fenn trap network ringing the reserve). Taylor & Thomas (1993) described one of the initial ground-based control operations using bait stations, where norway rats (*Rattus norvegicus*) were eradicated from Breaksea Island (170ha) during 1986. Within a period of two-five years after this rat eradication, seedling numbers of many tree and shrub species had increased substantially on Breaksea Island (Allen *et al.* 1994), outlining the effect of rats, and the potential response of vegetation to intensive rat control. Brockie (1992) wrote from an intensive 24-year study, that both ship rats (*Rattus rattus*) and possums were the "*most pervasive and devastating agents of change*" within a New Zealand lowland forest. Thus rodents are monitored not only as they target pest, but also to confer any valid relationship to the response of indigenous components of the ecosystem in both the treatment regimes of the BSMIP.

4.3.7.3 Rodent Monitoring Programme – Biological Relevance

Both rats and mice have an omnivorous diet of adult arthropods, lepidopteran larvae, fruits and seeds (Gales 1982; Badan 1986; Innes 1990; Moors 1990; Murphy & Pickard 1990; Innes 2001; Ruscoe 2001), though their abundance in habitats such as beech forests appears to linked through the invertebrate fauna abundance, rather than seeds (Studholme 2000; Alley *et al.* 2001). Both Allen *et al.* (1994), and Campbell (2000) found seedling numbers, and numbers of many tree and shrub species increased substantially after rat eradication occurred on off-shore islands. Such a breath of direct impact on the indigenous biota, means that net outcomes of rodent management must be measured at the community level, so as to register expected and unexpected responses such as diet switching by predators (Innes 2001). Such studies lead to the determination that the relative intensity and scope of the pest mammal control (whether a suite, or single-pest species focus) are highly important if restoration objectives are to be met.

4.3.7.4 Rodent Monitoring Programme – Statistical Reliability

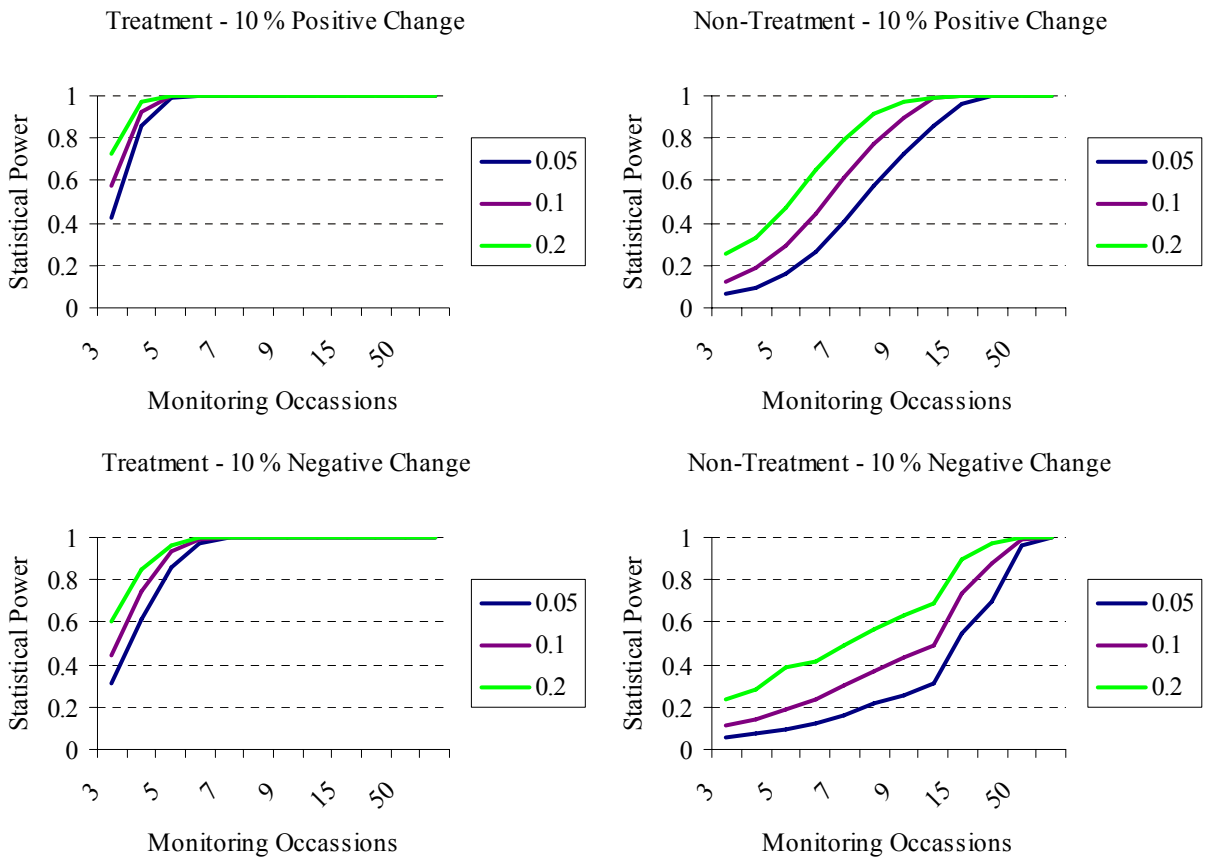

Figure 4.3.7.4.1 Statistical Power for Rodent Monitoring Programme

The calculated statistical power values for the Rodent monitoring programme (Total Numbers) at the Treatment site show a very robust design (from initial figures) for a 10% positive and negative change at all alpha levels. The Non-Treatment sites have a moderately robust design (reaching a calculated statistical power of 1.00) for all alpha levels. The expected timeframe for the Rodent monitoring programme (Total numbers: per run, four times per year) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to one and ¼ years for the Treatment site, and three years for the Non-Treatment sites (for the determination of a negative change).

The rodents (tracking tunnel) monitoring programme design was found to be strongly robust, though the Non-Treatment took up almost twice as long as the Treatment site to reach a similar level of reliability (Appendix: 4, 5). Simulations for the three significance

criterion or alpha levels, α (0.05, 0.1, 0.2) reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence.

While rodent tracking tunnels have been shown to be an extremely reliable method of detecting rodents, and their relative activity, I suggest that the standard methodology be changed or at least reviewed (hence a possible change to the standard method throughout the Department of Conservation) in order to use more lines (of less numbers of tracking tunnels) in any given area, similar to the current (national standard) possum monitoring design. This is in response to the effectiveness of the rat control that is occurring within the BSMIP Treatment site, and other such intensive pest control operations. It is difficult to monitor populations at low densities (Beier & Cunningham 1996), and greater spatial coverage of areas especially the Treatment Site should be better able to detect eruptions in rat numbers prior to them becoming an impact. An increased number of smaller tracking tunnel lines would not only give greater coverage spatially, many smaller lines or groups (e.g. ten tracking tunnels per line, nine in a square grid, or seven in triangular grid) is likely to increase the statistical power lines of the monitoring design, and be able to give a more reliable estimate of the relative rodent abundance.

I suggest that repeated measures analysis be used as a data analysis model for the BSMIP rodent monitoring programme, because as with the tree wētā, ground invertebrate, lizard, bird, and vegetation monitoring programmes, the collection of data (rodent tracking rate) is 'repeated' over time.

4.3.7.5 Rodent Monitoring Programme – Conclusion

The BSMIP rodent monitoring programme design is effective, balanced, and will fulfil the important need for long-term trend rodent monitoring within the BSMIP. It encompasses comparative Non-Treatment sites, and has a valid ("outcome" of tree wētā and ground invertebrate abundance, and plant species composition and abundance) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects. The reduction of rat numbers to very low densities at the Treatment site

(McRitchie 2000; King & McRitchie 2001; King 2002), raises the question of what is our confidence of determining the trend of rat numbers due to the ongoing management. Blackwell *et al.* (2002) advocated using a second density estimate apart from tracking rates for rodents, and I would certainly support this approach to better determine rodent density, as well as a comparative measure of tracking tunnel effectiveness and precision. Ruscoe *et al.* (2001) states that the use of such tracking tunnel indices to establish rodent (mice) population trends assumes that a systematic relationship between the index and actual rodent density. A double sampling methodology as mentioned by Blackwell *et al.* (2002) would advance the determination of a relationship between the tracking tunnel index, and the actual rodent density.

4.3.7.6 Rodent Monitoring Programme – Recommendations

Recommendations include a review of standard tracking tunnel monitoring design, as more smaller lines (e.g. ten tracking tunnels per line, nine in a square grid, or six or ten in triangular grid) would be far more robust. Should such sampling sites be established, they should be placed in ideally comparative, or at least similar vegetation habitats between that of the Treatment site and Non-Treatment sites, so as to enable direct habitat comparisons to be made. More, smaller lines or groups of tracking tunnels, would provide greater statistical power, greater coverage spatially, and be more likely (because of the increased spatial coverage) to recognize where, and when rodents are "irrupting". Analysis of the rodent monitoring data should incorporate a repeated-measures protocol in any interpretative statistical examination.

4.3.8 Possums

4.3.8.1 Possum Monitoring – General

The possum trap-catch monitoring is the main result or operational method used by regional councils, Department of Conservation, and researchers, and is an essential part of possum management (Warburton 2000). Control techniques have allowed the reduction of many possum populations to very low levels, and now natural resource managers are requesting monitoring methods that can differentiate between residual possum densities that may only differ by 3-4% (Warburton 2000). Similar to rats within the BSMIP Treatment site, possums are at low animal densities, as recorded by current index values. At such low animal densities, index values and indeed monitoring in general has a lack of precision and large confidence intervals (CI), this makes the determination of changes, or comparisons between management and/or sites difficult (Thomas & Brown 2000).

4.3.8.2 Possum Monitoring Programme – Guiding Objectives

The BSMIP possum monitoring, acts as an operational or result measure of the intensive management (i.e. the initial 1080 aerial poison drop in 1996, and the subsequent presentation of toxins through the 150m x 150m bait station network). While possums do cause damage to indigenous New Zealand forest communities, the degree, an extent of this damage varies widely (Norton 2000; Payton 2000). Thus there is a valid need for a broad vegetation monitoring programme, as an outcome reference to the restoration efforts, including specific monitoring of highly preferred plant species at sites (Norton 2000) such as mistletoe (Norton 1997; Norton & Reid 1997; Ogle 1997) if possible. The vegetation plot network acts as the "backbone" of the outcome monitoring within the BSMIP, although requires specific programmes such as threatened plant monitoring to cover the assessment of impacts from the "suite" of pest animals.

While possum control does benefit native vegetation in general, the determination of vegetation response has always been adequately considered. Norton (2000) makes the case for proper design of vegetation monitoring programmes, and specifying replication and adequate statistical power as key criteria in their design. Veltman (2000) states that the evidence for native wildlife benefiting from control specifically targeting possums is weak,

and correctly points out the need for researchers to perform experiments to "*explicitly test for the effects of possums on populations of native wildlife*". This is a need that the established Mainland Island Projects, and similar "Intensive Multi-Pest Species Control" projects cannot achieve, as their overall goals are fundamentally different. Mainland Island Projects have broad, and comprehensive pest control as currently possible – so the indigenous ecosystem and its components benefit, whereas foodweb research as outlined by Veltman (2000) above, needs to be focused to a specific pest at a single site or have multiple treatments if targeting more than one pest.

4.3.8.3 Possum Monitoring Programme – Biological Relevance

The brushtail possum (*Trichosurus vulpecula*) has very broad impacts on New Zealand's forest biota, and indigenous ecosystem. While possums consume large amounts of indigenous vegetative matter, including a large number of plant species, the influence and damage to forest ecosystems communities widely (Norton 2000; Payton 2000). A review on the evidence of possums preying on native animals has shown a large range of indigenous animals in possum diet: native birds, their chicks and eggs, snails, and insects (Sadlier 2000). This has been a relatively recent discovery in the last twenty years, with developments in time-lapse photography shown possums eating adult birds, their chicks, and eggs (Sadlier 2000). Sadlier (2000) states that limited studies have been performed on the frequency of possum predation on birds, and their nests, though the incidence of possum predation was found to range from 6%-40%. Clearly the result monitoring of this target pest is highly important, although the relative effect on wildlife by different pests is a need best investigated in specific research experiments.

4.3.8.4 Possum Monitoring Programme – Statistical Reliability

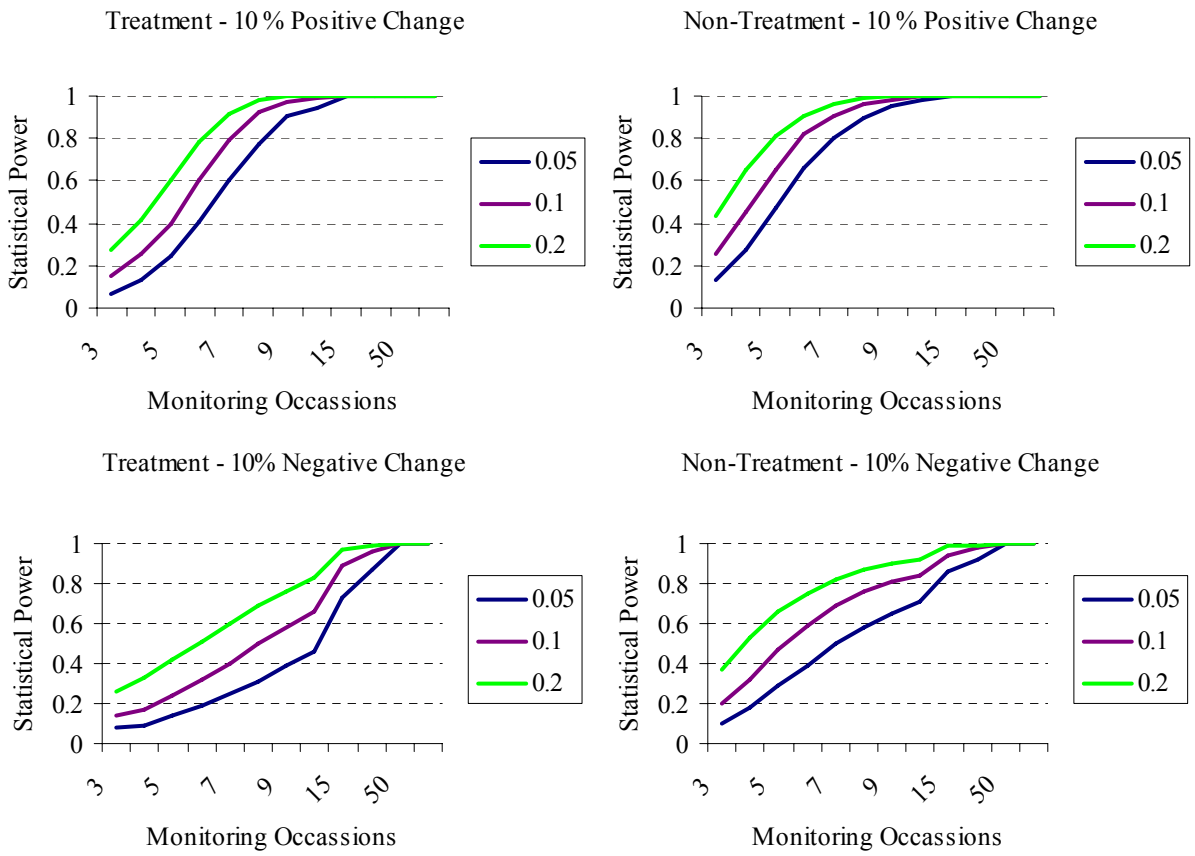

Figure 4.3.8.4.1 Statistical Power for the Initial Possum Monitoring Programme

The calculated statistical power values for the Initial Possum monitoring programme (Total Numbers) at the Treatment site show a very robust design (from initial figures) for a 10% positive and negative change at all alpha levels. Both management regimes show a moderately robust design (reaching a calculated statistical power of 1.00) for all alpha levels. The expected timeframe for the initial Possum monitoring programme (Total numbers: monitored once per year) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to fourteen years for the Treatment site, and seven years for the Non-Treatment sites (for the determination of a negative change).

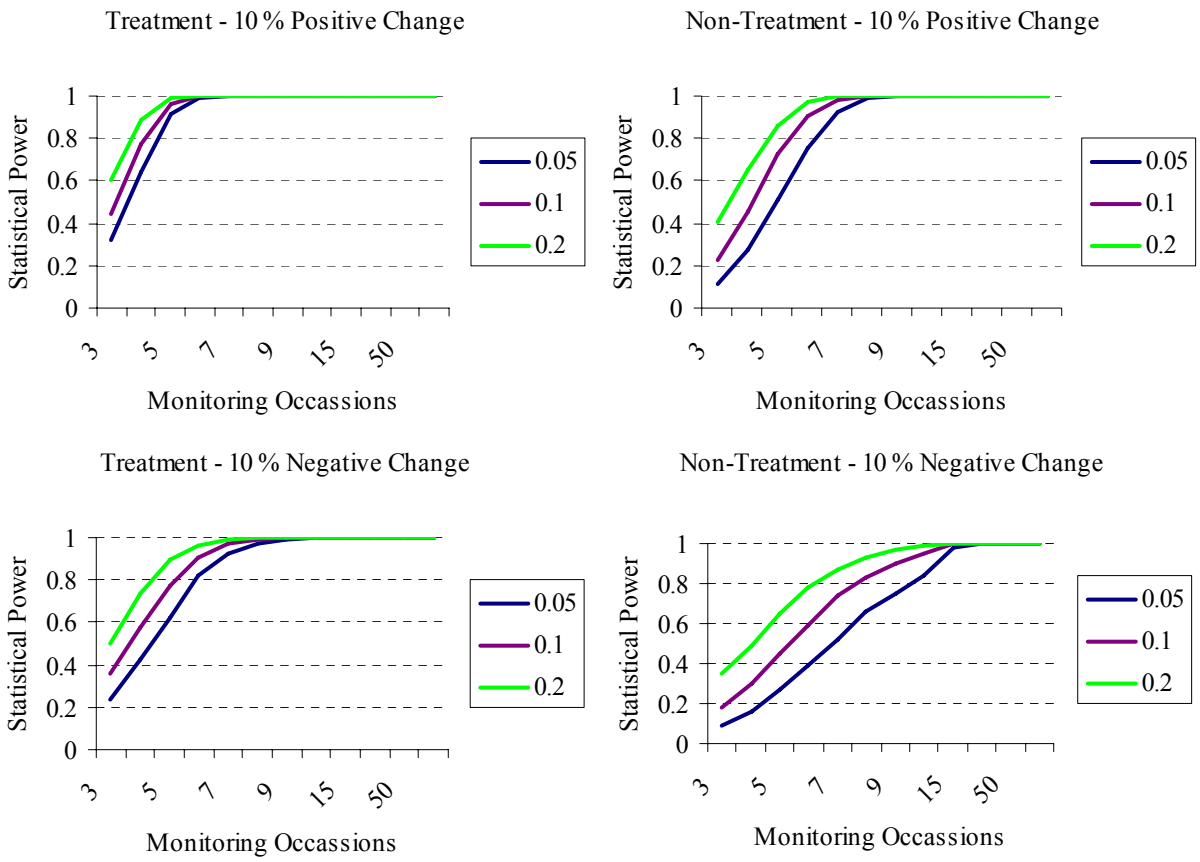

Figure 4.3.8.4.2 Statistical Power for the Current Possum Monitoring Programme

The calculated statistical power values for the Current Possum monitoring programme (Total Numbers) at the Treatment site show a change to very robust design (from initial figures) for a 10% positive and negative change at all alpha levels. Both management regimes show a robust design (reaching a calculated statistical power of 1.00) for all alpha levels. The expected timeframe for the current Possum monitoring programme (Total numbers: monitored once per year) to reach a 'robust' (0.8 Statistical Power; Alpha = 0.2) level equates in this case to six years for the Treatment site, and seven years for the Non-Treatment sites (for the determination of a negative change).

The possum (Trap-Catch Index) monitoring programme design was found to be highly robust, with the change to the current monitoring design (10 lines of twenty traps) cutting

the expected timeframe in half (Appendix: 4, 5). Simulations for the three significance criterion or alpha levels, α (0.05, 0.1, 0.2) reached the 'robust' (0.8 Statistical Power) level for both a positive and negative 10% change over time (up to 100 monitoring occasions), indicating that the monitoring design was reliable for a high level of confidence.

The possum monitoring design has similar issues as that of the rodent monitoring design (for rats). While the trap-catch methodology is an effective method for detecting possums, and the monitoring design is effective for determining the relative abundance at moderate to high densities, monitoring low-density populations (due the effectiveness of the management) has a number of challenges (Beier & Cunningham 1996; Brown 2002). An adaptive two-phase sampling design, using the information obtained in the annual trap-catch monitoring to adapt the sampling or monitoring programme (of the Treatment site) would be very useful for the possum management of BSMIP. By targeting lines where possums have been found, and monitoring those lines immediately again or temporary lines alongside, we would receive a more reliable estimate of possums in those areas (Brown 2003; 2003). This would be very useful as the current possum management in the BSMIP is concerned primarily about controlling 'eruptions' (site specific increases in pest animal numbers), and then immigration by possums. Systematic sampling (Brown & Thomas 2000; Brown 2002) would be useful for the BSMIP, as the entire Treatment and Non-Treatment sites could be covered spatially to obtain a more representative measure of where the possums reside. Advances in the monitoring methodology, such as greater sensitivity to the abundance of animals, increased precision of mean counts, ease of use, and cheaper economics, by the use of wax tags (Thomas *et al.* 2003) for example would confer a greater statistical reliability and hence validity to the monitoring programme, and should be considered for inclusion.

I suggest that repeated measures analysis be used as the general data analysis model for the BSMIP possum monitoring programme, because as with the tree wētā, ground invertebrate, lizard, bird, vegetation, and rodent monitoring programmes, the collection of data (Possum Trap-Catch Index) is 'repeated' spatially over time. Should adaptive sampling be included as a secondary measure, then a second level of analysis should be incorporated such as single factor ANOVA between the trap-catch numbers of the different lines.

4.3.8.5 Possum Monitoring Programme – Conclusion

The BSMIP possum monitoring programme design is effective, balanced, and will fulfil the important need for long-term trend possum monitoring within the BSMIP. It encompasses comparative Non-Treatment sites, and along with the rodent monitoring programme has a valid ("outcome" of plant species composition and abundance) basis monitoring of management. This fulfils Arand & Stephens's (1999) conservation monitoring guidelines, with a direct linkage between results and outcomes in the measurement of conservation projects. The current BSMIP possum monitoring programme is a strong improvement on the initial design. The reduction of possum numbers to very low densities at the Treatment site (Cranwell 2000), raises the question of what is our confidence of determining the trend of possums number due to the ongoing management.

The current possum monitoring programme design was found to be highly robust for possum numbers. It is reliable to detect a trend (+/- 10%) for a high level of confidence (α: 0.05, or 95% CI) upon ten years of its inception. I consider this to be an effective management timeframe to determine the comparative possum control effectiveness of the restoration management, as the techniques available can ensure greater than 80% reduction in possum numbers. Brown (2002) states that for pest monitoring a small α may be preferable to a small β, as falsely concluding a reduction in pest numbers would be more damaging for conservation outcomes than incorrectly concluding an operation has not been successful.

Previous possum monitoring occurred in the years prior to the initial intensive multi-pest species management at areas within the BSMIP, additionally a Before-After-Control-Impact monitoring design occurred for the large scale "knockdown" of the possums at the initiation of the BSMIP. This provides an effective comparative measure of possum result change due to the intensive multi-pest species management. The change to the current monitoring design produced an improvement in statistical power, and coverage throughout the reserves. For example, the monitoring programme at the Treatment site was changed from four lines of 30 (120 traps), to ten lines of twenty (200 traps) supporting Brown & Thomas's (2000) comments that it is better having more shorter lines of leg-hold traps, than a few long lines. Recent improvements in the monitoring methodology of detection could be used to gain a

more precise estimates of possum presence and abundance, which would benefit the project especially should additional and highly susceptible indigenous species be re-introduced or translocated into the Treatment Site.

Similar to the data from the tree wētā, ground invertebrate, lizard, bird, vegetation, rodent, and mustelid monitoring programmes, the statistical analysis on the possum trap-catch data requires consideration of space-time correlation (Millard *et al*. 1985; Buckland *et al*. 1997), which is best addressed by repeated measures analysis (Green 1984; Green 1989; Green 1993; Underwood 1993; Norton 1996; Ribic & Ganio 1996).

4.3.8.6 Possum Monitoring Programme – Recommendations

I recommend that the current core BSMIP possum monitoring programme is retained. If additional trap-catch lines are added, theses should placed in ideally comparative, or at least similar vegetation habitats to that of the Treatment site, so as to enable direct habitat comparisons to be made. Consideration should be given to current advances in the monitoring methodology such as the use of wax tags for monitoring possums. Repeated measures analysis should be used to analyse the possum trap-catch data.

Chapter 5 General Conclusion

5.1 Monitoring Programme Design & Lessons Learnt

"*Monitoring is vital to effective conservation,…it's not particularly difficult, and the rewards are great*" (Walls 1996)

Walls (1996) comment above outlines the importance of monitoring to conservation. Monitoring should confer knowledge, and with deliberate attention to the planning, justification, and objectives of monitoring this can be achieved. The three key components of an effective monitoring programme outlined in this thesis; Guiding Objectives, Biological Relevance, and Statistical Reliability are all inter-linked. A need exists to critically evaluate the reliability, and statistical robustness of conservation monitoring programmes in New Zealand. The Boundary Stream Mainland Island Project is one of a number of examples of the current forefront of conservation management, while also being a showcase for conservation where the public can view, experience, and be involved (if they choose). The BSMIP is a useful project to evaluate its monitoring programmes, though this critical evaluation of monitoring programmes should be a common place activity for all conservation projects.

Hellawell (1991) in the design of monitoring strategies, asks the key questions;

'*what are my objectives*?', and '*what is to be monitored*?"

This thesis is advance on such questions, and a evaluation on the monitoring design, by attempting to answer the question;

'Is the monitoring BSMIP performs going to show anything?'

and the answer is yes, in time, generally the programmes are robust and balanced between Result and Outcome monitoring programmes, although changes need to be made to increase the number of sample units at the Non-Treatment Sites, and the overall lizard monitoring programme requires reassessment. Awareness of the necessity for the long-term scale of conservation monitoring needs to be increased, outlining the balance between biological and statistical reliability timeframes (Appendix 4 & 5), determining when a response due to

conservation management (e.g. around five years for wētā, and at least ten years for vegetation) is reliable to warrant analysis.

5.1.1 Guiding Objectives

Monitoring programmes need to have a prescribed basis (Norton 1996), with an emphasis on *a priori* planning and modelling (Anderson *et al.* 2001). The prescribed basis, or key guiding objective for the BSMIP monitoring programme; is the change in the monitoring index values over time between the different management sites. This would reflect the changes of the indigenous ecosystem in response to the ecosystem restoration work undertaken, such as the management of pest species.

A hierarchical approach to monitoring has been proven to deliver effective guidance for conservation management in New Zealand (Hutcheson *et al.* 1999). The hierarchical approach that I have taken for the evaluation of the BSMIP monitoring programme: Guiding Objectives, Biological Relevance, and Statistical Reliability, can be used to establish a valid monitoring programme for conservation management. The BSMIP monitoring programme is also hierarchical in terms of the scale of inquiry for the analysis of each of the separate monitoring programmes. This hierarchy of inquiry, or "sub-sampling" can substantially reduce variation (Gibbs & Melvin 1997), although cannot be considered as true replication (Eberhardt & Thomas 1991). Sub-sampling is likely to be useful for the BSMIP monitoring programmes in this regard purely for the increase in precision. The hierarchical monitoring approach enables the ground invertebrate pitfall-trap monitoring to investigate a range of parameters – e.g. total insects, total beetles, total *colydiidae*, three sizes of *colydiidae* – total numbers of sizes of orders of insects for each pitfall, each group, and each line (habitat-type), and each reserve and Treatment site.

It is important in conservation monitoring to perform if possible solid "Before" intervention monitoring, that covers a range of annual variation, although baseline monitoring has limited relevance to long-term post operational monitoring (Eberhardt 1976). The BSMIP should be trying to determine the trends in ecosystem health and condition as the two Treatment regimes move "apart" from each other over time, as the before monitoring was only partially comprehensive and the lack of established and repeated pre-operational monitoring is

probably the major shortcoming of the project. The BACI design should be the basic starting point for the design of monitoring programmes. The monitoring within BSMIP has passed the transition from BACI to become trend monitoring. It would be useful to comprehensively outline the result and outcome monitoring programmes over time, which would provide an overall planning methodology to determine the dynamics of these relationships as well as the actual monitoring occasions. This would visually depict the transition from a BACI design – to trend monitoring design for each monitoring programme, and how each level of inquiry of each programme exists within the likely monitoring hierarchy.

5.1.2 Biological Relevance

Monitoring should be able to define some key parameters so that wise management can occur. It should be able not only to direct attention, but to also direct management. Sweetapple & Burns (2002) states that the objectives for sustained pest goat control operations should relate directly to ecosystem health, rather than goat population level. This relationship; management of ecosystem health by sustained pest control, is a key concept that the BSMIP is addressing in the overall result monitoring programme for all the managed mammalian pests. The BSMIP monitoring programme is well balanced between result and outcome monitoring programmes, fulfilling Arand & Stephens (1999) conservation monitoring guidelines, for the measurement of conservation projects. This should advance the comprehension of the ecological mechanisms (Krebs 1991) of a forest ecosystem where intensive multi-pest species management is occurring, as Christensen (2003) has done for the response of wētā populations to such management.

Hayes & Steidl (1997) states that monitoring of populations should concern itself with statistical power analysis, and that the focus of researchers should concentrate on the biological relevant factors of; effect size, rate of change, and how much change is important, especially over the long-term.

Inference is only justified when statistical power is high, or when the confidence interval for the parameter of interest excludes values hypothesized to be biologically significant (Hayes & Steidl 1997). The BSMIP monitoring programme should focus on these three key

parameters, and perform statistical analysis only once the specific monitoring programme (including site, and level) has reached a level of statistical power that is deemed reliable for the hypothesis of interest.

5.1.3 Statistical Reliability

Sedlmeier & Gigerenzer (1989) ask the question;

> '*Do we really need power at all?*',

and go on to answer their own (provocative) question by asking researchers (and I include natural resource managers in the case of conservation) whether they believe that they have to make a decision after an experiment or intervention. In the conservation field, it is of consequence even if nothing is done, a decision has been made, whether a default or not. This is the core premise of monitoring; we monitor so that we can make conservation decisions; if necessary we can perform interventions, e.g. if a threatened species is declining we can make a decision to up our management (control of predators, *etc*), though we need to have some understanding of whether it is necessary or not. Thus it is important to consider the statistical power of any and all experiment or monitoring programme.

The use of power analysis is vital in the planning stages of a survey or monitoring design (Gerard *et al.* 1998), or as this thesis has done, to determined the length of time (monitoring events) required before the survey design becomes statistical robust for a range of alpha levels. Any monitoring programme design must incorporate an adequate number of plots, and be monitored frequently enough to determine trends, especially because of the inherent variability of the population (Gibbs *et al.* 1998). An effective design will reduce the Coefficient of Variation (CV), while efficiently managing the resources to monitor the sample of interest. Because of the statistical power of many ecological methods, long-term studies are essential to measure time trends in ecosystems (Krebs 1990). I expect that the Boundary Stream Mainland Island Project, like other long-term studies will not only show the effect of conservation management (intensive multi-species pest control, and general conservation management) in terms on the actual population changes, but also how the population variability indices changes as well (Pimm & Redfearn 1988). This can only be determined with a statistically sound monitoring programme design like that within the Boundary Stream Mainland Island Project.

The key criteria for the evaluation of the BSMIP monitor programme's statistical reliability; was the determination of a conservative change within a biologically appropriate timeframe. The precision or confidence needs to be set by managers before any monitoring is performed (Warburton 2000). Underwood (1993) notes that a key problem to solve in most environmental research is how to calculate, in advance, the power of any statistical analysis (monitoring design) to detect the biological effects. Lougheed *et al.* (1999) demonstrates how the trends detected by a monitoring programme, and the efficiency of such a programme itself, can be evaluated quantitatively with "retrospective" power analysis (Thomas 1997). This information (from either pilot studies, or preliminary data as used in this thesis) can be used for determining appropriate levels of effort (the monitoring design) dedicated towards a monitoring programme.

Statistical power of the BSMIP monitoring programmes was increased when the alpha, or Type I error level (α), was increased. An alpha level of the range (0.1 – 0.2) would be well-suited to the overall goals of the BSMIP outcome monitoring programmes. It is imperative that we detect any important changes in populations, because as conservationists we need to protect the population, and if the population crashes over the period of a few years, and we missed the start of the event (when extinction/extirpation might possibly have been prevented) because we set our standards for detecting trends too high (by setting our alpha level too low). Consequently, the penalty of possibly being wrong 20% of the time in our statements regarding the presence of "significant change" (Brown & Miller 1998) is well worth the aggravation of sometimes crying wolf. The alternative choice is to let a population slip through the cracks (cracks caused by variability in our counting technique) because we waited too long for the numbers to tell the story. However for Result monitoring, a smaller α value may well be more suited to measure a decline in pest numbers in control operations, as the relative cost of marking a Type I error would be greater for conservation objectives (Brown & Miller 1998; Brown 2002). Mapstone (1995) suggests that the acceptable errors should be set from these two costs. What is apparent is the need to express the relative likelihood of each error, i.e. the expression of the alpha level used, and the statistical power of the monitoring design and/or analysis, so that the reliability and validity of the respective monitoring programme can be established.

The statistical power to detect trends is influenced by the variability of the population data (Gibbs *et al.* 1998; Gerrodette 1987). Power is based on variances within the actual data and this would account for some of the fluctuation seen, particularly in the Non-treatment sites. The BSMIP monitoring programme objective is to detect any important change in the different populations of interest, and a comprehensive and competent monitoring programme should result in more effective conservation actions, by virtue of providing managers with more accurate/legitimate information. The monitoring design should direct the statistical analysis test, an as the BSMIP sampling units (plots, wētā houses, *etc*) are fixed spatially, it is necessary to use a repeated measures analysis (Green 1993; Norton 1996), such as repeated measures ANOVA.

The computer program MONITOR (Gibbs 1995), (if somewhat limited in the ranges of percentage change) proved useful for the "preliminary" investigation into the statistical power of the individual monitoring programmes at the Boundary Stream Mainland Island Project. As MONITOR could only determine statistical power on trends as opposed to the differences between the Treatment regimes, the "true" statistical power of the Boundary Stream Mainland Island Project's monitoring programmes in the comparison of differences between the Treatment and Non-Treatment site data would likely be far more robust. Additional freeware packages such as POWERPACK, GPOWER, TRENDS, can be easily downloaded and used in the design of monitoring programs (Appendix 10). These and other programs that perform power analysis would be useful in further monitoring design analysis, outlining the statistical power of each level of inquiry as related to larger changes in the populations of interest.

5.2 Recommendations for BSMIP

"*Far better an approximate answer to the right question, which is often vague, than an exact answer to the wrong question, which can always be made precise*" (Tukey 1962).

The following are recommendations to the management of the Boundary Stream Mainland Island Project for the improvement of the BSMIP monitoring programme, as well as offering suggestions to the establishment of biodiversity monitoring programmes within the Department of Conservation. Specific recommendations for each monitoring programme are given in Chapter 4; sections 4.3.#.6.

5.2.1 General

- Keep the core BSMIP monitoring programmes (though some may have changes – addressed in Chapter 4: Case Study of BSMIP Monitoring Programmes.

5.2.3 Guiding Objectives

- For the future strategic direction of the Boundary Stream Mainland Island Project to incorporate a landscape-scale restoration focus;

  Protection of the Maungaharuru Range (Boundary Stream as its Heart),

  Protection of the water catchment (from the Range to the Sea).
- Add changes that will provide a benefit of increasing information, such as adaptive designs, systematic designs, and subsidiary monitoring levels, second density estimates (e.g. snap-traps along side tracking tunnels), and surveys that will increase the site knowledge, such as the vegetation composition in gullies, and bluffs.
- Produce a comprehensive outline of the result and outcome monitoring programmes over time should be produced, which would provide an overall planning methodology to determine the dynamics of these relationships as well as the actual monitoring occasions. This would depict the transition from a BACI design – to trend monitoring design for each monitoring programme, and how each level of inquiry of each programme exists within the likely monitoring hierarchy.

5.2.3 Biological Relevance

- Perform a detailed comparative (similarity) analysis of habitat-types (e.g. vegetation composition: using twinspan analysis of habitat-types), and the influence of the relative sizes of each managed site.
- For the calculation of statistical power of a monitoring design; use a conservative level of change e.g. 5 – 10% for threatened, or endangered species, and a greater (moderate) level of change (20 – 50 %) for general monitoring (including pest species).

5.2.4 Statistical Relevance

- Specific recommendations for each monitoring programme are given in the respective recommendation section 4.3.#.6, and include the incorporation of better detection devices, extra monitoring groups, especially in the Non-Treatment sites, wider surveys about the Maungaharuru range site, in similar habitat-types, or (vegetation composition). The only major change is the need to review of the lizard monitoring programme (including objectives, and methodologies – though some direction is given).
- Investigate a range of monitoring designs to ensure that statistical power is maximized (Faith *et al.*1991), and the Coefficient of Variation (CV) is reduced, after preliminary, or pilot studies are completed. Use a conservative level of change (minimum biologically significant effect sizes (Steidl *et al.* 1997) for population abundance/indices *etc*.) for Power analyses.
- A reminder that costs exist for all decisions involving Type I & II errors; e.g. Brown (2002) states that for pest monitoring a small α may be preferable to a small β, as falsely concluding a reduction in pest numbers would be more damaging for conservation outcomes than incorrectly concluding an operation has not been successful. Else increasing alpha (α) is particularly relevant to cases where the cost of Type II errors is much larger than the cost of Type I errors. In a general sense an alpha level range of 0.10 – 0.20, an a statistical power of 0.8+ (80+%) would be well-suited to the goals of a monitoring programme, with a higher statistical power required for more special or threatened biota populations.

- Appendices 1, and 2 summarize the expected timeframe for each monitoring programme to reach a level of statistical robustness (qualified as 0.8+). This should be used as a guide to perform future statistical analyses on the data.
- Use Repeated Measures Analysis as a criteria for statistical tests.
- Report the biological ramifications of making a Type II error (Peterman 1990b).
- Report the Statistical Power (1-β), Alpha level (α), Confidence Interval CI, Coefficient of Variation CV, an error variance (S^2) (Green 1989) values when writing up the analysis of the monitoring programme data.

References

Adams, J. 1995. (Revised 1997). *Boundary Stream Mainland Island Project – Strategic Plan*. Unpublished Report. Hawkes Bay Conservancy. Department of Conservation, Napier.

Adams, J. 1997. Eradication of norway rats from Motu-o-kura. *Ecological Management.* 5: 5-10. Department of Conservation. Wellington.

Aebischer, N. J. 1990. Assessing pesticide effects on non-target invertebrates using long-term monitoring and time-series modelling. *Functional Ecology*. 4: 369-373.

Allen, R. B.; Payton, I. J.; Knowlton, J. E. 1984. Effects of ungulates on structure and species composition in the Urewera forest as shown by exclosures. *New Zealand Journal of Ecology*. 7: 119-130.

Allen, R. B. 1993. *A permanent plot method for monitoring changes in indigenous forests*. Maanaki Whenua – Landcare Research New Zealand Ltd. Christchurch.

Allen, R. B.; Fitzgerald, A. E.; Efford, M. G. 1997. Long-term changes and seasonal patterns in possum (*Trichosurus vulpecula*) leaf diet, Orongorongo valley, Wellington, New Zealand. *New Zealand Journal of Ecology.* 21: 181-186.

Allen, R. B.; Lee, W. G.; Rance, B. D. 1994. Regeneration of indigenous forest after eradication of Norway Rats, Breaksea Island, New Zealand. *New Zealand Journal of Botany*. 32: 429-439.

Allen, R. B.; Payton, I. J.; Knowlton, J. E. 1984. Effects of ungulates on structure and species composition in the Urewera forests as shown by exclosures. *New Zealand Journal of Ecology*.7: 119-130.

Alley, J. C.; Berben, P. H.; Dugdale, J. S.; Fitzgerald, B. M.; Knightbridge, P. I.; Meads, M. J.; Webster, R. A. 2001. Responses of litter-dwelling arthopods and house mice to beech seedling in the Orongorongo valley, New Zealand. *Journal of The Royal Society of New Zealand*. 31: 425-452.

Andersen, A. N.; Sparling, G. P. 1997. Ants as indicators of restoration success: relationship with soil microbial biomass in the Australian seasonal tropics. *Restoration Ecology*. 5: 109-114.

Anderson, D. R. 2001. The need to get the basics right in wildlife studies field studies. *Wildlife Society Bulletin*. 29: 1294-1297.

Anderson, D. R.; Burnham, K. P.; Franklin, A. B.; Gutierrez, R. J.; Forsman, E. D.; Anthony, R. G.; White, G. C.; Shenk, T. M. 1999. A protocol for conflict resolution in analyzing empirical data related to natural resource controversies. *Wildlife Society Bulletin*. 27: 1050-1058.

Anderson, D. R.; Burnham, K. P.; Gould, W. R.; Cherry, S. 2001. Concerns about finding effects that are actually spurious. *Wildlife Society Bulletin*. 29: 311-316.

Anderson, D. R.; Burnham, K. P.; Thompson, W. L. 2000. Null hypothesis testing: problems, prevalence, and an alternative. *Journal of Wildlife Management*. 64: 912-923.

Anderson-Cook, C. M. 1998. Designing a first experiment: a project for design of experiment courses. *The American Statistician*. 54: 338-346.

Andrén, H. 1996. Population responses to habitat fragmentation: statistical power and the random sample hypothesis. *Oikos*. 76: 235-242.

Anon. 1994. *Conservation management strategy: for the Hawkes Bay conservancy*. Department of Conservation. Napier.

Anon. 1997. *The state of New Zealand's environment report*. Ministry for the Environment. Wellington.

Anon. 1999. *Restoring the dawn chorus: Department of Conservation strategic business plan 1998-2002*. Department of Conservation, Wellington.

Anon. 2000. *Boundary Stream Mainland Island Project First Annual Report: 1996-1998*. Department of Conservation. Napier.

Anon.1. 2000. *Boundary Stream Mainland Island: Nature Restoration Project: Restoring our Past – Revitalising our Future*. Unpublished Report. Department of Conservation. Gisborne.

Anon. 2. 2000. *The New Zealand Biodiversity Strategy – our chance to turn the tide*. Department of Conservation & Ministry for the Environment. Wellington.

Anon. 2002. *Restoring the dawn chorus: Department of Conservation strategic business plan 2002-2005*. Department of Conservation, Wellington.

Arand, J.; Stephens, T. 1998. *Measuring Conservation Management Projects: definitions, principles and guidelines*. Department of Conservation, Wellington.

Armstrong, D. P.; Raeburn, E. H.; Powlesland, R. G.; Howard, M.; Christensen, B.; Ewen, J. G. 2002. Obtaining meaningful comparisons of nest success: data from New Zealand robin (*Petroica australis*) populations. *New Zealand Journal of Ecology*. 26: 1-13.

Aronson, J.; le Floc'h, E. 1996. Hierarchies and landscape history: dialoguing with Hobbs & Norton. *Restoration Ecology.* 4: 327-333.

Atkinson, I. A. E. 2001. Introduced mammals and models for restoration. *Biological Conservation*. 99: 81-96.

Austin, M. P. 1981. Permanent quadrats: an interface for theory and practice. *Vegetatio*. 46: 1-10.

Badan, D. 1986. Diet of the house mouse (*Mus musculus* L.) in two pine and kauri forest. *New Zealand Journal of Ecology*. 9: 137-141.

Beier, P.; Cunningham, S. C. 1996. Power of track surveys to detect changes in cougar populations. *Wildlife Society Bulletin*. 24: 540-546.

Bellingham, P. J.; Allan, C. A. 2002. Forest regeneration and the influences of white-tailed deer (*Odocoileus virginianus*) in cool temperate New Zealand rain forests. *Forest Ecology and Management*. 2002: 1-16.

Bellingham, P. J.; Wiser, S. K.; Coomes, D.; Dunningham, A. 2000. Review of permanent plots for long-term monitoring of New Zealand's indigenous forests**.** *Science for Conservation.* 151. Department of Conservation. Wellington.

Benedetti-Cecchi, L. 2001. Beyond BACI: Opitmization of environmental sampling designs through monitoring and simulation. *Ecological Applications.* 11: 783-799.

Benn, G. A.; Kemp, A. C.; Begg, K. S. 1995. The distribution and trends of the saddlebilled stork *Ephippiorhynchus senegalensis* population in South Africa. *South African Journal of Wildlife Research*. 25: 98-105.

Berkson, J. 1942. Tests of significance considered as evidence. *Journal of the American Statistical Association*. 37: 325-335.

Berger, J. J. 1991. A generic framework for evaluating complex restoration and conservation projects. *The Environmental Professional*. 13: 254-262.

Berger, J. O.; Berry, D. A. 1988. Statistical analysis and the illusion of objectivity. *American Scientist*. 76: 159-165.

Bernstein, B. B.; Zalinski, J. 1983. An optimal sampling design and power tests for environmental biologists. *Journal of Environmental Management*. 16: 35-43.

Berry, C. J. J. 1999. European hedgehogs (*Erinaceus eurpaeus* L.) and their significance to the ecological restoration of Boundary Stream Mainland Island, Hawkes Bay. Unpublished Masters of Conservation Science Thesis, Victoria University, Wellington.

Bishop, M-A.; Meyers, P. M.; McNeley, P. F. 2000. A method to estimate migrant shorebird numbers on the copper river delta, Alaska. *Journal of Field Orntihology.* 71: 627-637.

Blackwell, G. L. 2000. *An investigation of the factors regulating house mouse (Mus musculus) and ship rat (Rattus rattus) population dynamics in forest ecosystems at*

Lake Waikaremoana, New Zealand. Unpublished PhD thesis, Massey University, Palmerston North, New Zealand.

Blackwell, G. L.; Potter, M.; McLennan, J. A. 2002. Rodent density indices from tracking tunnels, snap-traps, and Fenn traps: do they tell the same story? *New Zealand Journal of Ecology.* 26: 43-52.

Block, W. M.; Franklin, A. B.; Ward, J. P. Jr.; Ganey, J. L.; White, G. C. 2001. Design and implementation of monitoring studies to evaluate the success of ecological restoration on wildlife. *Restoration Ecology*. 9: 293-303.

Bohner, M.; Peterson, A. C. 2001. *Dynamic equations on time scales: an introduction with applications.* Birkhauser.

Bohner, M.; Peterson, A. C. 2002. *Advances in dynamic equations on time scales.* Birkhauser.

Brockie, R. E. 1992. *A living New Zealand forest*. Bateman. Auckland.

Brockie, R. E. 1990. *European Hedgehog*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Brown, J. A. 2002. A review of monitoring low density animal populations. *Research Report UCDMS2002/2*. Biomathematics Research Centre. University of Canterbury. Christchurch.

Brown, J. A. 2003. *Adaptive sampling.* In. Thompson. W. L. (Ed). *Sampling rare and elusive species.* (Draft).

Brown, J. A.; Miller, C. J., 1998. Monitoring stoat *Mustela erminea* control operations: power analysis and design. *Science for Conservation.* 96. Department of Conservation. Wellington.

Brown, K. P.; Moller, H.; Innes, J.; Alterio, N. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand Forest. *New Zealand Journal of Ecology*. 20: 271-275.

Brown, R.; Norton, D. A. 2001. *Experimental design and ecosystem restoration: Rotoiti nature recovery project treatment and non-treatment vegetation*. Unpublished Report. University of Canterbury. Christchurch.

Brown, J. A.; Thomas, M. D. 2000. Residual trap-catch methodology for low-density possum populations. *Research report UCDMS 2000/6*. Biomathematics Research Centre. University of Canterbury. Christchurch.

Buckland, S. T.; Anderson, D. R.; Burnham, K. P.; Laake, J. L. 1993. *Distance Sampling: estimating abundance of biological populations.* Chapman and Hall. London.

Buckland, S. T.; Burnham, K. P.; Augustin, N. H. 1997. Model selection: an integral part of inference. *Biometrics*. 53: 603-618.

Burnham, K. P.; Anderson, D. R. 2001. Kullbuck-Leibler information as a basis for strong inference in ecological studies. *Wildlife Research*. 28: 111-120.

Cairns, J. Jr. 1991. The status of the theoretical and applied science of restoration ecology. *The Environmental Professional*. 13: 186-194.

Cairns J, Jr.; McCormick, P. V.; Niederlehner, B. R. 1993. A proposed framework for developing indicators of ecosystem health. *Hydrobiologia*. 263: 1-44.

Caley, P.; Morriss, G. 2001. Summer/autumn movements, mortality rates and density of feral ferrets (*Mustela furo*) at a farmland site in North Canterbury, New Zealand. *New Zealand Journal of Ecology*. 25: 53-60.

Campbell, D. J. 2002. Changes in numbers of woody plant seedlings on Kapiti Island after rat eradication. *Science for Conservation*. 193. Department of Conservation. Wellington.

Carpenter, S. R. 1990. Large-scale perturbations: opportunities for innovations. *Ecology*. 71: 453-463.

Carver, R. P. 1978. The case against statistical significance testing. *Harvard Educational Review*. 48: 378-399.

Cassey, P. 1999. Estimating animal abundance by distance sampling techniques. *Conservation Advisory Notes*. 237. Department of Conservation. Wellington.

Caughley, G.; Sinclair, A. 1994. *Wildlife Ecology and Management*. Blackwell Science. Cambridge.

Chamberlin, T. C. 1995. The method of multiple working hypotheses. *The Journal of Geology.* 103: 349-354.

Cherry, S. 1998. Statistical tests in publications of the wildlife society. *Wildlife Society Bulletin*. 26: 947-953.

Choquenot, D.; Parkes, J. 2001. Setting thresholds for pest control: how does pest density affect resource viability. *Biological Conservation*. 99: 29-46.

Choquenot, D.; Ruscoe, W. A.; Murphy, E. 2001. Colonization of new areas by stoats: time to establishment and requirements for detection. *New Zealand Journal of Ecology*. 25: 83-88.

Christensen, B. R. 1996. *Possum management at Boundary Stream Mainland Island Project*. Internal Report. Department of Conservation. Napier.

Christensen, B. R. 1999. Invertebrate monitoring for the Boundary Stream Mainland Island Project. *The Weta*. 21: 12-14.

Christensen, B. R. 2000. *Introduction to The Boundary Stream Mainland Island Project*. In, Anon. 2000. *Boundary Stream Mainland Island Project First Annual Report: 1996-1998*. Department of Conservation. Napier.

Christensen, B. R. 2001. *Invertebrate monitoring*. In, Anon. 2001. *Boundary Stream Mainland Island Project Biannual Report: 1998-2000*. Unpublished report. Department of Conservation. Napier.

Christensen, B. R. 2002. *Vegetation monitoring*. In, Anon. 2002. *Boundary Stream Mainland Island Project Annual Report: 2000-2001*. Unpublished report. Department of Conservation. Napier.

Christensen, B. R. 2002a. New and most northern records of Hawke's Bay Tree Wētā (*Hemideina trewicki*) (Orthoptera: Anostosmatidae). *The Weta.* 24: 18-19.

Christensen, B. R. 2003. *Impact of intensive pest mammal control on Tree wētā (Hemideina spp.) (Orthoptera: Anostosmatidae) occupancy of artificial roosts at Boundary Stream Mainland Island Project, Maungaharuru Range, New Zealand*. Unpublished Report. Department of Conservation. Rotorua.

Clapperton, B. K. 2001. Advances in New Zealand Mammalogy: Feral ferret. *Journal of the Royal Society of New Zealand*. 31: 185-203.

Clewell, A.; Rieger, J. P. 1997. What practitioners need from restoration ecologists. *Restoration Ecology*. 5: 350-354.

Clout, M. N. 1980. Ship rats (*Rattus rattus* L.) in a *Pinus radiata* plantation. *New Zealand Journal of Ecology*. 3: 141-145.

Cochrane, H. C.; Norton, D. A.; Miller, C. J.; Allen, R. B. 2003. Brushtail possum (*Trichosurus vulpecula*) diet in a north Westland mixed-beech (*Nothofagus*) forest. *New Zealand Journal of Ecology*. 27: 61-66.

Cohen, J. 1988. *Statistical power analysis for the behavioural sciences*. 2nd ed. Academic Press, Inc. New York.

Cohen, J. 1992. A power primer. *Psychological Bulletin*. 112: 155-159.

Cohen, J. 1994. The earth is round ($p < .05$). *American Psychologist*. 49: 997-1003.

Cowan, P. E. 1990. *Brushtail Possum*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Cowan, P. E. 2001. Advances in New Zealand Mammalogy: Brushtail Possum. *Journal of the Royal Society of New Zealand*. 31: 15-29.

Cranwell, S. D. 2000. *Possum management*, In Anon. 2000. *Boundary Stream Mainland Island Project First Annual Report: 1996-1998*. Department of Conservation. Napier.

Crisp, P. N.; Dickinson, K. J. M.; Gibbs, G. W. 1998. Does native invertebrate diversity reflect native plant diversity? a case study from New Zealand and implications for conservation. *Biological Conservation*. 83: 209-220.

Crosby, T. K. 1992. Motu-o-Kura (Bare Island): report on invertebrates in pitfall traps. *Conservation Advisory Science Notes*. 136. Department of Conservation. Wellington.

Cuthbert, R.; Sommer, E.; Davis, L. S. 2000. Seasonal variation in the diet of stoats in a breeding colony of Hutton's shearwaters. *New Zealand Journal of Zoology. 27*: 367-373.

Cutten, H. N. C. 1994. *Geology of the middle reaches of the Mohaka river*. *Institute of Geological & Nuclear Sciences geological map 6*. Institute of Geological & Nuclear Sciences Ltd, Lower Hutt, New Zealand.

Dallimeier, F.; Comiskey, J. A. 1998. *Forest biodiversity assessment, monitoring, and evaluation for adaptive management*. In Dallimeier, F., Comiskey, J. A. (Eds) 1998. *Forest biodiversity research, monitoring and modelling: conceptual background and old world case studies*. Man and the biosphere series, Volume 20. UNESCO. Paris and Parthenon publishing, Carnforth.

Daniel, M.J. 1973. Seasonal diet of the ship rat in lowland forest in New Zealand. *Proceedings of the New Zealand Ecological Society*. 20: 21-30.

Dawson, D. G.; Bull, P. C. 1975. Counting birds in New Zealand Forests. *Notornis*. 22: 101-109.

Dawson, J. 1988. *Forest vines to snow tussocks: the story of New Zealand plants*. Victoria University Press. Wellington.

Daugherty, C. H.; Towns, D. R.; Atkinson, I. A. E.; Gibbs, G. W. 1990. The significance of the biological resources of New Zealand islands for ecological restoration. In. Towns, D. R.; Daugherty, C. H.; Atkinson, I. A. E. (Eds). Ecological restoration of New Zealand islands. *Conservation sciences publication*. 2: 9-21.

de Bres, J. 2002. The Department of Conservation's Statement of Intent (SOI). *Public Sector: the publication of the New Zealand Institute of Public Administration.* 25: 6-9.

Diamond, J. 1987. *Reflections on goals and on the relationship between theory and practice*. In Jordan III, W., Gilpin, M. E., Aber, J. D. Eds. *Restoration ecology: a synthetic approach to ecological research*. Cambridge University Press. Cambridge.

Dickinson, K. L. M.; Mark, A. F.; Lee, W. G. 1992. Long-term monitoring of non-forest communities for biological conservation. *New Zealand Journal of Botany* 30: 163-179.

Dixon, P. M.; Garrett, K. A. 1993. *Statistical issues for field experimenters*. In. Kendall, R. J.; Lacher, T. E. Jr. *Wildlife Toxicology and Population Modelling: Integrated Studies of Agroecosystems*. Lewis Publishers. Boca Raton.

Droege, S. 1999. *Power analysis of monitoring programmes: designing effective surveys*. Patuxent Wildlife Research Center. United States Geological Survey. Web-site. http://www.mp1-pwrc.usgs.gov/powcase/index.html.

Druce, A. P. 1985. (Revised; 1986, 1989). *Indigenous Plant Species List for Boundary Stream Scenic Reserve.* Unpublished Report. Department of Scientific and Industrial Research. Havelock North.

Duffey, E. 2001. Introduced pest species and biodiversity conservation in New Zealand. *Biological Conservation*. 99: 1.

Eberhardt, L. L. 1976. Quantitative ecology and impact assessment. *Journal of Environmental Management*. 4: 27-70.

Eberhardt, L. L. 1978. Appraising variability in population studies. *Journal of Wildlife Management*. 42: 207-238.

Eberhardt, L. L.; Thomas, J. M. 1991. Designing environmental field studies. *Ecological Monographs*. 61: 53-73.

Eckblad, J. W. 1991. How many samples should be taken? *Bioscience*. 41: 346-348.

Edwards, E. F.; Perkins, P. C. 1992. Power to detect linear trends in dolphin abundance: estimates from tuna-vessel observer data, 1975-89. *Fishery Bulletin*. 90: 625-631.

Efron, B.; Tibshirani, R. 1991. Statistical data analysis in the computer age. *Science*. 253: 390-396.

Ehrenfeld, J. G.; Toth, L. A. 1997. Restoration ecology and the ecosystem perspective. *Restoration Ecology*. 5: 307-317.

Enge, K. M. 2001. The pitfalls of pitfall traps. *Journal of Herpetology*. 35: 467-478.

Engeman, R. M.; Pipas, M. J.; Gruver, K. S.; Bourassa, J.; Allen, L. 2002. Plot placement when using a passive tracking index to simultaneously monitor multiple species of animals. *Wildlife Research.*29: 85-90.

Ewel, J. J. 1987. *Restoration is the ultimate test of ecological theory*. In *Restoration ecology: a synthetic approach to ecological research.* Jordan III W, R.; Gilpin, M, E.; Aber, J. D. Eds. Cambridge University Press. Cambridge.

Fairweather, P. G. 1991. Statistical power and design requirements for environmental monitoring. *Australian Journal of Marine and Freshwater Research*. 42: 555-567.

Faith, D. P.; Humphrey, C. L.; Dostine, P. L. 1991. Statistical power and BACI designs in biological monitoring: comparative evaluation of measures of community dissimilarity based on benthic macroinvertebrate communities in rockhole mine creek, northern territory, Australia. *Australian Journal of Marine and Freshwater Research*. 42: 589-602.

Finlayson, C. M.; Eliot I. 2001. Ecological assessment and monitoring of coastal wetlands in Australia's wet-dry tropics: a paradigm for elsewhere? *Coastal Management.* 29: 105-115.

Fisher, B. L. 1998. Insect behaviour and ecology in conservation: preserving functional species interactions. *Annals of the Entomological Society of America*. 91: 155-158.

Fitzgerald, B. M. 1990. *House Cat*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Fitzgerald, B. M.; Daniel, M. J.; Fitzgerald, A. E.; Karl. B. J.; Meads, M. J.; Notman, P. R. 1996. Factors affecting the numbers of house mice (*Mus musculus*) in hard beech (*Nothofagus truncata*) forest. *Journal of the Royal Society of New Zealand*. 26: 237-249.

Forbes, L. S. 1990. A note on statistical power. *Auk*. 107: 438-439.

Forman, R. T. T. 1995. *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge University Press. Cambridge.

Freeman, A. B. 1997. Comparative ecology of two *Oligosoma* skinks in coastal Canterbury: a contrast with Central Otago. *New Zealand Journal of Ecology*. 21: 153-160.

Gales, R. P. 1982. Age- and sex-related differences in diet selection by *Rattus rattus* on Stewart Island, New Zealand. *New Zealand Journal of Zoology*. 9: 463-466.

Gaze, P. 1994. *Rare and endangered New Zealand birds: conservation and management*. Canterbury University Press. Christchurch.

Gerber, L. R.; DeMaster, D. P.; Roberts. S. P. 2000. Measuring success in conservation. *American Scientist*. 88: 316-325.

Gerard, P. D.; Smith, D. R.; Weerakkody, G. Limits of retrospective power analysis. *Journal of Wildlife Management*. 62: 801-807.

Gerrodette, T. 1987. A power analysis for detecting trends. *Ecology*. 68: 1368-1372.

Gerrodette, T. 1991. Models for power of detecting trends-a reply to Link and Hatfield. *Ecology*. 72: 1889-1892.

Gibbs, G. 1998. *New Zealand Wētā.* Reed. Auckland.

Gibbs, J. P. 1995. MONITOR 6.3: Power analysis for population monitoring program. Computer Programme.

Gibbs, J. P.; Droege, S.; Eagle, P. 1998. Monitoring populations of plants and animals. *Bioscience*. 48: 935-941.

Gibbs, J. P., Melvin, S. C. 1997. Power to detect trends in waterbird abundance with call-response surveys. *Journal of Wildlife Management.* 61: 1262-1267.

Gill, B. J.; Whitaker, A. H. 1996. *New Zealand frogs & reptiles*. David Bateman Limited. Auckland.

Gillies, C. A. 2001. Advances in New Zealand mammalogy. *Journal of the Royal Society of New Zealand*. 31; 205-218.

Gillies, C.; Murphy, E. 1997. *Mustelids as conservation pests, control and research perspectives*. In. *Seminar proceedings on possum and mustelid control research.* National Possum Control Agencies. Wellington.

Girardet, S. A. B.; Veitch, C. R.; Craig, J. L. 2001. Bird and rat numbers on Little Barrier Island, New Zealand, over the period of cat eradication 1976-1980. *New Zealand of Journal of Zoology*. 28: 13-30.

Goldsmith, B. Ed. 1991. *Monitoring for conservation and ecology*. Chapman & Hall. London.

Goodman, S. N.; Berlin, J. A. 1994. The use of predicted confidence intervals when planning experiments and the misuse of power when interpreting results. *Annals of Internal Medicine*. 121: 200-206.

Graafhuis, R. B. 2001. *Stratigraphy and sedimentology of pliocene strata in the forearc basin (Waikoau and Waikari river catchments), northern Hawkes Bay*. Unpublished Masters of Science Thesis. University of Waikato. Hamilton.

Gram, W. K.; Sork, V. L.; Marquis, R. J.; Le Corff, J.; Lill, J.; Renken, R. B.; Clawson, R. L.; Fantz, D. K.; Faaborg, J.; Porneluzi, P. A. 2001. Evaluating the effects of ecosystem management: a case study in a missouri ozark forest. *Ecological Applications*. 11: 1667-1679.

Grant, P. J. 1996. *Hawke's Bay forests of yesterday: a description and interpretation*. Patrick J. Grant. Havelock North.

Green, C. 1996. Survey and monitoring techniques for insects. *Ecological Management*. 4: 73-90. Department of Conservation.

Green, R. H. 1979. *Sampling design and statistical methods for environmental biologists*. John Wiley & Sons. New York.

Green, R. H. 1984. Statistical and non-statistical considerations for environmental monitoring studies. *Environmental Monitoring and Assessment*. 4: 293-301.

Green, R. H. 1989. Power analysis and practical strategies for environmental monitoring. *Environmental Research*. 50: 195-205.

Green, R. H, 1993. Application of repeated measures designs in environmental impact and monitoring studies. *Australian Journal of Ecology*. 18: 81-98.

Green, R. H. 1994. *Aspects of power analysis in environmental monitoring*. In. Fletcher, D. J.; Manly, B. F. J. Eds. 1994. *Statistics in ecology and environmental monitoring*. University of Otago Press, Dunedin.

Green, R. H.; Montagna, P. 1996. Implications for monitoring: study designs and interpretation of results. *Canadian Journal of Aquatic Sciences*. 53: 2629-2636.

Greenslade, P. J. M. 1964. Pitfall trapping as a method for studying populations of carabidae (Coleoptera). *Journal of Animal Ecology*. 33: 301-310.

Greenwood, J. J. D. 1993. Statistical power. *Animal Behaviour.* 46: 1011.

Gryska, A. D.; Hubert, W. A.; Gerow, K. G. 1997. Use of power analysis in developing monitoring protocols for the endangered Kendall Warm Springs dace. *North American Journal of Fisheries Management*. 17: 1005-1009.

Guthrie-Smith, H. 1953. *Tutira: the story of a New Zealand sheep station*. William Blackwood & Sons Ltd. Edinburgh.

Hairston, N. G. 1989. *Ecological experiments: purpose, design, and execution*. Cambridge University Press. Cambridge.

Hansen, J. D. 1988. Trapping methods for rangeland insects in burned and unburned sites: a comparison. *Great Basin Naturalist*. 48: 383-387.

Hargrove, W. W.; Pickering, J. 1992. Pseudoreplication: a *sine qua non* for regional ecology. *Landscape Ecology*. 6: 251-258.

Hatfield, J. S.; Gould IV, W. R.; Hoover, B. A.; Fuller, M. R.; Lindquist, E. L. 1996. Detecting trends in raptor counts: power and Type I error rates of various statistical tests. *Wildlife Society Bulletin*. 24: 505-515.

Haw, J. M.; Clout, M. N.; Powelsland, R. G. 2001. Diet of moreporks (*Ninox novaeseelandiae*) in Pureora forest determined from prey remains in regurgitated pellets. *New Zealand Journal of Ecology*. 25: 61-67.

Hayes, J. P. 1987. The positive approach to negative results in toxicology studies. *Ecotoxicology and Environmental Safety.*14: 73-77.

Hayes, J. P.; Steidl, R. J. 1997. Statistical power analysis and amphibian population trends. *Conservation Biology*.11: 273-275.

Heather, B.; Robertson, H. 1996. *The field guide to the New Zealand Birds*. Viking. Auckland.

Hellawell, J. M. 1991. *Development of a rationale for monitoring*. In Goldsmith, B. Ed. 1991. *Monitoring for Conservation and Ecology*. Chapman & Hall. London.

Hendra, R. 1999. Seasonal abundance patterns and dietary preferences of hedgehogs at Trounson Kauri Park. *Conservation Advisory Science Notes*. 267. Wellington, Department of Conservation.

Hinds, W. T. 1984. Towards monitoring of long-term trends in terrestrial ecosystems. *Environmental Conservation*. 11: 11-18.

Hobbs, R. J.; Harris, J. A. 2001. Restoration ecology: repairing the earth's ecosystems in the new millennium. *Restoration Ecology*. 9: 239-246.

Hobbs, R. J.; Norton, D. A. 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology.* 4: 93-110.

Holling, C. S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs*. 62: 447-502.

Hooker, S.; Innes, J. 1995. Ranging behaviour of forest-dwelling ship rats, *Rattus rattus*, and effects of poisoning with brodifacoum. *New Zealand Journal of Zoology*. 22: 291-304.

Hutcheson, J. 1999. Characteristics of Mapara insect communities as depicted by Malaise trapped beetles: changes with time and animal control. *Science for Conservation*. 135. Department of Conservation. Wellington.

Hutcheson, J.; Walsh, P.; Given, D. 1999. Potential value of indicator species for conservation and management of New Zealand terrestrial communities. *Science for Conservation*. No. 109. Department of Conservation. Wellington.

Innes, J. 1979. Diet and reproduction of ship rats in the northern Tararuas. *New Zealand Journal of Ecology*. 2: 85-86.

Innes, J. G. 1990. *Ship Rat*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Innes, J. 2001. Advances in New Zealand mammalogy 1990-2000: european rats. *Journal of the Royal Society of New Zealand*. 31: 111-125.

Innes, J. G.; Skipworth, J. P. 1983. Home ranges of ship rats in a small New Zealand forest as revealed by trapping and tracking. *New Zealand Journal of Zoology*. 10: 99-110.

Jedrzejewski, W.; Jedrzejewski, B.; Szymura, L. 1995. Weasel population response, home range and predation on rodents in a deciduous forest in Poland. *Ecology*. 76: 179-197.

Jamieson, I. G.; Forbes, M. R.; McKnight, E. B. 2000. Mark-recapture study of mountain stone wētā *Hemideina maori* (Orthoptera: Anostostomatidae) on rock tor 'islands'. *New Zealand Journal of Ecology*. 24: 209-214.

Johnson, D. H. 1999. The insignificance of statistical significance testing. *Journal of Wildlife Management*. 63: 763-772.

Jones, D.; Matloff, N. 1986. Statistical hypothesis testing in biology: a contradiction in terms. *Journal of Economic Entomology*. 79: 1156-1160.

Kareiva, P., Bergelson, J. 1997. The nuances of variability: beyond mean square error and platitudes about fluctuating environments. *Ecology*. 78: 1299-1300.

Kendall, K. C.; Metzgar, L. H.; Patterson, D. A.; Steele, B. M. 1992. Power of sign surveys to monitor population trends. *Ecological Applications*. 2: 422-430.

Kenkel, N. C.; Juhász-Nagy, P.; Podani, J. 1989. On sampling procedures in population and community ecology. *Vegetatio*. 83: 195-207.

Keesing, V.; Wratten, S. D. 1998. Indigenous invertebrate components in ecological restoration in agricultural landscapes. *New Zealand Journal of Ecology.* 22: 99-104.

King, C. M. 1984. *Immigrant Killers: introduced predators and the conservation of birds in New Zealand.* Oxford University Press. Auckland.

King, C. M. 1990. *Stoat*. In. *The Handbook of New Zealand mammals*. King, C. M. Ed. Oxford University Press. Auckland.

King, C. M. 1990a. *Weasel*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

King, C. M.; Edgar, R. L. 1977. Techniques for trapping and tracking stoats (*Mustela erminea*); a review, and a new system. *New Zealand Journal of Zoology*. 4: 193-212.

King, C. M.; Griffiths, K.; Murphy, E. C. 2001. Advances in New Zealand Mammalogy. *Journal of the Royal Society of New Zealand*. 31: 165-183.

King, C.M.; Moody, J.E. 1982. The biology of the stoat (*Mustela erminea*) in the National Parks of New Zealand. *New Zealand Journal of Zoology*. 9: 57-80.

King, S.; McRitchie, L. 2001. *Rodent Management*. In, Anon. 2001. *Boundary Stream Mainland Island Project Biannual Report: 1998-2000*. Unpublished report. Department of Conservation. Napier.

King, S. 2002. *Rodent Management*. In, Anon. 2002. *Boundary Stream Mainland Island Project Biannual Report: 2000-2001*. Unpublished report. Department of Conservation. Napier.

Kondolf, G. M. 1995. Five elements for effective evaluation of stream restoration. *Restoration Ecology*. 3: 133-136.

Kraemer, H. C.; Thiemann, S. 1987. *How many subjects?: statistical power analysis in research*. Sage Publications. Newbury Park.

Krebs, C. J. 1991. The experimental paradigm and long-term population studies. *Ibis*. 133: (1, Supplement): 3-8.

Ladley, J. J.; Kelly, D. 1996. Dispersal, germination and survival of New Zealand mistletoes (Loranthaceae): dependence on birds. *New Zealand Journal of Ecology*. 20: 69-79.

Ladley, J. J.; Kelly, D.; Robertson, A. W. 1997. Explosive flowering, nectar production, breeding systems, and pollinators of New Zealand mistletoes (Loranthaceae). *New Zealand Journal of Botany*. 35: 345-360.

Lavers, R. B.; Clapperton, B. K. 1990. *Ferret*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Lawless, P.; Stephens, T. 1996. *The task of conserving biodiversity in New Zealand*. In *Biodiversity: papers from a seminar series on biodiversity*. Science and Research Division, Department of Conservation, Wellington.

Lawton, J. H. 1997. The science and non-science of conservation. *Oikos*. 79: 3-5.

Lawton, J. H. 1999. Are there general laws in ecology? *Oikos*. 84: 177-192.

Likens, G. E. 1985. An experimental approach for the study of ecosystems. *Journal of Ecology*. 73: 381-396.

Lindley, S. T.; Mohr, M.; Prager, M. H. 2000. Monitoring protocol for Sacramento River winter chinook salmon, *Oncorhynchus tshawytscha*: application of statistical power analysis to recovery of an endangered species. *Fishery Bulletin*. 98: 759-766.

Link, W. A.; Barker, R. J.; Sauer, J. R.; Droege, S. 1994. Within-site variability in surveys of wildlife populations. *Ecology*. 75: 1097-1108.

Link, W. A.; Nichols, J. D. 1994. On the importance of sampling variance to investigations of temporal variation in animal population size. *Oikos* 69:3 539-544.

Lipsey, M. W. 1990. *Design sensitivity: statistical power for experimental research*. Sage Publications. Newbury Park.

Lord, J. M.; Norton, D. A. 1990. Scale and the spatial concept of fragmentation. *Conservation Biology.* 4: 197-202.

Lougheed, L. W.; Breault, A.; Lank, D. B. 1999. Estimating statistical power to evaluate ongoing waterfowl population monitoring. *Journal of Wildlife Management.* 63: 1359-1369.

McArdle, B. H. 1996. Levels of evidence in studies of competition, predation and disease. *New Zealand Journal of Ecology*. 20: 221-266.

McBride, G. B.; Loftis, J. C.; Adkins, N. C. 1993. What do significance tests really tell us about the environment? *Environmental Management*. 17: 423-432.

McEwen, W. M. (Ed). 1987. *Ecological regions and districts of New Zealand*. New Zealand Biological Resources Centre. Publication No. 5: 2. Department of Conservation. Wellington.

McIlroy, J. C. 1990. *Feral Pig*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

McIlroy, J. C. 2001. Advances in New Zealand Mammalogy: Feral Pig. *Journal of The Royal Society of New Zealand*. 31: 225-231.

McKinlay, B. 2001. Counting terrestrial bird species in mixed habitats: an assessment of relative conspicuousness. *Notornis*. 48: 47-53.

McLennan, J. 1997. Ecology of brown kiwi and cause of population decline in Lake Waikaremoana catchment. *Conservation Advisory Science Notes.* 167. Department of Conservation. Wellington.

McLennan, J.; Potter, M. A.; Robertson, H. A.; Wake, G. C.; Colbourne, R.; Dew, L.; Joyce, L.; McCAnn, J. A.; Miles, J.; Millar, P.; Reid, J. 1996. Role or predation in the decline of the kiwi *Apteryx* spp. in New Zealand. *New Zealand Journal of Ecology*. 20:27-35.

McLeod, B. J.; Thompson, E. G. 2002. Predation on house sparrows (*Passer domesticus*) and hedge sparrows (*Prunella modularis)* by brushtail possums (*Trichosurus vulpecula*) in captivity. *Notornis*. 49: 95-99.

McRitchie, L. 2000. *Rodent management*, In Anon. 2000. *Boundary Stream Mainland Island Project First Annual Report: 1996-1998*. Department of Conservation. Napier.

Magnuson, J. J. 1990. Long-term ecological research and the invisible present. *Bioscience*. 40: 495-502.

Mapstone, B. D. 1995. Scalable decision rules for environmental impacts studies: effect studies, Type I and Type II errors. *Ecological Applications*. 5: 401-410.

Marsh, D. S. 2001. Fluctuations in amphibian populations: a meta-analysis. *Biological Conservation.* 101: 327-335.

Michener, W.K. 1997. Quantitatively evaluating restoration experiments: research design, statistical analysis, and data management considerations. *Restoration Ecology*. 5: 324-337.

Millard, S. P. 1987. Environmental monitoring, statistics, and the law: room for improvement. *The American Statistician*. 41: 249-253.

Millard, S. P.; Yearsley, Lettenmaier, D. P. 1985. Space-time correlation and its effects on methods for detecting aquatic ecological change. *Canadian Journal of Fisheries & Aquatic Sciences*. 42: 1391-1400.

Miller, C.; Elliot, M.; Alterio, N. 2001. Home range of stoats (*Mustela erminea*) in podocarp forest, south Westland, New Zealand: implications for a control strategy. *Wildlife Research*. 28: 165-172.

Miller, C. J.; Miller, T. K. 1995. Population dynamics and diet of rodents on Rangitoto island, New Zealand. *New Zealand Journal of Ecology*. 19: 19-27.

Miles, J. R. G.; Potter, M. A.; Fordham. R. A. 1997. Northern brown kiwi (*Apteryx australis mantelli*) in Tongariro National Park and Tongariro forest – ecology and threats. *Science for Conservation*. 51. Department of Conservation. Wellington.

Mills, L. S.; Soule, M. E.; Doak, D. F. 1993. The keystone-species concept in ecology and conservation. *Bioscience*. 43: 219-225.

Miskelly, C. M. 1997. Whitaker's skink *Cyclodina whitakeri* eaten by a weasel *Mustela nivalis. Conservation Science Notes.* 146. Department of Conservation. Wellington.

Moller, H. 1985. Tree wētā (*Hemideina crassicruris*) (Orthoptera: Stenopelmatidae) of Stephens Island, Cook Strait. *New Zealand Journal of Zoology*. 12: 55-69.

Montalvo, A. M. 1997. Restoration biology: a population biology perspective. *Restoration Ecology.* 5: 277-290.

Moors, P. J. 1990. *Norway Rat*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Morgan-Richards, M. 1995. A new species of tree wētā from the North Island of New Zealand (*Hemideina*: Stenopelmatidae: Orthoptera). *New Zealand Entomologist*. 18: 15-23.

Morrison, R. I. G.; Downes, C.; Collins, B. 1994. Population trends on shorebirds on fall migration in eastern Canada. *Wilson Bulletin.* 106: 431-447.

Moseby, K. E.; Read, J. L. 2001. Factors affecting pitfall capture rates of small ground vertebrates in arid South Australia. II. optimum pitfall trapping effort. *Wildlife Research*. 28: 61-71.

Moss, K.; Sanders, M. 2001. Advances in New Zealand mammalogy: Hedgehog. *Journal of the Royal Society of New Zealand*. 31: 31-42.

Muller, K. E.; Benignus, V. A. 1992. Increasing scientific power with statistical power. *Neurotoxicology and Teratology*. 14: 211-219.

Murphy, E. C.; Dowding, J. E. 1994. Range and diet of stoats (*Mustela erminea*) in a New Zealand beech forest. *New Zealand Journal of Ecology*. 18: 11-18.

Murphy, E. C.; Dowding, J. E. 1995. Ecology of the stoat in *Nothofagus* forest: home range, habitat use and diet at different stages of the beech mast cycle. *New Zealand Journal of Ecology*. 19: 97-109.

Murphy, E. C.; Pickard, C. R. 1990. *House Mouse*. In. *The Handbook of New Zealand Mammals*. King, C. M. Ed. Oxford University Press. Auckland.

Newell, C. L.; Payton, I. J.; Allen, R. B. 2002. *A permanent plot method for monitoring changes in indigenous forests*. Department of Conservation. Wellington.

Norton, B. G.; Ulanowicz. R. E. 1992. Scale and Biodiversity Policy: A Hierarchical Approach. *Ambio*. 21: 244-249.

Norton, D. A. 1993. Mainland habitat islands: a vision for New Zealand nature conservation. *West Coast Conservancy Technical Report Series* No. 2, Department of Conservation, Hokitika.

Norton, D. A. 1997. *An assessment of possum (Trichorsurus vulpecula) impacts on loranthaceous mistletoes*. In. de Lange, P. J.; Norton. D. A. (Eds). 1997. *New Zealand's loranthaceous mistletoes*. Proceedings of a workshop hosted by Threatened Species Unit. Department of Conservation. Cass, 17-20 July 1995.

Norton, D. A.; Reid, N. 1997. Lessons in ecosystem management from management of threatened and pest loranthaceous mistletoes in New Zealand and Australia. *Conservation Biology*. 11: 759-769.

Norton, D. A. 1996. *Monitoring biodiversity in New Zealand's terrestrial ecosystems*. In *Biodiversity: papers from a seminar series on biodiversity*. Science and Research Division, Department of Conservation, Wellington.

Norton, D. A. 2000. *Benefits of possum control for native vegetation*. In. *The Brushtail Possum: biology, impact, and management of an introduced marsupial.* Montague, T. L. Ed. Manaaki Whenua Press. Lincoln. New Zealand.

Noss, R. F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology*. 4: 355-364.

Nugent, G.; Fraser, K. W.; Asher, G. W.; Tustin, K. G. 2001. Advances in New Zealand Mammalogy: Deer. *Journal of the Royal Society of New Zealand.* 31: 233-241.

Nugent, G.; Sweetapple, P.; Coleman, J.; Suisted, P. 2000. *Possum feeding patterns: Dietary tactics of a reluctant folivore*. In. *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial.* Manaaki Whenua Press. Lincoln.

O'Donnell, C. F. J.; Langton, S. 2003. Power to detect trends in abundance of long-tailed bats (*Chalinobus tuberculatus*) using counts on line transects. *Science for Conservation*. 224. Department of Conservation. Wellington.

Odum, E. P. 1983. *Basic Ecology*. Holt-Saunders. Philadelphia.

Ogle, C. C. 1997. *Evidence for the impacts of possums on mistletoes.* In. de Lange, P. J.; Norton. D. A. (Eds). 1997. *New Zealand's loranthaceous mistletoes*. Proceedings of a workshop hosted by Threatened Species Unit. Department of Conservation. Cass, 17-20 July 1995.

Ogle, M. 1999. *Sample size for foliar browse scoring*. In. *Outcome Monitoring – Workshop proceedings*. Frimmel, S. M.; Turner, S. J. (Compilers). Science & Research Internal Report. 170. Department of Conservation. Wellington.

Oliver, I.; Beattie, A. J. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. *Ecological Applications*. 6: 594-607.

Ordish, R. G. 1992. Aggregation and communication of the Wellington wētā *Hemideina crassidens* (Blanchard) (Orthoptera: Stenopelmatidae). *New Zealand Entomologist*. 15: 1-8.

Osenberg, G. W.; Schmitt, R. J.; Holbrook, S. J.; Abu-Saba, K. E.; Flegal, A. R. 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. *Ecological Applications*. 4: 16-30.

Ottenbacher. K, J. 1996. The power of replications and replications of power. *The American Statistician*. 50: 271-276.

Park, G. 2000. *New Zealand as ecosystems: the ecosystem concept as a tool for environmental management and conservation*. Department of Conservation, Wellington.

Parkes, J. 2001. Advances in New Zealand Mammalogy: Feral Livestock. *Journal of the Royal Society of New Zealand.* 31: 233-241.

Payton, I. 2000. *Damage to native forests.* In. *The Brushtail Possum: biology, impact, and management of an introduced marsupial.* Montague, T. L. Ed. Manaaki Whenua Press. Lincoln. New Zealand.

Pechmann, J. H. K.; Scott, D. E.; Semlitsch, R. D.; Caldwell, J. P.; Vitt, L. J.; Gibbons, J. W. 1991. Declining amphibian populations: the problem of separating human impacts from natural fluctuations. *Science*. 253: 892-895.

Pelton, M. R.; van Manen, F. T. 1996. Benefits and pitfalls of long-term research: a case study of black bears in Great Smokey Mountains National Park. *Wildlife Society Bulletin*. 24: 443-450.

Perry, J. N. 1986. Multiple-comparison procedures: A dissenting view. *Journal of Economic Entomology*. 79: 1149-1155.

Peterman, R. M. 1989. Application of statistical of statistical power analysis to the Oregon coho salmon (*Oncorhynchus kisutch*) problem. *Canadian Journal of Aquatic Science*. 46: 1183-1187.

Peterman, R. M. 1990a. Statistical power analysis can improve fisheries research and management. *Canadian Journal of Fisheries and Aquatic Science*. 47: 2-15.

Peterman, R. M. 1990b. The importance of reporting statistical power: the forest decline and acidic deposition example. *Ecology*. 71: 2024-2027.

Peterman, R. M.; McGonigle, M. 1992. Statistical power analysis and the precautionary principle. *Marine Pollution Bulletin*. 24: 231-234.

Pierce, R. J. 2002. Kiore (*Rattus exulans*) impact on breeding success of Pycroft's petrels and little shearwaters. *Science Internal Series.* 39. Department of Conservation. Wellington.

Pierce, R. J.; Graham, P. J. 1995. Ecology and breeding biology of Kukupa (*Hemiphaga novaeseelandiae*). *Science & Research Series*. 91. Department of Conservation. Wellington.

Pimm, S. L.; Redfearn, A. 1988. The variability of population densities. *Nature*. 334: 613-614.

Pledger, S. 1998. Monitoring protocols for Hamilton's frog *Leiopelma hamiltoni* on Stephens island. *Conservation Advisory Science Notes*. 205. Department of Conservation. Wellington.

Rangen, S. A.; Hobson, K. A.; Clark, R. G. 2000. A comparison of density and reproductive indices of songbirds in young and old boreal forest. *Wildlife Society Bulletin.* 28: 110-118.

Reed, J. M.; Blaustein, A. R. 1997. Assessment of "nondeclining" amphibians populations using power analysis. *Conservation Biology*. 9: 1299-1300.

Ribic, C. A.; Ganio, L. M. 1996. Power analysis for beach surveys of marine debris. *Marine Pollution Bulletin*. 32: 554-557.

Rickard, C. G. 1996. *Introduced small mammals and invertebrate conservation in a lowland podocarp forest South Westland, New Zealand*. Unpublished Masters of Forestry Science thesis. University of Canterbury, Christchurch, New Zealand.

Rotenberry, J. T.; Wiens, J. A. 1985. Statistical power analysis and community-wide patterns. *American Naturalist* 125: 164-168.

Rudge, M. R. 1990. *Feral Goat*. In. King, C. M. Ed. *The Handbook of New Zealand Mammals*. Oxford University Press. Auckland.

Rufaut, C. G. 1995. A *comparative study of the Wellington tree weta Hemideina crassidens (Blachard, 1851) in the presence and absence of rodents*. Unpublished Masters Thesis. Victoria University. Wellington.

Ruscoe, W. A. 2001. Advances in New Zealand mammalogy: House mouse. *Journal of The Royal Society of New Zealand.* 31: 127-134.

Ruscoe, W. A.; Goldsmith, R.; Choquenot, D. 2001. A comparison of population estimates and abundance indices for house mice inhabiting beech forests in New Zealand. *Wildlife Research*. 28: 173-178.

Ryan, T. J.; Philippi, T.; Leiden, Y. A.; Dorcas, M. E.; Wigley, T. B.; Gibbons, J. W. 2002. Monitoring herpetofauna in a managed forest landscape: effects of habitat types and census techniques. *Forest Ecology and Management*. 167: 83-90.

Sadlier, R. 2000. *Evidence of Possums as Predators of Native Animals*. In. *The Brushtail Possum: Biology, Impact and Management of an Introduced Marsupial.* Montague, T. L. Ed. Manaaki Whenua Press. Lincoln.

Saunders, A. 1994. *Introduction*. In. *Threatened terrestrial insects: a workshop to enhance conservation*. Cresswell, M.; Veitch, D. eds. *Threatened Species Occasional Publication*. 6. Department of Conservation. Wellington.

Saunders, A. 2000. *Mainland Islands - a review*. Department of Conservation. Wellington.

Saunders, A.; Norton, D. A. 2001. Ecological restoration at mainland islands in New Zealand. *Biological Conservation*. 99: 109-119.

Sedlmeier, P.; Gigerenzer, G. 1989. Do studies of statistical power have an effect on the power of studies? *Psychological Bulletin*. 105: 309-316.

Sessions, L. A.; Rance, C.; Grant, A.; Kelly, D. 2001. Possum (*Trichosurus vulpecula*) control benefits native beech mistletoes (Loranthaceae). *New Zealand Journal of Ecology.* 25: 27-33.

Sheppard, C. R. C. 1999. How large should my sample be? Some quick guides to sample size and the power of tests. *Marine Pollution Bulletin*. 38: 439-447.

Sherley, G. 1996. Biodiversity research for conservation in New Zealand: Lessons from Australia : An Australia New Zealand Foundation Fellowship Report 1996. *Science & Research Internal Report*. 154. Department of Conservation. Wellington.

Skalski, J. R. 1990. A design for long-term status and trends monitoring. *Journal of Environmental Management*. 30: 139-144.

Smale, M. C.; Hall, G. M. J.; Gardner, R. O. 1995. Dynamics of kanuka (*Kunzea ericoides*) forest on south kaipara spit, New Zealand, and the impact of fallow deer (*Dama dama*). *New Zealand Journal of Ecology*. 19: 131-141.

Spellerberg, I. F. 1991. *Monitoring Ecological Change*. Cambridge University Press.

Spurr, E. B.; Drew, K. W. 1999. Invertebrates feeding on baits used for vertebrate pest control in New Zealand. *New Zealand Journal of Ecology*. 23: 167-173.

Stanturf, J. A.; Schoenholtz, S. H.; Schweitzer, S. H.; Shephard, J, P. 2001. Achieving restoration success: myths in bottomland hardwood forests. *Restoration Ecology*. 9: 189-200.

Steidl, R. J.; Hayes, J. P.; Schauber, E. 1997. Statistical power analysis in wildlife research. *Journal of Wildlife Management*. 6: 270-279.

Stewart, G.H.; Burrows, L.E. 1989. The impact of white-tailed deer *Odocoileus virginianus* on regeneration in the coastal forests of Stewart Island, New Zealand. *Biological Conservation* 49: 275-293.

Stephens, T.; Barnett, S. 1998. A review of possum monitoring in Waikato conservancy. *Conservation Advisory Notes.* 171. Department of Conservation. Wellington.

Stephenson, B. M. 1998. *The ecology and breeding biology of morepork, Ninox novaeseelandiae, and their risk from secondary poisoning, in New Zealand*. Unpublished Masters of Science Thesis. Massey University, Palmerston North.

Strayer, D. L. 1999. Statistical power of presence-absence data to detect population declines. *Conservation Biology.* 13: 1034-1038.

Studholme, B. 2000. Ship rat (*Rattus rattus*) irruptions in South Island beech (Nothofagus) forests. *Conservation Advisory Science Notes*. 318. Department of Conservation. Wellington.

Sweetapple, P. J.; Burns, B. R. 2002. Assessing the response of forest understoreys to feral goat control with and without possum control. *Science for Conservation.* 201. Department of Conservation. Wellington.

Taylor, B. L.; Gerrodette, T. 1993. The uses of statistical power in conservation biology: the vaquita and northern spotted owl. *Conservation Biology*. 7: 489-500.

Taylor, R. H.; Thomas, B. W. 1993. Rats eradicated from rugged Breaksea Island (170ha), Fiordland, New Zealand. *Biological Conservation*. 65: 191-198.

Thomas, G. 2001. Where property rights and biodiversity converge part III: incorporating adaptive management and the precautionary principle into HCP design. *Endangered Species Update*. 18: 32-40.

Thomas, L. 1996. Monitoring long-term population change: why are there so many analysis methods? *Ecology*. 77: 49-59.

Thomas, L. 1997. Retrospective power analysis. *Conservation Biology*. 11: 276-280.

Thomas, L.; Krebs, C. J. 1997. A review of statistical power analysis software. *Bulletin of the Ecological Society*. 78: 126-136.

Thomas, L.; Juanes, F. 1996. The importance of statistical power analysis: an example from animal behaviour. *Animal Behaviour*. 52: 856-859.

Thomas, M. D. 1999. Feasibility of using wax-blocks to measure rodent and possum abundance and changes in population size. In Anon. 1999. *Science for Conservation.* 127. Department of Conservation. Wellington.

Thomas, M. D.; Brown, J. A. 2000. Possum monitoring using raised leg-hold traps. *Science for Conservation.* 164. Department of Conservation. Wellington.

Thomas, M. D.; Brown, J. A.; Maddigan, F. W.; Sessions, L. A. 2003 (*In Press*). *Comparison of trap-catch interference methods and estimating possum densities*. New Zealand Plant Protection Society.

Thomas, M. D.; Frampton, C. M.; Briden, K. W.; Hunt, K. G. 1995. Evaluation of Brodifacoum Baits for Maintenance Control of Possums in Small Forest Reserves. *Forests and Environment*. X :256-259.

Thompson, P. M.; Wilson, B.; Grellier, K.; Hammond, P. S. 2000. Combining power analysis and population viability analysis to compare traditional and precautionary approaches to conservation of coastal cetaceans. *Conservation Biology*. 14: 1253-1263.

Thompson, W. L. 2002. Towards reliable bird surveys: accounting for individuals present but not detected. *The Auk*. 119: 18-25.

Tocher, M. 1999. Big Bay skink (*Oligosoma* sp.): taxonomy, distribution and habitat requirements. *Conservation Advisory Science Notes*. 228. Department of Conservation. Wellington.

Toft, C. A.; Shea, P. J. 1983. Detecting community-wide patterns: estimating power strengthens statistical inference. *American Naturalist*. 122: 618-25.

Towns, D. R. 1994. The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *Journal of Zoology*. 21: 457-472.

Towns, D. R. 1996. Changes in habitat use by lizards on a New Zealand island following removal of the introduced Pacific rat *Rattus exulans*. *Pacific Conservation Biology*. 2: 286-292.

Towns, D. R.; Williams, M. 1991. *Single species conservation in New Zealand: towards a redefined conceptual approach.* Science and Research Division. Department of Conservation. Wellington.

Towns, D. R.; Neilson, K. A.; Whitaker, A. H. 2002. North Island *Oligosoma* spp. skink recovery plan: 2002-2012. *Threatened Species Recovery Plan.* 48. Department of Conservation. Wellington.

Townsend, A. J. 1996. *Maungaharuru Ecological District: Survey report for the Protected Natural Areas Programme*. Department of Conservation. Wellington.

Townsend, J. A.; Brown, B.; Stringer, I. A. N.; Potter, M. A. 1997. Distribution, habitat, and conservation status of *Hemideina ricta* and *H. femorata* on Banks peninsula, New Zealand. *New Zealand Journal of Ecology*. 21: 43-49.

Townsend, J. I. 1996. An insect survey of Paengaroa Scenic Reserve, Mataroa. *Conservation Advisory Science Notes*. 152. Department of Conservation. Wellington.

Trenkel, V. M. 2001. Exploring the red deer culling strategies using a population-specific calibrated management model. *Journal of Environmental Management.* 62: 37-54.

Treshow, M.; Allan, J. 1985. Uncertainties associated with the assessment of vegetation. *Environmental Management*. 9: 471-478.

Trewick, S. A.; Morgan-Richards, M. 1995. On the distribution of tree wētā in the North Island, New Zealand. *Journal of the Royal Society of New Zealand*. 25: 485-493.

Trewick, S. A.; Morgan-Richards, M. 2000. Artificial wētā roosts: A technique for ecological study and population monitoring of Tree Wētā (*Hemideina*) and other invertebrates. *New Zealand Journal of Ecology*. 24: 201-208.

Tukey, J. W. 1962. The future of data analysis. *Annals of Mathematical Statistics.* 33: 1-67.

Turner, M.G.; Gardner, R.H.; O'Neill, R.V. 1995. Ecological dynamics at broad scales: ecosystems and landscapes. *BioScience (Special supplement: biodiversity policy)* 45: 29-35.

Turner, S. R.; O'Neill, V.; Conley, V.; Conley, M. R.; Humphries, H. C. 1990. *Pattern and Scale: Statistics for Landscape Ecology*. In. Turner, M.G., Gardner, R.H. Eds. 1990. *Quantitative Methods in Landscape Ecology; the Analysis and Interpretation of Landscape Heterogeneity*. Springer-Verlag. New York.

Underwood, A. J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research*. 42: 569-587.

Underwood, A. J. 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable world. *Journal of Experimental Marine Biology & Ecology*. 161: 145-178.

Underwood, A. J. 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. *Australian Journal of Ecology*. 18: 99-116.

Underwood, A. J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications*. 4: 3-15.

van Diggelen, R.; Grootjans, A. P.; Harris, J. A. 2001. Ecological restoration: state of the art or state of the science? *Restoration Ecology*. 9: 115-118.

Veltman, C. 2000. *Do native wildlife benefit from possum control?* In. *The Brushtail Possum: biology, impact, and management of an introduced marsupial.* Montague, T. L. Ed. Manaaki Whenua Press. Lincoln. New Zealand.

Walls, G. 1989. The crest of the Maungaharuru Range, Hawke's Bay: conservation values. *Botany Division Report. 667*. Department of Scientific and Industrial Research. Havelock North.

Walls, G. 1995. (Revised; 1996, 1997). *Additions to Indigenous Plant Species List for Boundary Stream Scenic Reserve.* Unpublished Report. Hawke's Bay Conservancy. Napier.

Walls, G. 1996. *Monitoring Review: Hawke's Bay Conservancy*. Unpublished Report. Hawke's Bay Conservancy. Napier.

Walls, G. 1997. Forest gems in Hawke's Bay's mainland island – an appreciation of Pat Grant's contribution. *The New Zealand Botanical Society Newsletter*. 50. 10-11.

Warburton, W. 2000. *Monitoring Possum Populations*. In. *The Brushtail Possum: biology, impact, and management of an introduced marsupial.* Montague, T. L. Ed. Manaaki Whenua Press. Lincoln. New Zealand.

Wardle, D. A.; Barker, G. M.; Yeates, G. W.; Bonner, K. I.; Ghani, A. 2001. Introduced browsing mammals in New Zealand natural forests: aboveground and belowground consequences. *Ecological Monographs*. 71: 587-614.

Wassenaar, T. D.; Ferreira, S. M. 2002. Measuring conservation outcomes for depleted biological assets: Convergence theory and its application to New Zealand. *Department of Conservation Internal Science Series.* 38. Department of Conservation. Wellington.

Watts, C. H.; Gibbs, G. W. 2000. Species richness of indigenous beetles in restored plant communities on Matiu-Somes Island, Wellington Harbour, New Zealand. *New Zealand Journal of Ecology*. 24: 195-200.

Whitaker, A. H. 1993. *The striped skink (Leiolopisma striatum (Buller 1871)): a review with recommendations for conservation and management*. Unpublished report. Wanganui Conservancy. Department of Conservation. Wellington.

Wiens, J. A. 1989. Spatial Scaling in Ecology. *Functional Ecology*. 3: 385-397.

Wiens, J. A.; Parker, K. R. 1995. Analyzing the effects of accidental environment impacts: approaches and assumptions. *Ecological Applications*. 5: 1069-1083.

Williams, K. S. 1993. Use of terrestrial arthropods to evaluate restored riparian woodlands. *Restoration Ecology.* 1: 107-116.

Wilson, B.; Hammond, P. S.; Thompson, P. M. 1999. Estimating size and assessing trends in a coastal bottlenose dolphin population. *Ecological Applications*. 9: 288-300.

Wilson, G. J.; Delahay, R. J. 2001. A review of methods to estimate the abundance of terrestrial carnivores using field signs and observation. *Wildlife Research*. 28: 151-164.

Wilson, P. R.; Toft, R. J.; Beggs, J. R.; Taylor, R. H. 1993. The role of predators and competitors in the decline of kaka (*Nestor meridionalis*) populations in New Zealand. *Biological Conservation.* 83: 175-185.

Wolfe, D. A.; Champ, M. A.; Flemer, D. A.; Mearns, A. J. 1987. Long-term biological data sets: their role in research, monitoring, and management of estuarine and coastal marine systems. *Estuaries*. 10: 181-193.

Yoccoz, N. G. 1991. Use, overuse, and misuse of significance tests in evolutionary, biology and ecology. *Bulletin of the Ecological Society of America*. 72: 106-111.

Zar, J. H. 1999. *Biostatistical Analysis*. 4th Ed. Prentice Hall International, Inc. London.

Zielinski, W. J.; Stauffer, H. B. 1996. Monitoring Martes populations in California: survey design and power analysis. *Ecological Applications*. 6: 1254-1267.

Appendices

Appendix 1. Statistical Power Analysis used in Biological Monitoring Programmes.

Edwards & Perkins (1992) used *a priori* statistical power analysis (on actual data by numerical methods) to detect linear trends in abundance of dolphin stocks affected by the tuna purseseine fishery in the eastern tropical Pacific Ocean.

Kendall *et al.* (1992) used *a priori* statistical power analysis (on actual data by simulation using a beta-binomial model) to provide guidelines for effective monitoring of grizzly bear (*Ursus arctos*) populations using sign surveys in North America.

Taylor & Gerrodette (1993) used *a priori* statistical power analysis (by algebraic calculation on simulated data, and simulated distributions based on a population model) to detect trend declines in abundance of the vaquita (*Phocoena sinus*) – an endangered porpoise in the northern Gulf of California Mexico, and the northern spotted owl (*Strix occidentalis caurina*) in western North America.

Benn *et al.* (1995) used *a priori* statistical power analysis (on actual data with TRENDS computer programme, see Appendix 10) to determine whether the survey methods used in the Kruger National Park (South Africa) were sufficient to detect trends in population size of the saddlebilled stork (*Ephippiorhynchus senegalensis*).

Ribic & Ganio (1996) used *a priori* statistical power analysis (on actual data by numerical methods) to determine sample sizes required for the monitoring of marine pollution (US National beach survey for debris) along the North American coastline.

Beier & Cunningham (1996) used *a priori* statistical power analysis (on actual data by simulation using a poisson model) to detect changes in cougar (*Puma concolor*) populations by the use of track surveys, in south-eastern Arizona, United States of America.

Hatfield *et al.* (1996) used statistical power analysis (on actual data by simulation using a population growth model) in the detection of trends in raptor – Bald eagle (*Haliaeetus leucocephalus*) counts, Michigan, Minnesota, and Wisconsin, United States of America.

Morrison *et al. (*1996) used *a priori* statistical power analysis (on actual data by numerical methods) to evaluate the data from the Canadian Maritimes Shorebird Survey, for 13 shorebird species.

Zielinski & Stauffer (1996) used *a priori* statistical power analysis (on actual data by simulation using a binomial model) to determine sample size for Fishers (*Martes pennanti*) and American martens (*M. americana*) track counts, in California, United States of America.

Eggeman *et al.* (1997) used *a priori* statistical power analysis (on actual data using custom-made simulation software) to evaluate an aerial quadrat survey method for monitoring populations of 14 wintering duck species, in Florida, United States of America.

Gibbs & Melvin (1997) used *a priori* statistical power analysis (on actual data with simulation software MONITOR, see Appendix 10) in the determination of the most effective monitoring programme designs to detect trends in waterbird abundance – Pied-billed grebe (*Podilymbus podiceps*), American bittern (*Botaurus lentiginosus*), Virginia rail (*Rallus limicola*), and sora (*Porzana carolina*) with call-response surveys in Maine, United States of America.

Gryska *et al.* (1997) used *a priori* statistical power analysis (on actual data by numerical methods) to develop a monitoring protocol for the endangered Kendall warm springs dace (*Rhinichthys osculus thermalis*), in Wyoming, United States of America.

Brown & Miller (1998) used *a priori* statistical power analysis (on actual data by simulation using a binomial model) to determine an effective tracking tunnel monitoring programme for stoat (*Mustela erminea*) control operations, in the West Coast region, New Zealand.

Lougheed *et al.* (1999) used retrospective statistical power analysis (on actual data by numerical methods and algorithms available as freeware on the World Wide Web, examples given in Appendix 10) to evaluate population trends and survey designs in waterfowl surveys conducted by the Canadian Wildlife Service, Riske Creek, British Columbia, Canada.

Maxwell (1999) used *a priori* statistical power analysis (on actual data with TRENDS computer programme – see Appendix 10) to determine an effective level of monitoring for the bull trout (*Salvelinus confluentus*) – a threatened species, in the Columbia and Klamath river drainages, Montana, United States of America.

Wilson *et al.* (1999) used *a priori* statistical power analysis (on actual data by numerical methods) to investigate the length of time for a monitoring programme to detect changes in the abundance of a bottlenose dolphin (*Tursiops truncatus*) population, Moray firth, Scotland.

Bishop *et al.* (2000) used *a priori* statistical power analysis (on actual data with MONITOR computer programme, see Appendix 10) to estimate the number of years of survey effort required to detect a change in migrant shorebird – Western sandpipers (*Calidris mauri*), and Dunlin (*Calidris alpina pacifica*) numbers on the copper river delta, Alaska, United States of America.

Lindley *et al.* (2000) used *a priori* statistical power analysis (on actual data by numerical methods) to determine whether a recovery benchmark had been met for the endangered Sacramento river winter chinook salmon (*Oncorhynchus tshawytscha*), Northern California, United States of America.

O'Donnell & Langton (2003) used *a priori* statistical power analysis (on pilot study data with a route-regression technique similar to that of the MONITOR computer programme) in the design of long-term monitoring programmes for the threatened New Zealand long-tailed bat (*Chalinolobus tuberculatus*).

Appendix 2. Treatment Monitoring Programme Designs.

Monitoring Programme	Design
Wētā	5 Lines x 5 Groups (5 x 5: MD) x 4 Roosts (SD) x 2 (MO): Comparison between habitats. 25 Groups x 1 Treatment (25 x 1: MD) x 4 Roosts (SD) x 2 MO): Pooled Treatment.
Ground Invertebrates	5 Lines x 1 Treatment (5 x 1: MD) x 20 Traps (SD) x 1 (MO): Pooled Treatment, by habitat. 5 Lines x 5 Groups (5 x 5: MD) x 4 Traps (SD) x 1 (MO): Comparison between habitats. 25 Groups x 1 Treatment (25 x 1: MD) x 4 Traps (SD) x 2 (MO): Pooled Treatment.
Lizards	3 Lines x 12 Checks (3 x 12: MD) x 50 Traps (SD) x 1 (MO): Pooled Treatment, by habitat. Pooled Lines x 1 (1 x 1: MD) x 150 Traps (SD) x 1 (MO): Pooled Treatment. Pooled Lines x 2 (1 x 1: MD) x 150 Traps (SD) x 2 (MO): Pooled Treatment. 10 Lines x 1 (10 x 1: MD) x 10 Traps (SD) x 1 (MO): Pooled Treatment (High-Catch Sites), one monitoring period. 10 Lines x 2 (10 x 1: MD) x 10 Traps (SD) x 2 (MO): Pooled Treatment (High-Catch Sites), two monitoring periods.
Birds	3 Lines x 1 Treatment (3 x 1: MD) x 10 count sites (SD) x 8 (MO): Seasonally x 2, Pooled Treatment, by Lines/habitat. 3 Lines x 1 Treatment (3 x 1: MD) x 10 count sites (SD) x 4 (MO): Biannually x 2, Pooled Treatment, by Lines/habitat.
Vegetation	16 x 1 Treatment (16 x 1: MD) x 20 subplots (SD) x 1 (MO: once every five years): Pooled Treatment, by habitats.
Mustelids	8 Lines x 1 Treatment (8 x 1: MD) x 5 tracking-tunnels (SD) x 1 (MO: once per run): Pooled Treatment, by habitats, seasonally. 8 Lines x 1 Treatment (8 x 3: MD) x 5 tracking-tunnels (SD) x 3 (MO: Checked three times per run): Pooled Treatment, by habitats, seasonally. 8 Lines x 1 Treatment (8 x 4: MD) x 5 tracking-tunnels (SD) x 4 (MO: Annually): Pooled Treatment, by habitats.

Monitoring Programme	Design
Rodents	1 Line x 1 Treatment (1 x 1: MD) x 25 tracking-tunnels (SD) x 1 (MO: seasonally): Comparison between habitats. 1 Line x 1 Treatment (1 x 4: MD) x 25 tracking-tunnels (SD) x 1 (MO: annually): Comparison between habitats. 4 Lines x 1 Treatment (4 x 1: MD) x 25 tracking-tunnels (SD) x 1 (MO: seasonally): Pooled Treatment, by habitats. 4 Lines x 1 Treatment (4 x 4: MD) x 25 tracking-tunnels (SD) x 1 (MO: annually): Pooled Treatment, by habitats.
Possums	4 Lines x 1 Treatment (4 x 1: MD) x 30 Traps (SD) x 1 (MO: annually): Initial design. Pooled Treatment, by habitats. 10 Lines x 1 Treatment (10 x 1: MD) x 20 Traps (SD) x 1 (MO: annually): Current design. Pooled Treatment, by habitats.

MD: Monitoring Design, SD: Standard Deviation components, MO: Monitoring Occasion, generally per year.

Appendix 3. Non-Treatment Monitoring Programme Designs.

Monitoring Programme	Design
Wētā	4 Lines x 5 Groups (4 x 5: MD) x 4 Roosts (SD) x 2 (MO): Comparison between habitats. 20 Groups x 1 (Non-) Treatment (20 x 1: MD) x 4 Roosts (SD) x 2 MO): Pooled (Non-)Treatment.
Ground Invertebrates	4 Lines x 1 (Non-) Treatment (4 x 1: MD) x 20 Traps (SD) x 1 (MO): Pooled (Non-) Treatment, by habitat. 4 Lines x 5 Groups (4 x 5: MD) x 4 Traps (SD) x 1 (MO): Comparison between habitats. 20 Groups x 1 (Non-) Treatment (20 x 1: MD) x 4 Traps (SD) x 2 (MO): Pooled (Non-) Treatment.
Lizards[1]	
Birds	2 Lines x 1 (Non-) Treatment (2 x 1: MD) x 15 count sites (SD) x 8 (MO): Seasonally x 2, Pooled (Non-) Treatment, by Lines/habitat. 2 Lines x 1 (Non-) Treatment (2 x 1: MD) x 15 count sites (SD) x 4 (MO): Biannually x 2, Pooled (Non-) Treatment, by Lines/habitat.
Vegetation	7 x 1 (Non-) Treatment (7 x 1: MD) x 20 subplots (SD) x 1 (MO: once every five years): Pooled Exclosures, by habitats. 4 x 1 (Non-) Treatment (4 x 1: MD) x 20 subplots (SD) x 1 (MO: once every five years): Pooled (Non-) Treatment, by habitats.

Monitoring Programme	Design
Mustelids	6 Lines x 1 (Non-) Treatment (6 x 1: MD) x 5 tracking-tunnels (SD) x 1 (MO: once per run): Pooled (Non-) Treatment, by habitats, seasonally.
	6 Lines x 1 (Non-) Treatment (6 x 3: MD) x 5 tracking-tunnels (SD) x 3 (MO: Checked three times per run): Pooled (Non-) Treatment, by habitats, seasonally.
	6 Lines x 1 (Non-) Treatment (6 x 4: MD) x 5 tracking-tunnels (SD) x 4 (MO: Annually): Pooled (Non-) Treatment, by habitats.
	3 Lines x 1 (Non-) Treatment (3 x 1: MD) x 5 tracking-tunnels (SD) x 1 (MO: once per run): Design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats, seasonally.
	3 Lines x 1 (Non-) Treatment (3 x 3: MD) x 5 tracking-tunnels (SD) x 3 (MO: Checked three times per run): Design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats, seasonally.
	3 Lines x 1 (Non-) Treatment (3 x 4: MD) x 5 tracking-tunnels (SD) x 4 (MO: Annually): Design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats.
Rodents	1 Line x 1 (Non-) Treatment (1 x 1: MD) x 25 tracking-tunnels (SD) x 1 (MO: seasonally): Comparison between habitats.
	1 Line x 1 (Non-) Treatment (1 x 4: MD) x 25 tracking-tunnels (SD) x 1 (MO: annually): Comparison between habitats.
	2 Lines x 1 (Non-) Treatment (2 x 1: MD) x 25 tracking-tunnels (SD) x 1 (MO: seasonally): Design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats.
	2 Lines x 1 (Non-) Treatment (2 x 4: MD) x 25 tracking-tunnels (SD) x 1 (MO: annually): Design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats.
	4 Lines x 1 (Non-) Treatment (4 x 1: MD) x 25 tracking-tunnels (SD) x 1 (MO: seasonally): Pooled (Non-) Treatment, by habitats.
	4 Lines x 1 (Non-) Treatment (4 x 4: MD) x 25 tracking-tunnels (SD) x 1 (MO: annually): Pooled (Non-) Treatment, by habitats.

Monitoring Programme	Design
Possums	4 Lines x 1 (Non-) Treatment (4 x 1: MD) x 30 Traps (SD) x 1 (MO: annually): Initial design. Pooled (Non-) Treatment, by habitats. 2 Lines x 1 Treatment (2 x 1: MD) x 30 Traps (SD) x 1 (MO: annually): Initial design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats. 6 Lines x 1 Treatment (6 x 1: MD) x 20 Traps (SD) x 1 (MO: annually): Current design. Pooled (Non-) Treatment, by habitats. 3 Lines x 1 Treatment (3 x 1: MD) x 20 Traps (SD) x 1 (MO: annually): Current design for a single (Non-Treatment) reserve. Pooled (Non-) Treatment, by habitats.

MD: Monitoring Design, SD: Standard Deviation components, MO: Monitoring Occasion, generally per year.
[1] As of this analysis, no Non-Treatment Lizard Pitfall trap monitoring has been initiated.

Appendix 4. Power Table Timeframes: 10 % Positive Change.

Expected timeframes (Years of monitoring from 1996) to reach 0.8 Statistical Power (alpha: 0.05, 0.1, 0.2, 0.25*), to determine 10% positive change;

Programme	Group	Design	Boundary Stream	Non-Treatment Sites (combined)	Single Non-Treatment Reserve †
Wētā [1] (2x a year)	Total No.s	Lines	4.5, 3.5, 3, 3	-, 6.5, 4.5, 4	-, -, -, -
		Groups	3, 3, 2.5, 2.5	5, 4.5, 4, 3.5	6.5, 5, 4.5, 4
Invertebrates (1x a year)	Total No.s	Lines	5, 4, 4	6, 5, 4	
		Groups	6, 5, 5	6, 5, 4	
	Large Inverts	Lines	9, 9, 5	12, 10, 5	
		Groups	-, 12, 5	-, -, 6	
	Beetles	Lines	10, 10, 5	13, 10, 7	
		Groups	-, -, 6	-, -, 8	
Lizards (1x a year)	Total No.s	Pooled Lines	18, 17, 15, 14		
		3 Lines	-, -, -, -		
		Ten Lines	14, 13, 12, 10		
(2x a year)		Pooled Lines	7.5, 6.5, 6, 5.5		
		Ten Lines	6.5, 5, 4.5, 4.5		
Birds (4x a year)	Overall No.s		1 ½ , 1 ¼ , 1	-, 2 ¾ , 1 ¼	
	Species No.s		1 ¼ , 1, 1	3 ½, 1 ¾, 1 ¼	
	Indigenous No.s		1 ½ , 1 ¼ , 1	-, 3, 1 ½	
	Bellbird No.s		1, 1 ¾, 1 ¼	-, 4 ¼, 2	
(2x a year)	Overall No.s		3, 2 ½, 2	-, 5 ½, 2 ½	
	Species No.s		2 ½, 2, 2	7, 3 ½, 2 ½	
	Indigenous No.s		3, 2 ½, 2	-, 6, 3	
	Bellbird No.s		2, 3 ½, 2 ½	-, 9, 4	
Vegetation (every 5 years)	Species No.s		15, 10, 10	10, 10, 10	10, 10, 10
	Sapling No.s		10, 10, 10	45, 45, 40	40, 40, 35

Programme	Group	Design	Boundary Stream	Non-Treatment Sites (combined)	Single Non-Treatment Reserve †
Mustelids	Total Activity	All Lines	-, -, 4, 3¾	-, 8½, 3½, 3¼	
Rodents	Total Activity				
	Seasonal	One Line	1¾, 1½, 1½	2¾, 2½, 2¼	2½, 2¼, 2¼
	Seasonal	All Lines	1¾, 1½, 1¼	2½, 2¼, 2	1¾, 1½, 1½
	All Year	One Line	7, 6, 5	8, 7, 6	10, 9, 9
	All Year	All Lines	6, 5, 4	7, 6, 5	7, 6, 6
Possums ‡ (1x a year)	Catch Rate				
	(Initial Design)		11, 9, 8, 7	7, 6, 5	-, -, 14
	(Current Design)		5, 5, 4	7, 6, 5	12, 8, 6

[1] Wētā monitoring started during summer 1997/98.
* For Wētā, Lizards, and Mustelids only
† Vegetation Exclosure sites
‡ Alpha levels (0.01, 0.05, 0.1, 0.2) for initial possum monitoring design at Boundary Stream
- Does not reach statistical power of 0.8, hence no timeframe

Appendix 5. Power Table Timeframes: 10 % Negative Change.

Expected timeframes (Years of monitoring from 1996) to reach 0.8 Statistical Power (alpha: 0.05, 0.1, 0.2, 0.25*), to determine 10% negative change;

Programme	Group	Design	Boundary Stream	Non-Treatment Sites (combined)	Single Non-Treatment Reserve †
Wētā [1] (2x a year)	Total No.s	Lines	11.5, 7, 5, 4.5	-, 30, 9.5, 8	-, -, -, -
		Groups	4.5, 4, 3.5, 3	12, 9, 7, 6	-, -, 15, 12
Invertebrates (1x a year)	Total No.s	Lines	6, 5, 4	8, 5, 4	
		Groups	9, 7, 6	8, 5, 5	
	Large Inverts	Lines	22, 18, 8	40+, 30+, 7	
		Groups	-, -, 8	-, -, 11	
	Beetles	Lines	30+, 25+, 6	40+, 30+, 10	
		Groups	-, -, 6	-, -, 14	
Lizards (1x a year)	Total No.s	Pooled Lines	-, -, -, -		
		3 Lines	-, -, -, -		
		Ten Lines	60+, 40+, 30+, 20+		
(2x a year)		Pooled Lines	-, -, 40+, 20+		
		Ten Lines	15, 13, 10, 9		
Birds (4x a year)	Overall No.s		2, 1 ½, 1 ¼	-, 4 ½, 1 ¾	
	Species No.s		1 ¾, 1 ¼, 1	11+, 3, 1 ½	
	Indigenous No.s		2, 1 ½, 1 ¼	-, 20+, 2 ¼	
	Bellbird No.s		10+, 2 ½, 1 ½	-, -, 3 ¼	
(2x a year)	Overall No.s		4, 3, 2 ½	-, 9, 3 ½	
	Species No.s		3 ½, 2 ½, 2	22+, 6, 3	
	Indigenous No.s		4, 3, 2 ½	-, 40+, 4 ½	
	Bellbird No.s		20+, 5, 3	-, -, 6 ½	

Programme	Group	Design	Boundary Stream	Non-Treatment Sites (combined)	Single Non-Treatment Reserve †
Vegetation (every 5 years)	Species No.s		15, 15, 10	15, 10, 10	15, 10, 10
	Sapling No.s		10, 10, 10	100+, 100+, 100+	100+, 85, 75
Mustelids	Total Activity	All Lines	-, -, -, 13	-, -, -, 25+	
Rodents	Total Activity				
	Seasonal	One Line	2¼, 2, 1¾		
	Seasonal	All lines	1¼, 1¼, 1		
	All Year	One Line	5, 5, 4		
	All Year	All lines	10, 6, 4		
Possums ‡ (1x a year)	Catch Rate				
	(Initial Design)		40+, 16, 13, 10	14, 9, 7	
	(Current Design)		6, 6, 5	10, 8, 7	

[1] Wētā monitoring started during summer 1997/98.
* For Wētā, Lizards, and Mustelids only
† Vegetation Exclosure sites
‡ Alpha levels (0.01, 0.05, 0.1, 0.2) for initial possum monitoring design at Boundary Stream
@ Approximation
\- Does not reach statistical power of 0.8, hence no timeframe

Appendix 6. Summary of Wētā Variability: Initial Values.

Table of the Coefficient of Variation (CV) for Tree wētā monitoring programme.
From initial 1997/98 (pilot) data

Monitoring Programme	Monitoring Subject	Mean CV (Coefficient of Variation)			
		Treatment	Non-Treatment	Non-Treatment Single Reserve †	
Wētā ‡					
By Lines	Total occupants	0.546641	0.817756	1.574106	t
	All wētā	0.450264	0.810340	1.687791	c
	Tree Wētā	0.693952	1.305582	1.595712	c
	Females	0.592655	1.071429	1.443760	c
	Males	0.912871	1.767767	2.5	t
	Auckland Tree Wētā	0.959166	1.305582	1.595712	c
	Females	0.947859	1.071429	1.443760	c
	Males	1.031510	1.767767	2.5	t
	Hawkes Bay Tree Wētā £	2.236068	-	-	
	Females	2.236068	-	-	
	Males	2.236068	-	-	
By Groups	Total occupants	0.202304	0.261201	0.436651	t
	All wētā	0.154840	0.293899	0.510657	t
	Tree Wētā	0.190697	0.453947	0.822851	t
	Females	0.265043	0.532357	0.971825	t
	Males	0.234060	0.512989	0.805076	t
	Auckland Tree Wētā	0.230940	0.453947	0.822851	t
	Females	0.347985	0.532257	0.971825	t
	Males	0.250720	0.512989	0.805076	t
	Hawkes Bay Tree Wētā	0.590727	-	-	
	Females	0.552771	-	-	
	Males	1	-	-	

Lines: Determination by Habitats.
Groups: Total Treatment.
‡: 1997/1998 Data
£ Hawkes Bay Tree Wētā have only been found (alive) within the habitats of Lines 1 & 8 (Christensen 2002a), thus a CV for Lines will always be a constant in this case: 2.236068.
†: c = Cashes Bush Scenic Reserve, t = Thomas Bush Scenic Reserve

Appendix 7. Summary of Wētā Variability per Year: Treatment Site.

Species count

Monitoring Programme, Subject	CV (Coefficient of Variation) †.						
	1995/96	1996/97	1997/98	1998/99	1999/00	2000/01	2001/02
Wētā (By Lines)							
Total occupants			0.54664	0.27617	0.29665	0.16363	0.11812
All wētā			0.45026	0.29533	0.26087	0.12656	0.20760
Tree Wētā (TW)			0.69395	0.26438	0.30090	0.35088	0.48426
Females			0.59266	0.50619	0.43241	0.33756	0.46515
Males			0.91287	0.34468	0.33503	0.38088	0.62769
Auckland TW			0.95917	0.23385	0.30090	0.43039	0.56621
Females			0.94786	0.50619	0.43241	0.40937	0.51645
Males			1.03151	0.39123	0.33503	0.46904	0.51645
Hawkes Bay TW ‡			2.23607	2.23607	-	2.23607	2.23607
Females			2.23607	-	-	2.23607	2.23607
Males			2.23607	2.23607	-	2.23607	2.23607
Wētā (By Groups)							
Total occupants			0.20230	0.11401	0.11431	0.08436	0.04830
All wētā			0.15484	0.13704	0.11774	0.08319	0.07724
Tree Wētā (TW)			0.19070	0.14029	0.14786	0.12705	0.13825
Females			0.26504	0.20497	0.21527	0.13920	0.13636
Males			0.23406	0.23028	0.20203	0.15720	0.18417
Auckland (TW)			0.23094	0.13502	0.14786	0.13567	0.15559
Females			0.34799	0.20497	0.21527	0.14813	0.14803
Males			0.25072	0.23490	0.20203	0.16583	0.19764
Hawkes Bay (TW)			0.59073	1	-	0.60285	0.49768
‡			0.55277	-	-	0.73283	0.69222
Females			1	1	-	0.73283	0.69222
Males							

† Six s.f.

‡ Hawkes Bay Tree Wētā have only been found (alive) within the habitats of Lines 1 & 8 (Christensen 2002a), thus a CV for Lines (1) will always be a constant in this case: 2.236068.

Appendix 8. Summary of Wētā Variability per Year: (Combined) Non-Treatment Sites.

Monitoring Programme, Subject	CV (Coefficient of Variation)†.						
	1995/96	1996/97	1997/98	1998/99	1999/00	2000/01	2001/02
Wētā (By Lines)							
Total occupants			0.81776	1.04450	0.94133	0.46116	0.09887
All wētā			0.81034	0.82063	0.94133	0.44117	0.18823
Tree Wētā (TW)			1.30558	1.44338	1.34568	0.85829	0.95743
Females			1.07143	1.59571	1.89297	0.94879	0.71091
Males			1.76777	1.35015	1.08237	0.83961	1.69251
Auckland TW			1.30558	1.44338	1.34568	1.01436	0.99916
Females			1.07143	1.59571	1.89297	0.94879	0.73598
Males			1.76777	1.35015	1.08237	1.19024	1.69251
Hawkes Bay TW ‡			-	-	-	2.5	2.5
Females			-	-	-	-	2.5
Males			-	-	-	2.5	-
Wētā (By Groups)							
Total occupants			0.26120	0.26361	0.22609	0.14150	0.05884
All wētā			0.29390	0.21480	0.22609	0.18687	0.15034
Tree Wētā (TW)			0.45395	0.30875	0.31383	0.22595	0.30349
Females			0.53236	0.47603	0.47295	0.29602	0.28357
Males			0.51299	0.37390	0.33587	0.25698	0.58770
Auckland TW			0.45395	0.30875	0.31383	0.24715	0.30880
Females			0.53226	0.47603	0.47295	0.29602	0.29245
Males			0.51299	0.37390	0.33580	0.30349	0.30349
Hawkes Bay TW ‡			-	-	-	0.81560	1.11803
Females			-	-	-	-	1.11803
Males			-	-	-	0.81560	-

†: Six s.f.

‡ Hawkes Bay Tree Wētā have only been found (alive) within the habitats of Lines 1 & 8 (Christensen 2002a), thus a CV for Lines (8) will always be a constant in this case: 2.5.

Appendix 9. Summary of Wētā Variability per Design: Habitat Types.

Total Lines (by groups): Treatment, Non-Treatment inclusive.
Total variation *i.e.* from start of Monitoring Programme 1997/98-2001/02

Wētā	CV (Coefficient of Variation)†.								
Lines	1	2	3	4	5	6	7	8	9
Total occupants	0.229	0.213	0.153	0.256	0.370	0.274	0.345	0.550	0.306
All wētā	0.433	0.268	0.230	0.290	0.508	0.342	0.327	0.834	0.497
Tree Wētā (TW)	0.695	0.186	0.232	0.242	0.637	0.600	0.309	1.369	0.767
Females	0.697	0.334	0.259	0.316	0.646	0.697	0.422	1.491	0.812
Males	0.919	0.236	0.395	0.319	0.698	0.677	0.365	1.565	1
Auckland TW	0.929	0.186	0.232	0.242	0.637	0.600	0.309	1.401	0.767
Females	0.903	0.333	0.259	0.316	0.646	0.697	0.422	1.630	0.812
Males	1.083	0.236	0.395	0.319	0.678	0.677	0.366	1.630	1
Hawkes Bay TW	0.559	-	-	-	-	-	-	1.630	-
Females	0.948	-	-	-	-	-	-	2.236	-
Males	1.087	-	-	-	-	-	-	1.491	-

†: Four s.f.
To note: this index of variability reflects both a mix of the temporal variation inherent in the samples (population) as well as the sampling error associated with the monitoring design (Link *et al.* 1994).

Table: Tree Wētā Roost Habitat-types.

Lines	Management	Reserve	Habitat-types[1]
1	Treatment	Boundary Stream	High-Altitude Mixed Broad-leaved Forest
2	Treatment	Boundary Stream	Kamahi Forest
3	Treatment	Boundary Stream	Tall Tawa & Podocarp Forest
4	Treatment	Boundary Stream	Lowland Mixed Broad-leaved Forest
5	Treatment	Boundary Stream	Tall Kanuka Forest
6	Non-Treatment	Thomas Bush	Tall Tawa & Podocarp Forest
7	Non-Treatment	Thomas Bush	Kamahi Forest
8	Non-Treatment	Cashes Bush	High-Altitude Mixed Broad-leaved Forest
9	Non-Treatment	Cashes Bush	Mixed Broad-leaved Forest

[1]Adapted from Walls (1995).

Appendix 10. Specialized Statistical Power Analysis (Standalone) Freeware.

Software	Internet URL	Calculation Method[1]
GPOWER	http://www.psychologie.uni-trier.de:8000/projects/gpower.html	E and A
MACANOVA	http://www.stat.umn.edu/~gary/macanova/macanova/.home.html	E
MONITOR	http://www.mp1-pwcr.ugs.gov./powcase/monitor.html	S
POWERPACK	ftp://ftp.stat.uiowa.edu/pub/rlenth/POWERPACK/	E
POWER PLANT	ftp://ftp.per.its.csiro.au/csiro-wa/biomterics/	E
PS	ftp://ftp.vanderbilt.edu/pub/biostat/ps.zip	A
STPLAN	http://odin.mdacc.tmc.edu/anonftp/	A
TRENDS	ftp://.im.nbs.gov/pub/software/CSE/wbs21515/trends.zip	E

Sources: (Thomas & Krebs 1997), world-wide web (*c* 2002).

[1] E = exact non-central distributions when appropriate; A = approximations to non-central distributions; S = simulations.