Threatened Species and Farming

Corangamite Water Skink:

Monitoring of modified management regimes.

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CONTENTS

SUMMARY

This study was funded by the *Ecologically Sustainable Agriculture Initiative* (ESAI) of the Department of Primary Industries. It is one of seven case studies investigating management techniques for threatened species in the context of improvements in agricultural production that are ecologically sustainable over the long-term.

The Corangamite Water Skink (*Eulamprus tympanum marnieae*) is an endangered skink species endemic to the Victorian Volcanic Plain in the states central southwest. The skink is a habitat specialist, inhabiting localities that combine deeply fissured basaltic rock piles with remnant arboreal vegetation and permanent or ephemeral wetlands. Approximately 90% of its populations are found on private land generally used for agricultural practices and in particular, livestock grazing, which has been identified as a threat to both the skink and its grassy shrubland/wetland habitat.

The aims of this project were to assess the effect of implemented management actions in particular grazing exclusion, on *E. t. marnieae* and its habitat, to maintain a population and habitat monitoring program to continually assess the effectiveness of management measures at ameliorating threatening agricultural processes, and to provide information that can be readily utilised by stakeholders on how best to manage *E. t. marnieae* habitat and wetlands in general.

The results of this project show that any significant change in adult *E. t. marnieae* abundances in response to grazing exclusion through fencing could take numerous years to detect. However, recovery may be quicker depending on the degree of the previous disturbance. The rate of recovery is potentially linked to the response and rate of change of the vegetation and other habitat features in the absence of grazing.

Little change in *E. t. marnieae* habitat was detected in the 12 months following grazing exclusion, with an increase in perennial grass cover the only significant response. Recovery and change from other vegetation life-forms, however, may take longer to manifest. Factors that are likely to influence rate of change and recovery of the vegetation include degree of disturbance and degradation, species composition, habitat structure, and established or persistent seed bank.

As the area occupied by *E. t. marnieae* does not exceed 2% of a landowner's property and in most cases even less, combined with the fact that the skink and its rocky habitat is confined to the verges of wetlands, *E. t. marnieae* lends itself to being an ideal species for retention and preservation on private property with no net loss in agricultural productivity. Ongoing long term monitoring following grazing exclusion is, however required, while alternative management such as crash grazing or weed spraying may be needed. Supplementary planting may also be undertaken to help restore the original vegetation composition.

1 INTRODUCTION

1.1 Ecologically Sustainable Agriculture Initiative (ESAI)

“Threatened Species and Farming” is a sub-project of the ESAI. This project will identify how agricultural practices might be modified to help conserve selected threatened species as part of working toward ecological sustainability. The project will document case studies of selected threatened species in five bioregions: the Victorian Riverina, Wimmera, Victorian Volcanic Plain, Gippsland Plain and Glenelg Plain. The farms considered include examples from the meat, wool, dairy and grains industries. This case study focuses on the Corangamite Water Skink (*Eulamprus tympanum marnieae*).

1.2 The Corangamite Water Skink

The Corangamite Water Skink (*Eulamprus tympanum marnieae*) is currently recognised as critically endangered. It is listed under the following international, commonwealth, state and territory legislation and documentation: IUCN (IUCN 1996); ANZECC (ANZECC 1995); Action Plan for Australian Reptiles (Cogger *et al.* 1993); *Environment Protection and Biodiversity Conservation Act 1999* (Commonwealth); Schedule 2 of the *Flora and Fauna Guarantee Act* 1988 (Vic.); Advisory List of Threatened Vertebrate Fauna in Victoria – 2003 (DSE 2003).

Eulamprus tympanum marnieae is a recently described member of the *Eulamprus quoyii* species complex. This complex is a group of medium to moderately large Australian lygosomine scincid lizards, who’s association with water or very moist habitats throughout the coastal drainage systems of eastern and south eastern Australia has led them to be commonly known as "water skinks".

Adult *E. t. marnieae* may grow to 100 mm snout-vent length, with a tail length up to 150 mm and a body mass of up to 25 grams. The dorsal colouration of *E. t. marnieae* is pale olive to dark yellowish brown with distinct transverse bars, while the ventral colouration is bright yellow overlain on the underside of the body by two thick black longitudinal lines of either small specks, thick patches or bars originating from the chest and extending to the groin (Hutchinson and Rawlinson 1995).

1.3 Taxonomic Status

E. t. marnieae is currently recognised as a subspecies of the widespread, cool temperate Southern Water Skink (*E. t. tympanum*) (Hutchinson and Rawlinson 1995). It is distinguished from *E. t. tympanum* by its very small midbody and paravertebral scales, its black dorsal markings arranged as short irregular transverse bars and bold ventral patterns of black longitudinal strips on a bright yellow background. The initial conclusion following the discovery of *E. t. marnieae* was that it represented a new species. Subsequent collections of small numbers of individuals, however, suggested that intergradation with *E. t. tympanum* had occurred. These specimens appeared to be intermediate in both colour pattern and scalation. Suggestions of intergradation were further supported with the discovery of additional morphologically intermediate populations in 1997-98.

Recent molecular work has also supported the notion of its subspecies status (Scott and Keogh 2003). This work, however, has also revealed that the northern and southern *E. t. marnieae* populations have had different evolutionary origins. Both the northern and southern populations have evolved from distinctly different (at the molecular level) *E. t. tympanum* populations and are, therefore, significantly different from each other at the molecular level. Thus, *E. t. marnieae* requires urgent reclassification and separation into two distinct subspecies.

1.4 Distribution and habitat

E. t. marnieae was discovered in 1963 near Lismore, and described in 1995. Currently *E. t. marnieae* is known from 31 sites (2 extinct) that represent ten discrete extant populations and one extinct population. It is endemic to the Victorian Volcanic Plain in the states central southwest where it is restricted to three regions of the Later Newer Basalts east and northwest of Lake Corangamite.

Most *E. t. marnieae* populations are patchily distributed and isolated, with the bulk of sites having very small areas of available habitat. Most sites support extremely small populations with the estimated adult abundance at sites as of May 2000 ranging from 4 to 431 animals, with an average of 84 adult animals per site (Peterson 2000).

Within these Later Newer Basalt regions *E. t. marnieae* is a habitat specialist, inhabiting localities that combine deeply fissured basaltic rock piles with remnant arboreal vegetation and permanent or ephemeral wetlands (Robertson and Lowe 1999; Peterson 2000). The vegetation associated with these sites include a mixture of native and exotic species, with the lizards favouring locations that contain either or a mixture of Tree Violet (*Hymenanthera dentata*), African Box-thorn (*Lycium ferocissimum*), Scrub Nettle (*Urtica incisa*), Variable Groundsel (*Senecio pinnatifolius*) and Tall Sedge (*Carex appressa*) (Peterson 1997).

1.5 Ecology

E. t. marnieae is a diurnally active, heliothermic skink. It is active from September to the end of May, however, animals have been seen basking in the middle of July during sunny breaks (Peterson pers. comm.). *E. t. marnieae* is viviparous, producing one clutch per year of between two and seven live young. Parturition occurs between late December and late January. Ovulation in *E. t. marnieae* occurs in late October to early November, while males have been recorded with enlarged testes in April and September, with regressing testes in October to November, and fully regressed testes in December (Hutchinson and Rawlinson 1995). Females appear to first reproduce at two years of age however, this may vary between populations. Litter size and mass, increase with female size, while litter mass averages approximately 30% of female mass (Peterson 2002).

Following parturition it appears that both temporal and spatial partitioning of the cohorts occurs, with the majority of neonates occupying different microhabitat and activity periods from that of the adults. It is not known whether this is due to aggressive behaviour of the adults towards the neonates or due to specific requirements of the neonates, such as smaller prey items, which may only occur in the other microhabitat (Peterson 2002).

At present little is known about offspring mortality, recruitment into the adult population, and population age structure. This will, however, be rectified following the analysis of data collected as part of a 6 year mark-capture-recapture program. Life expectancy of *E. t. marnieae* is at present unknown, however, *E. t. tympanum* may live for up to 15 years (Rohr 1997).

Like *E. t. tympanum* it is assumed that *E. t. marnieae* possess territories and defined home ranges which vary between sexes, age cohorts and the time of the year (i.e. mating and post-mating periods). Whether these territories and home ranges are affected by population size and habitat availability is at present unknown.

E. t. marnieae has been found to be predominately insectivorous, like its congeners, feeding on terrestrial prey such as spiders, beetles and ants, as well as aquatic prey including mayfly and dragonfly larva (Knights 2003). It also consumes the fruit of the Tree Violet (*Hymenanthera dentata*), which may be an important component of the diet during some periods (Peterson 1997). The lizard may also play an important role in the dispersal and germination of this plant. The skin and seeds of the introduced African Boxthorn (*Lycium ferocissimum*) and other unidentified seeds have also been found in scats (Knights 2003).

Unlike other water skinks, *E. t. marnieae* is extremely shy. Usually observed perched on a large rock pile, it will often flee and take cover when a human observer is still tens of metres away (Hutchinson and Rawlinson 1995). It can and does swim, but usually takes refuge in the deep gaps and fissures in the rock piles.

1.6 Threatening Processes

The main factors involved in the decline of *E. t. marnieae* appear to be habitat loss and fragmentation, and degradation of remaining areas of habitat by a range of processes, in particular those induced by agricultural practices, including: changed grazing regimes, weed invasion, changed fire regimes, rock removal, changed hydrology and/or water quality, use of agricultural chemicals, and the impacts of introduced animals (foxes, cats, rabbits, mice, sheep, cattle) either by predation or by grazing (Robertson 1998; Robertson and Peterson 2000).

Approximately 90% of *E. t. marnieae* populations are found on private land generally used for agricultural practices. These practices include dairy farming, grain cropping, and beef and sheep farming. Aspects of all of these agricultural practices threaten the associated *E. t. marnieae* populations and their habitats.

Most sites where *E. t. marnieae* persists are subject to grazing by hard hoofed introduced herbivores such as dairy and beef cattle, and sheep. Grazing by introduced herbivores has been found to negatively impact on reptile population densities through a reduction in biomass, and as a consequence a loss of cover, habitat and social structure, invertebrate prey and direct casualties (Jones 1981; North *et al.* 1994; Hadden and Westbrooke 1996; Smith *et al.* 1996; Fair and Henke 1997; Read 2002). The indigenous vegetation of the rocky shrubland/wetland habitats of *E. t. marnieae* may be particularly susceptible to grazing. As some native shrubs and herbs have been shown to be important to *E. t. marnieae* (Peterson 1997; Peterson 2000), it is likely that heavy grazing will be detrimental to *E. t. marnieae*. Grazing may also be detrimental to neonatal and juvenile *E. t. marnieae* as the microhabitat they tend to inhabit is heavily vegetated and consequently heavily grazed. Grazing of the vegetation could reduce the amount of vegetation cover and/or the abundance and diversity of prey available to the neonates (Peterson 2001). Cattle may also be a threat by directly trampling the neonates or the scattered rocks, under which they may be sheltering.

Livestock also threaten the water body itself by pugging and disturbing the sediments, grazing the aquatic vegetation and increasing the nutrient loads through defecation leading to eutrophication of the water body, which facilitates algal blooms and decreases water quality (Hull 1993).

1.7 Grazing Management

Due to the threats and impacts livestock grazing pose to *E. t. marnieae* and its grassy shrubland/wetland habitat and to avoid further habitat degradation, it may be prudent to exclude grazing from *E. t. marnieae* habitat wherever possible (Robertson and Peterson 2000). In many instances, given the sympathetic attitudes of land managers, it may be possible to exclude livestock grazing through fencing of the small areas occupied by *E. t. marnieae* such that adverse impacts on the habitat are minimised with little, if any, decrease in agricultural productivity (Robertson 1998).

Stock exclusion at sites is potentially a valuable method of encouraging regeneration of native shrubs, which are an important component of *E. t. marnieae* habitat (Peterson 1997; Knights 2003). Ground layer species could also benefit. The exclusion of stock may also increase the survivorship and recruitment of neonate and juvenile *E. t. marnieae* by reducing grazing in the microhabitat they occupy.

Overall, grazing exclusion should alleviate and minimise any impacts of stock on *E. t. marnieae* and its habitat, allow regeneration and enhancement of the habitat, enable the skink population to stabilise or increase, and conserve the wetland community as a whole. Resultant vegetation changes must, however, be closely monitored to ensure that weed invasion does not become a problem and that ground layer structure remains suitable or improves (Robertson and Peterson 2000).

Since 2003, in Southwestern Victoria numerous projects have been undertaken on private land to exclude livestock from wetland habitats, including a number of sites inhabited by *E. t. marnieae*. The actually short and long term benefits or detriments of this management action, however, are yet to be defined.

1.8 Project Aims

This research project attempts to assess the effect of grazing regulation/exclusion on both *E. t. marnieae* and its habitat. The primary aims of this project are to:

- Assess the effect of implemented management actions, in particular grazing regulation/exclusion on both *E. t. marnieae* and its habitat;
- Ensure threatening agricultural processes, which modify habitat and threaten *E. t. marnieae* are managed appropriately;
- Ensure a stable and/or increasing total population is maintained, as determined by appropriate techniques for estimating and monitoring *E. t. marnieae* populations;
- Maintain a population and habitat monitoring program to continually assess the effectiveness of management measures at ameliorating threatening agricultural processes, and to identify any threats as they may arise;
- Statistically review the data from the monitoring strategy to assess its effectiveness at meeting monitoring objectives, and modify monitoring protocol if required;
- Involve the community in management of *E. t. marnieae* and its habitat;
- Provide information that can be readily utilised by agricultural extension officers, natural recourse managers and landowners on how best to manage *E. t. marnieae* habitat and wetlands in general with no net loss in agricultural productivity.

2 METHODS

2.1 Site selection

A subset of five *E. t. marnieae* sites was chosen from a total of 14 sites where monitoring to track population change, estimate population size, density, structure and recruitment had been conducted since 1998 as part of the author's Ph. D research project. The five sites were chosen from across the geographical range of *E. t. marnieae* and represent different land use practices including wool production, dairy and beef farming. Site choice was also influenced by landholder agreement to allow the establishment of the experimental fencing, by the ease at which fencing could be erected at the sites, and by the ability to observe any population change within the prescribed annual monitoring intensity (Scroggie 2002). Descriptions of the five sites are given below.

Site 1: Lake, Hays Rd Dreeite

Geomorphological landform: Basalt Plains II (Later Newer Basalts)

Ecological Vegetation Community (EVC): Stony knoll shrubland/Unidentified wetland community

The Hays Rd site is situated 3 km northeast of Dreeite on the Victorian Volcanic Plain. The site consists of a spring fed freshwater lake surrounded by steep basalt ridges. The lake margin is dominated by aggregated basalt boulders of varying size, a sparse shrub layer of Tree Violet (*Hymenanthera dentata*) and Kangaroo apple (*Solanum aviculare*) and a ground layer of Tall Sedge (*Carex appressa*), Variable Groundsel (*Senecio pinnatifolius*), Scrub Nettle (*Urtica incisa*) and a mix of exotic annual grasses. The shrub layer of *H. dentata* continues up onto the ridge with the occasional Blackwood (*Acacia melanoxylon*), Black Wattle (*A. mearnsii*) and African Boxthorn (*Lycium ferocissimum*), and an understorey of exotic annual grasses interspersed by basalt outcrops. The site has been grazed by beef cattle for the past 20 years, but prior to that it was grazed by sheep as part of the Dreeite Station, which incorporated all Dreeite *E. t. marnieae* populations.

Site 5: The Points Lake Colac

Geomorphological landform: Basalt Plains II (Later Newer Basalts)

EVC: Stony knoll shrubland/Freshwater to slightly brackish lake verge

The Points is located on the western shore of Lake Colac approximately 5 km east of Cororooke. Lake Colac is a large, permanent freshwater lake that covers approximately 2,668ha (Williams 1992). The Points is comprised of a number of large basalt lobes or tongues extending out into the lake from the main lava field that forms the lakes north western shoreline. The lobes consist of steep exposed basalt flows extending into the lake and heavily vegetated ridges interspersed by basalt outcrops. *Urtica incisa, Senecio pinnatifolius*, Sharp Clue-sedge (*Schoenoplectus pungens*) and a mix of native and exotic annual and perennial grasses and herbs grow throughout the steep basalt flows and at their base, while the ridges are vegetated with exotic annual grasses and sparse *Hymenanthera dentata*. The site is and historically was grazed by dairy cattle.

Site 11: Lake Logan Derrinallum

Geomorphological landform: Basalt Plains II (Later Newer Basalts)

EVC: Stony knoll shrubland/Freshwater to slightly brackish lake verge

Lake Logan is a shallow ephemeral freshwater lake situated approximately 2km north of the township of Derrinallum. The site is located on the western shore of the lake and is comprised of a small shelf of black basalt approximately 300m long and on average 6m wide, and a few other smaller basalt outcrops. The flanks of the basalt shelf and a number of the rock fissures are vegetated with a mixture of native and exotic grasses and herbs including *Austrodanthonia* *caespitose,* Salt Pratia (*Lobelia irrigua*), Knotty Spear-grass (*Stipa nodosa*) and Soft Brome (*Bromus hordeaceus ssp. hordeaceus*). No shrub or tree layer exists at the site. The site is and was historically grazed by sheep. The site is home to the threatened Salt Tussock-grass (*Poa sallacustris*) and the threatened Growling Grass Frog (*Litoria raniformis*).

Site 15: Deep Lake Nerrin Nerrin
Geomorphological landform: Basalt Plains II (Later Newer Basalts)
EVC: Plains grassland/Plains sedgy wetland

Deep Lake is situated in the centre of a series of six lakes that form the Nerrin Nerrin wetlands. The lake is a permanent freshwater lake enclosed on its northern, eastern and western shores by large basalt shelves up to 5m high. The basalt shelves have a sparse shrub layer of *Hymenanthera dentata* and back from the exposed basalt shelves and at their base is a dense ground layer of native and exotic grasses and herbs including *Austrodanthonia sp., Austrostipa sp.,* and Toowoomba Canary-grass (*Phalaris aquatica*). The site is grazed by sheep.

Site 16: Lake Coragulac Alvie
Geomorphological landform: Basalt Plains II (Later Newer Basalts)
EVC: Stony knoll shrubland/Crater Lake (Brackish)

Lake Coragulac is one of four maar crater lakes within the Red Rock volcanic complex just south of the township of Alvie (Joyce 1988). The lake is a freshwater lake covering an area of 22.4 ha and is enclosed by steep slopes which rise up to 70m above the lake floor (Timms 1983). Thought to be a permanent water body, the lake actually dried out in 2000 and has remained dry since. The site is located on the north eastern corner of the lake and consists of approximately 500m of aggregated basalt rocks of varying sizes running along the lake shoreline. A shrub layer of *Hymenanthera dentata, Lycium ferocissimum* and *Solanum aviculare*, which in some places is quite dense, is present throughout the basalt aggregations. A dense ground layer of *Urtica incisa, Senecio pinnatifolius* and a mix of native and exotic annual and perennial grasses and herbs vegetate many of the rock fissures and areas around the aggregations. In some sections the exotic grass Kikuyu (*Pennisetum clandestinum*) has engulfed the aggregations completely covering the basalt rocks. The site was grazed by beef cattle until 1999 when it was purchased and covenanted through Trust for Nature. Grazing by alpacas was introduced into the site in winter 2003.

2.2 Population monitoring

The population monitoring at the five sites was undertaken along pre-established 8m x 200m strip transects that were marked at 10m intervals with small metal tags and brightly coloured and numbered flagging tape at eye level, or with large numbers painted on the vertical surface of conspicuous basalt rocks. A transect width of 8m was used as it incorporates most of the potential lizard habitat along the lake shoreline (Peterson unpublished data).

The five transects were monitored every four to six weeks from October to May during the 2003/04 and 2004/05 field seasons totalling five monitoring seasons per site, per field season. Prior to 2003/04, sites had been monitored between two and five times per field season for the previous five or six field seasons. Transect monitoring began between the hours of 8:30am to 9:30am (Eastern Standard Time). Each monitoring session took approximately 2-3 hours to complete. The starting location of either 0m or 200m was alternated between monitoring visits.

Monitoring was undertaken as a visual transect count. This involved walking down the centre of the transect and stopping approximately every 5m for approximately 2-3 minutes. With the aid of binoculars, each individual lizard 5 -10 m further along the transect, their age class (neonates/juveniles (30-50mm), subadults (51-70mm) and adults (>70mm)) and their position along the transect were recorded. The reproductive status of females was also recorded. Gravid females are easily identified due to their large and distended abdomens and small head in proportion to their body size during spring and early summer.

Given the ectothermic physiology of reptiles, weather conditions may greatly affect their emergence and activity (Heatwole and Taylor 1987). Therefore, all monitoring was conducted during periods of optimal weather conditions for lizard activity. The optimal conditions found for *E. t. marnieae* are warm, calm days with an ambient temperature between 18 ^{o}C and 26 ^{o}C (Peterson unpublished data). Monitoring was only undertaken on windy days if the wind was blowing from the opposite direction from the main aspect of the transect, when the transect was sheltered from the wind.

2.3 Statistical review

As proposed in the ESAI Corangamite Water Skink Research and Management Plan (Peterson 2002), a statistical review of the *E. t. marnieae* data from the previous monitoring strategy to assess its effectiveness at meeting monitoring objectives was conducted prior to the commencement of further monitoring associated with this project. The statistical review was conducted by Dr Michael Scroggie from the Arthur Rylah Institute for Environmental Research.

The review (Scroggie 2002)(Appendix II) found that the originally implemented monitoring strategy was sufficient to detect biologically significant changes in population parameters over a short period of time i.e. the 3 year time frame of this project. As the monitoring intensity applied in this project was as frequent, and in some instances more frequent than conducted previously, it was confidently assumed that over the time frame of the project any significant changes in the populations, due to the modified management (grazing exclusion), would be detected using the monitoring method outlined above.

2.4 Grazing exclusion plots

At each of the five sites, a 100m x 15m section of the pre-established 200m monitoring transect was fenced off to examine the effects of complete stock exclusion on the *E. t. marnieae* population and its habitat, in particular vegetation type, density and cover. The nature of the 200m transect allowed it to be grided into twenty 10 x 8m quadrats which enabled manipulation and comparisons within the transect. The fencing grids consisted of steel star pickets placed approximately every 5m, with two hot wires powered by a solar energizer and one or two plain wires. The grids were erected by the author and a Green Corps crew from the Greening Australia *Borrell-a-kandelop* project in August 2003.

2.5 Habitat assessment

The effects of the modified management (grazing exclusion) on the habitat of *E. t. marnieae* were assessed by examining differences and changes in vegetation cover and diversity among sections of the transects where stock were excluded, as opposed to sections where grazing was continued (control).

After the erection of the fencing plots at the five monitoring transects, grazing initially continued along the entire transect. Stock was then removed from the fenced sections in spring 2003 after the habitat was assessed. Habitat assessment was then repeated in spring 2004. The habitat assessment per transect took 2-3 days to complete.

To assess the habitat, the % vegetation cover of each species at <10cm, 11-25cm, 26-50cm, 51-75cm and >76cm, % cover of bare ground, rock, water, litter and substrate were estimated. This was done by laying four 6m transects radiating at 90^{o} from a central point of origin and recording at 1 m intervals and at the point of origin, the contact with the habitat variables at the substrate, <10cm, 11-25cm, 26-50cm, 51-75cm and >76cm on a calibrated 1m long ruler. This was undertaken in the centre of the transect at 5m intervals along its 200m length. This procedure provided 25 contact points per central point and allowed the percentage of each variable (i.e. percent of contacts with each substrate and vegetation type) within each 5 x 8m section of the transect to be estimated (Martin and Salvador 1995).

2.6 Statistical analysis

Examination of the population monitoring data was undertaken by Dr Michael Scroggie from the Arthur Rylah Institute for Environmental Research (Scroggie 2005). No formal statistical analysis of this data was possible due to the short time (2 years) that had elapsed since the fencing was erected on the treated portions of the transects. Whilst a formal statistical analysis of the effects of fencing was not possible, the raw counts of adult *E. t. marnieae* on the fenced and unfenced portions of habitat at the five sites were graphed to allow an initial visual assessment of any changes in population trends in response to grazing exclusion. As only 2 years of post-fencing data have been collected, any response in adult abundance thus far is expected to be minimal, due to the time delays in any effect on survival and reproductive rate due to the fencing (Scroggie 2005).

The impact of grazing exclusion on the habitat of *E. t. marnieae* was analysed using a repeated measures randomized complete block analysis of variance (Quinn and Keough 2002). The sites represented five transect replicates with grazing exclusion/continued grazing as the treatments. Date (before grazing exclusion versus after grazing exclusion) represented the repeated measure. The impact of grazing exclusion on the vegetation cover and other habitat attributes was examined by the treatment x date interaction. Response variables were % rock and vegetation ground cover as well as total vegetation cover, herb, annual grass, perennial grass, sedge/non woody shrub and wood shrub cover at <10cm, 11-25cm, 26-50cm, 51-75cm, >76cm height and total cover. All % cover data were arcsine transformed prior to analyses.

The impact of grazing exclusion on the 10 dominate plant species at each site was assessed using a two-way ANOVA of the mean total % cover of each species measured before grazing exclusion (spring 2003) and after grazing exclusion (spring 2004) with grazing exclusion/continued grazing as treatments. All % cover data were arcsine transformed prior to analyses.

In environmental management, especially when there is a chance of serious environmental damage a precautionary principle is to use an alpha level set higher than the traditional 0.05, thereby minimising the chance of a Type II error (Quinn and Keough 2002). Due to the potential damage changed management may have on *E. t. marnieae* habitat an alpha level of 0.1 was used.

3 RESULTS

3.1 Exploratory analysis of the effects of grazing exclusion on Corangamite Water Skink populations

Population monitoring counts from the grazed (unfenced) and un-grazed (fenced) portions of the transects are presented in Figures 1-5 (figures from Scroggie 2005). The figures include counts conducted both before and after the erection of the livestock exclusion fencing on part of each of the transects. No formal statistical analysis was undertaken due to the short time that had elapsed since the fencing was erected on the treated portions of the transects (Scroggie 2005). Visual inspection of the plots in Figures 2, 3 and 4 does not show any obvious divergence in trends in abundance of adult skinks between the grazed and un-grazed portions of the transects.

The results in Figure 1 show no obvious divergence between the grazed and un-grazed plots since the fencing was erected at site 1. However, the number of censured adult lizards increased in both plots after fencing was established. It is interesting that the erection of the grazing exclusion fence coincided with an overall decrease in stocking rate from across the entire site (G. Mason pers. comm.).

At site 16, there was a distinct difference in the number of adult lizards observed in the grazed and un-grazed portions of the transect in the second year following the fence being erected (Figure 5), with more animals observed in the un-grazed section of the transect but no overall increase in abundance across the transect.

3.2 The impact of grazing exclusion on the habitat of the Corangamite Water Skink

Figures 6 and 7 and Figures 8 and 9 show the grazed (unfenced) and un-grazed (fenced) sections of the monitoring transect at Site 5 and Site 15 respectively, 12 months after the fencing was erected and grazing was excluded.

The pooled mean percentage cover for the 41 vegetation and habitat attributes measured in the grazed and un-grazed sections from the five transects in 2003 and 2004 can be found in Appendix I Table 1. The mean percentage cover for each of the vegetation and habitat attributes in the grazed and un-grazed sections for each of the five transects in 2003 and 2004 can be found in Appendix I Tables 2-6, while Appendix I Tables 7-11 display the mean percentage cover for each of the 10 dominate plant species measured in the grazed and un-grazed sections for each of the five transects in 2003 and 2004.

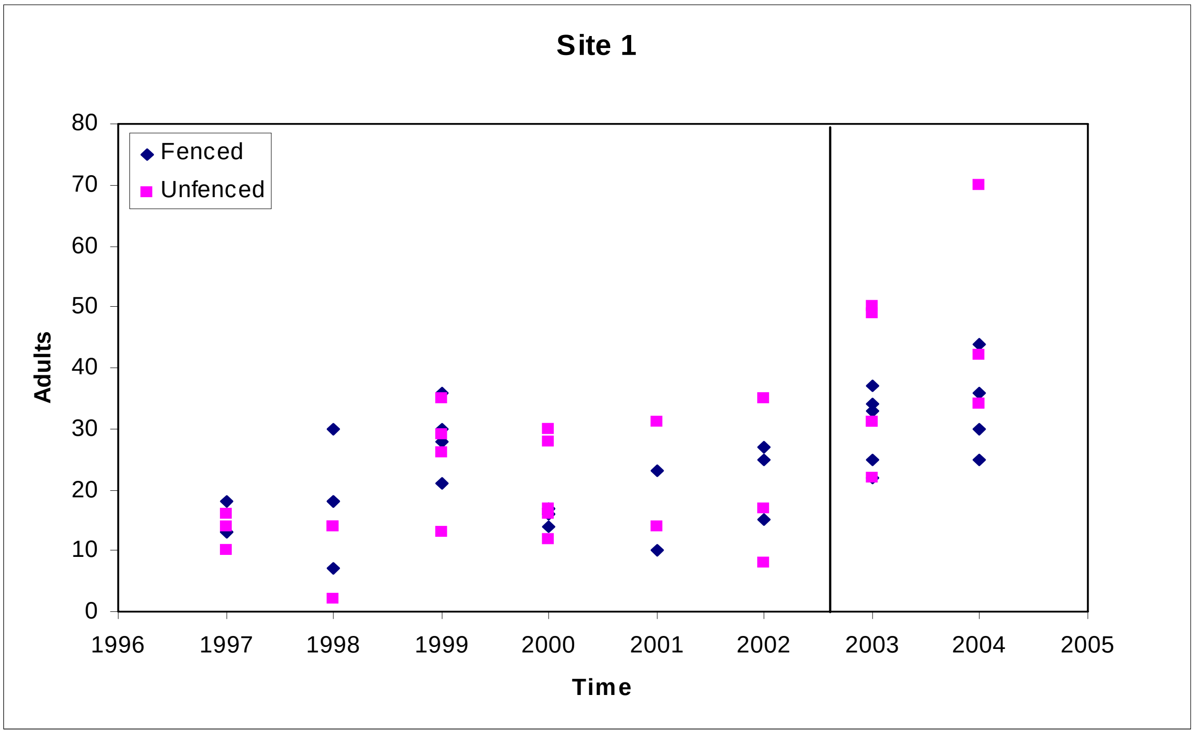

Figure 1. Monitoring counts of adult skinks in fenced and unfenced sections of the transect at Site 1 before and after fencing. The vertical line indicates the time at which the fence was erected on the fenced portion of the transect, excluding grazing by livestock.

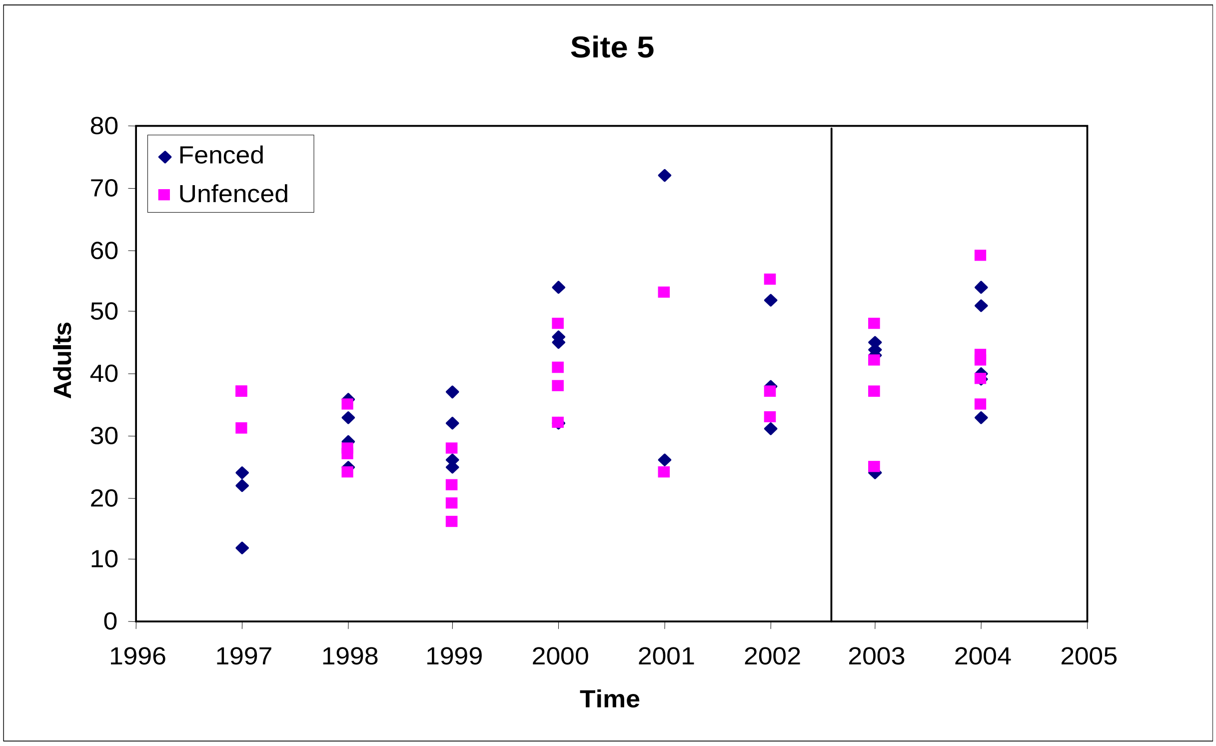

Figure 2. Monitoring counts of adult skinks in fenced and unfenced sections of the transect at Site 5 before and after fencing. The vertical line indicates the time at which the fence was erected on the fenced portion of the transect, excluding grazing by livestock.

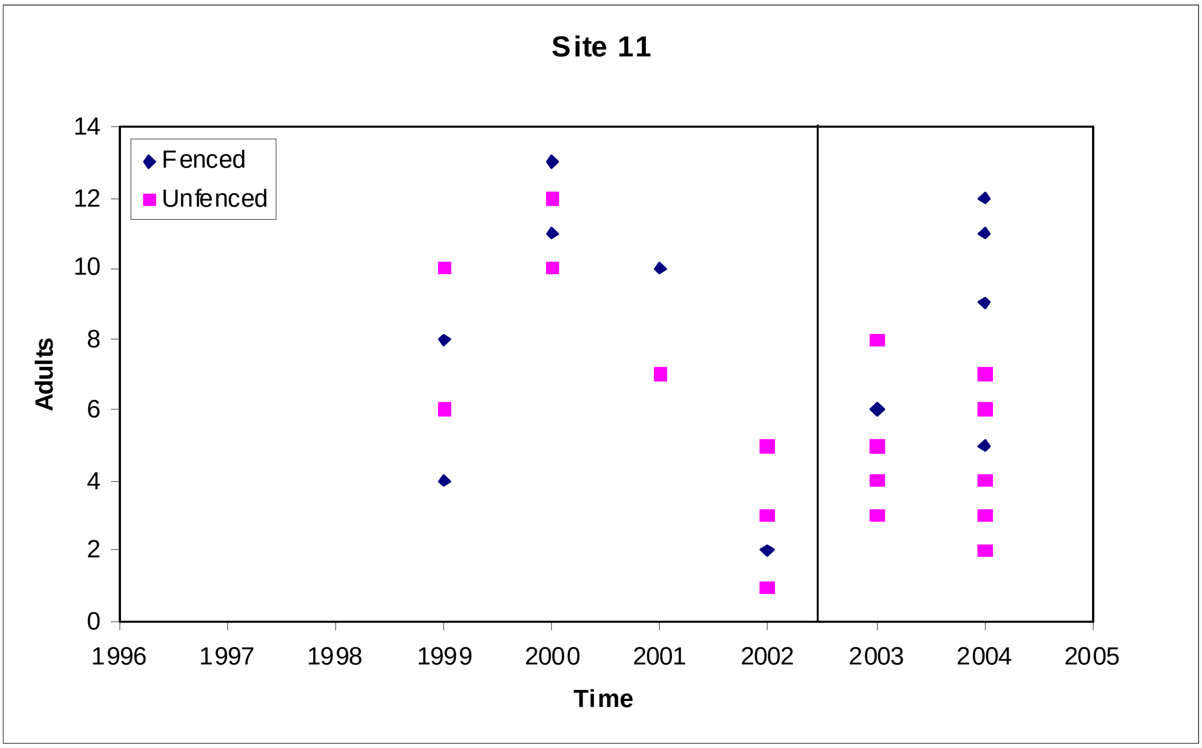

Figure 3. Monitoring counts of adult skinks in fenced and unfenced sections of the transect at Site 11 before and after fencing. The vertical line indicates the time at which the fence was erected on the fenced portion of the transect, excluding grazing by livestock.

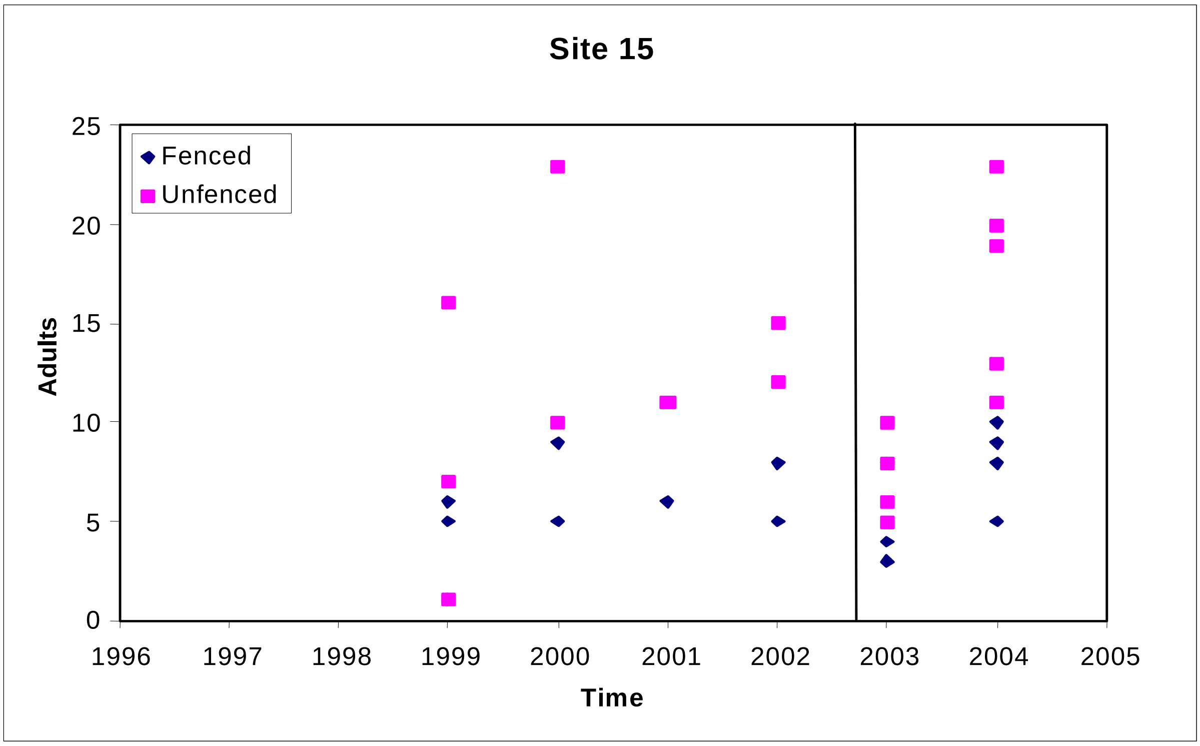

Figure 4. Monitoring counts of adult skinks in fenced and unfenced sections of the transect at Site 15 before and after fencing. The vertical line indicates the time at which the fence was erected on the fenced portion of the transect, excluding grazing by livestock.

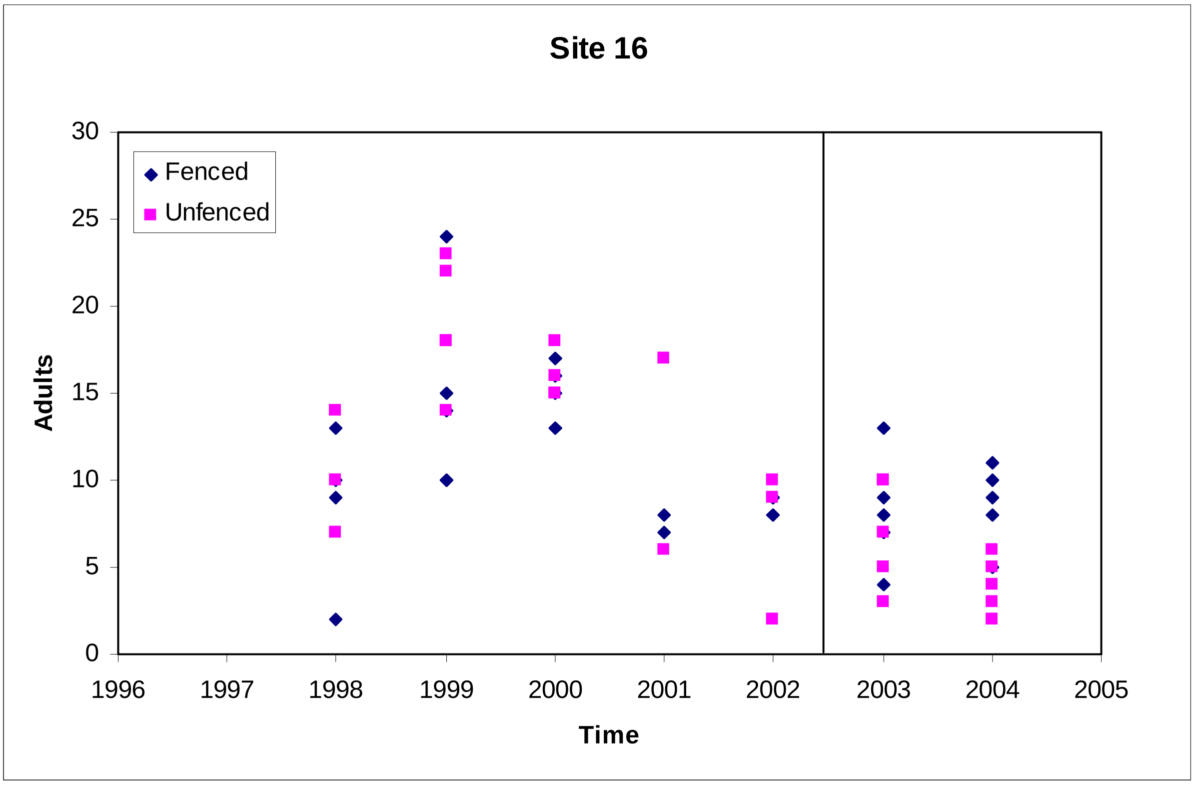

Figure 5. Monitoring counts of adult skinks in fenced and unfenced sections of the transect at Site 16 before and after fencing. The vertical line indicates the time at which the fence was erected on the fenced portion of the transect, excluding grazing by livestock.

Significant interactions between date and treatment were only found in the percentage vegetation cover at ground level and the percentage perennial grass cover >76cm in height between the un-grazed sections of the combined five transects and the grazed sections after the fencing was erected (Table 1). In the case of the percentage vegetation cover at ground level, the percentage cover decreased in the grazed section between 2003 and 2004, while no real change was detected in the un-grazed sections once the grazing was removed (Repeated measures randomized complete block ANOVA: $P = 0.085$)(Figure 10). In regard to the percentage perennial grass cover >76cm, the percentage cover was found to increase in the un-grazed sections once grazing was removed, while no change was recorded in the grazed sections (Repeated measures rcb ANOVA: $P = 0.098$)(Figure 11).

The interaction for the percentage perennial grass cover 11-25cm and the total percentage perennial grass cover were both close to significant (Repeated measures rcb ANOVA: $P = 0.116$ and $P = 0.151$ respectively)(Table 1). In both cases an increase in perennial grass cover was observed in the un-grazed plot, while no real change was witnessed in the grazed section (Appendix I Table 1). No other trends where evident between the grazed and un-grazed sections of the transects from the pooled data (Appendix I Table 1).

The results of the two-way ANOVA to assess the impact of grazing exclusion on the percentage cover of each of the 10 dominate plant species at each transect can be found in Table 2. Significant interacts were found between date and treatment for two species at both Transects 1 and 15, and for three species at both Transects 5 and 16 (Figure 12). Only three of the species were found to increase in percentage cover following grazing exclusion in comparison to the section of the transect still grazed (*Acetosella vulgaris* (Sheep Sorrel) at Transect 1, *Marrubium vulgare* (Horehound) at Transect 15 and *Holcus lanatus* (Yorkshire Fog) at Transect 16). Percentage cover of *Trifolium* sp. (Clover) was found to decrease considerably in both sections of the transect in 2004 at both Transects 1 and 5 while *Trifolium fragiferum* (Strawberry Clover) was found to largely decrease in the grazed section of Transect 16 while slightly increasing in the non-grazed section (Figure 12). *Dactylis glomerate* (Cocksfoot) was also found to largely decrease in the grazed section of Transect 16 while sightly decreasing in the non-grazed section in 2004 (Figure 12). At Transect 5 *Senecio pinnatifolius* (Variable Groundsel) was the only species found to noticeably decrease in percentage cover in a section once grazing was excluded while *Cynosurus echinatus* (Rough Dog's-tail) was found to greatly decrease in the un-grazed section following grazing exclusion but largely increase in the grazed section of the transect in 2004 (Figure 12). Finally *Phalaris aquatica* (Toowoomba Canary-grass) at Transect 15 was found to decrease in the grazed section between 2003 and 2004, while only slightly increase in the un-grazed section following grazing exclusion (Figure 12).

Figure 6. Grazed section of the monitoring transect at Site 15 Deep Lake Derrinallum.

Figure 7. Fenced section of the monitoring transect at Site 15 Deep Lake Derrinallum.

Figure 8. Grazed section of the monitoring transect at Site 5 Lake Corangamite.

Figure 9. Fenced section of the monitoring transect at Site 5 Lake Corangamite.

Table 1. Repeated measures randomized complete block analysis of variance comparing vegetation and habitat attribute cover in grazed and un-grazed plots (Treatment) in spring 2003 and 2004 (Date).

Table 1 Continue. Repeated measures randomized complete block analysis of variance comparing vegetation and habitat attribute cover in grazed and un-grazed plots (Treatment) in spring 2003 and 2004 (Date).

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Table 1 Continue. Repeated measures randomized complete block analysis of variance comparing vegetation and habitat attribute cover in grazed and un-grazed plots (Treatment) in spring 2003 and 2004 (Date).

Figure 10. Mean percentage vegetation cover from the grazed and un-grazed sections of the five transects pre-fencing (2003) and post fencing (2004).

Figure 11. Mean percentage perennial grass cover >76cm from the grazed and un-grazed sections of the five transects pre-fencing (2003) and post fencing (2004).

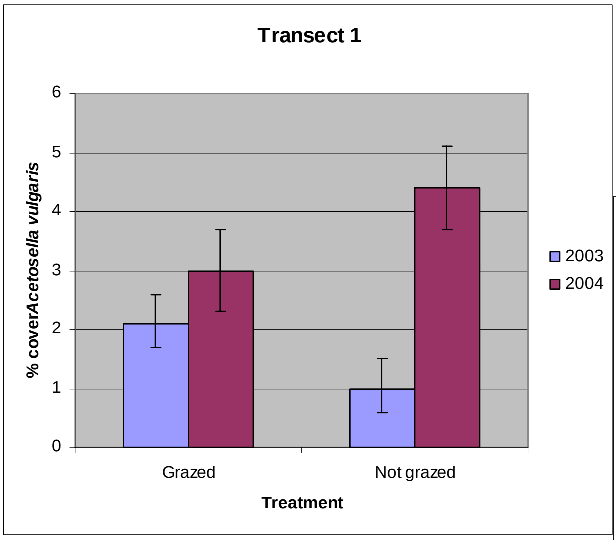
Table 2. Two-way ANOVA on mean percentage cover for each of the 10 dominant species in grazed and un-grazed sections (Treatment) of Transect 1 in spring 2003 and 2004 (Date). # indicates an indigenous native species. Other species are introduced.

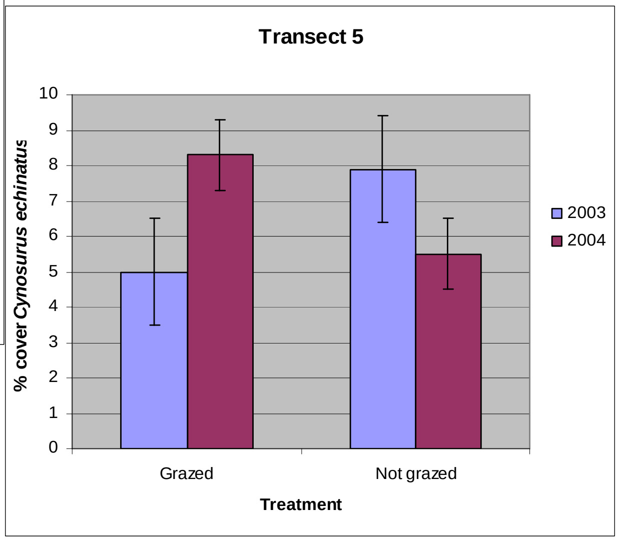
Table 2. Two-way ANOVA on mean percentage cover for each of the 10 dominant species in grazed and un-grazed sections (Treatment) of Transect 5 in spring 2003 and 2004 (Date). # indicates an indigenous native species. Other species are introduced.

Table 2. Two-way ANOVA on mean percentage cover for each of the 10 dominant species in grazed and un-grazed sections (Treatment) of Transect 11 in spring 2003 and 2004 (Date). # indicates an indigenous native species. Other species are introduced.

Table 2. Two-way ANOVA on mean percentage cover for each of the 10 dominant species in grazed and un-grazed sections (Treatment) of Transect 15 in spring 2003 and 2004 (Date). # indicates an indigenous native species. Other species are introduced.

Table 2. Two-way ANOVA on mean percentage cover for each of the 10 dominant species in grazed and un-grazed sections (Treatment) of Transect 16 in spring 2003 and 2004 (Date). # indicates an indigenous native species. Other species are introduced.





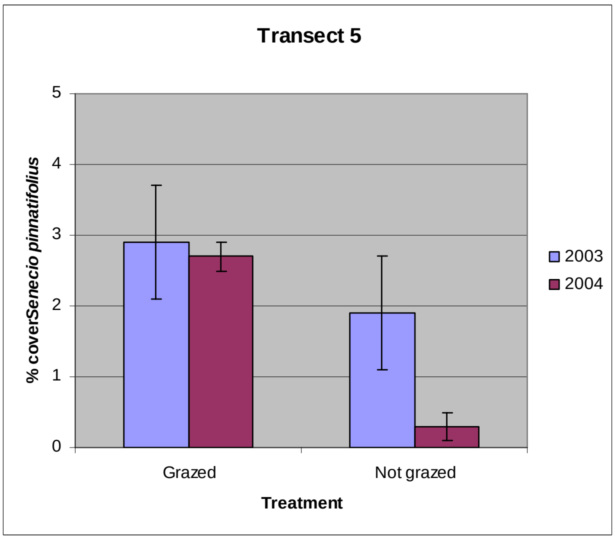
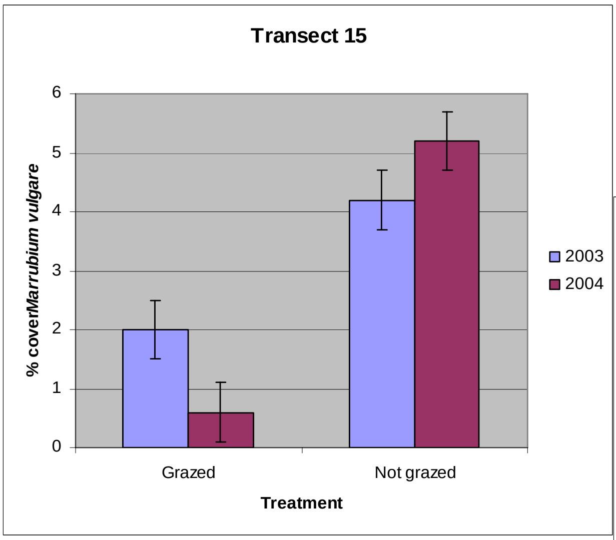


Figure 12. Mean percentage vegetation cover for each of the dominant species from the grazed and un-grazed sections of the transects pre-fencing (2003) and post fencing (2004) where a significant interaction was found between treatment and date.



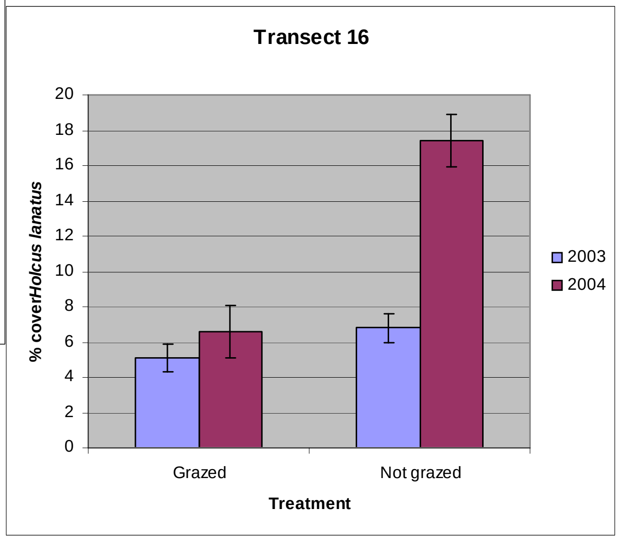


Figure 12. Continue. Mean percentage vegetation cover for each of the dominant species from the grazed and un-grazed sections of the transects pre-fencing (2003) and post fencing (2004) where a significant interaction was found between treatment and date.

4 DISCUSSION

4.1 Corangamite Water Skink population response to modified management

After two years of monitoring, no firm conclusions regarding the effects of grazing exclusion on the *E. t. marnieae* populations can be made. Given that *E. t. marnieae* take two years to reach adulthood, any significant effects of grazing exclusion or any modified management on the adult population due to changes in rates of reproduction, recruitment or survival would not be expected to manifest themselves in an increased density of adult skinks for a minimum of two years after implementation (Scroggie 2005). Such a response time was witnessed following the exposure of new rocky habitat at three *E. t. marnieae* sites in the late 1990's when the waters of Lake Corangamite and Lake Colac receded. In the two year's following the habitat exposure, no significant change in adult densities were observed, while an increase in juvenile and subadult recruitment was recorded. In the third year, however, this culminated in large increases in the adult populations (Peterson 2001).

In regard to the grazing exclusion, it may be expected that the delay between fencing and a response in adult numbers could actually be longer than 2-3 years, as it is likely that any response that may occur is due to changes in vegetation and other habitat features in the absence of grazing (Scroggie 2005). For instance, the recruitment and establishment of plant species beneficial to *E. t. marnieae* may take a number of years following the initial removal of livestock grazing. In contrast, the rapid increase in adult densities witnessed in the late 1990's was in response to the exposure of new rocky habitat; a beneficial change in habitat that was instantaneous once the water receded.

As sites are not homogenous, the removal of stock grazing may be more beneficial or recovery more rapid at one site over another. For example, removal of grazing from a site with scattered basalt outcrops interspersed by large areas of vegetation such as Site 1 may show greater or more rapid population recovery than a site of extensive pillow basalt flows with small areas of vegetation generally inaccessible to stock.

Whilst no conclusive response to grazing exclusion has been recorded, an increase in the number of adult skinks was witnessed across the entire transect at Site 1 in 2003/04 and 2004/05 followed the completion of a grazing lease at the site in August 2002. During the lease, stocking rates were well above appropriate levels, with the stock inflicting significant damage to the wetland vegetation throughout the skinks' habitat. Once the lease ended, the stocking rate was significantly reduced and the vegetation began to recover. The increase seen here 12 and 24 months after heavy grazing was removed may indicate that recovery is also influenced by the degree of grazing pressure and stocking rate. At sites where grazing has been light, the response of the population to grazing exclusion may take time to manifest in conjunction with the habitat recovery, while at sites where grazing pressure has been high the initial response to exclusion or even significant reduction may be rapid.

The immediate response to the removal of the high stocking rate in August 2002 may have been decreased mortality from trampling of the subadults and/or increased survivorship and recruitment of subadults over spring, coupled with the increased vegetation cover through the recovery of grasses and herbs. An increase in the number of subadult and juvenile skinks was observed at Transect 1 throughout the 2002/03 field season (Peterson unpublished data). The increased recruitment of these subadults may have then culminated with the increased number of adults observed in the following field season.

The distinct separation in the number of adult lizards observed in the grazed and un-grazed portions of Transect 16 in the second year following the fencing may suggest that the skinks have moved from the grazed to the un-grazed section of the transect. Although *E. t. marnieae* has been found to only move short distances (maximum 18m)(Knights 2003);(Malone and Peterson in prep.), it is plausible that these animals have dispersed into the un-grazed section of the transect, especially since a number of the additional animals recorded in the un-grazed section in 2004/05 were within 5-30m of the grazed section. Similarly, fewer animals were observed in the grazed section of the transect close to the un-grazed section in 2004/05 than in previous years. Analysis of capture-mark-recapture data taken prior to the grazing exclusion fencing and again in spring 2005 may help in determining this.

The fact that no decrease in adult skink numbers was observed once grazing was removed indicates that initially there was no detrimental effect to the adult population. One concern with the grazing exclusion was that an increase in biomass may reduce basking locations at the highly vegetated sites. An increase in biomass was also seen as potentially decreasing the visual detectability of skinks and thus, indicate over time an apparent population decline in the absence of any real change in absolute abundance (Scroggie 2005). To date, it appears that this has not eventuated. The capture-mark-recapture data, however, will help to determine and monitor this into the future.

Although the statistical assessment of the monitoring data prior to 2003/04 suggested that the monitoring strategy was sufficient to detect biologically significant population changes over a short period of time, it also stated that detection would depend on the expected population responses to the management effort, and on the timescale over which such responses were expected (Scroggie 2002). As only two years of post grazing exclusion data was available at the time of the assessment and as visual examination of the data showed that no significant divergence in abundance of adult skinks had occurred, no further statistical analysis of the data at this stage was warranted.

Data on juvenile and subadult *E. t. marnieae* were not assessed in regard to the impacts of grazing exclusion, as previous analysis demonstrated that generally the size of the sampling error associated with their measured abundances at this point would require an unrealistic sampling effort to assess change in their abundances with any precision (Scroggie pers. comm.).

4.2 Habitat response to modified management

Twelve months after the removal of grazing from the wetland/grassy shrubland habitat of *E. t. marnieae* at five sites, the only significant change in habitat cover was an increase in perennial grasses over 76cm in height, while an increase in the total perennial grass cover and percentage cover 11-25cm was evident but not significant. These results are not surprising considering the short time frame since grazing exclusion. An increase in the height of the palatable perennial grasses such as *Phalaris* would be expected once the pressure of stock grazing had ceased. Other shorter exotic perennials such as Yorkshire Fog and Kikuyu and natives such as Common Blown Grass and Wallaby Grass also increased in cover once grazing was removed (unpublished data). Any detectable response from other less palatable vegetation types such as woody shrubs would be expected to take much longer.

One major concern with the increase of weedy perennials upon grazing removal is the possibility of the grasses shading and/or growing over and covering the basalt rocks which are an integral part of *E. t. marnieae*'s habitat, especially in regard to basking and thermoregulation (Peterson 1997). Within the first 12 months no significant change in percentage exposed rock cover was observed, suggesting no reduction in basking site availability as the result of grazing exclusion. Continued monitoring of species in particular *Phalaris* and Kikuyu will be required, however, to ensure rock exposure is not compromised in the years following grazing exclusion. At some *E. t. marnieae* sites where stock exclusion fencing has previously been undertaken, increased cover of perennials and in particular *Phalaris* is evident 2-3 years after stock removal.

Associated with the increase in *Phalaris* cover is the potential for the creation of further fox harbour; an issue that has been highlighted by sheep farmers and conservationist alike. Greater fox numbers may increase predation pressure on *E. t. marnieae* as well as on livestock, in particular lambs. Efforts to reduce *Phalaris* cover (see below) and/or fox baiting may therefore be required.

As stated earlier, changes to other vegetation forms may take longer to manifest following grazing exclusion in this wetland/grassy shrubland habitat. Previous studies in grasslands and grassy woodlands have found that significant changes to vegetation cover, structure and recruitment can take years to occur, with rates of response varying across vegetation life-forms (Spooner *et al.* 2002; Lunt 2005). Recovery and recruitment from woody shrub species such as *Hymenanthera dentata,* which are damaged through trampling and breakage as well as grazing from stock, most likely would not be evident for a number of years following grazing. The percentage cover of annuals, of which most were exotics, may be expected to decrease following grazing exclusion as perennial cover increases or niches created by stock disturbance disappear. These changes would be expected to occur quicker, but it may still take a number of years before any change is evident (Spooner *et al.* 2002; Lunt 2005).

The degree of vegetation recovery will also vary according to the extent of degradation (Lunt 2005). Recovery following grazing exclusion will be slower and weaker in highly degraded areas dominated by exotics with few natives, than areas of low or mild disturbance (Lunt 2005). As sites were not homogenous in regards to habitat structure and species composition as well as in the extent of habitat degradation and disturbance, the rate of response from species is also expected to vary across sites. This is evident in the different trends from the percentage cover data from the individual transects and also the varying results from the 10 dominate plants species from each transect.

Establishment, recruitment and recolonisation of natives will develop over different time frames depending on life-form and response to grazing. Many grazing sensitive species (Poa tussocks and native sedges) were absent or in very low numbers at most sites. No significant recovery of any of these species was detected in the first year following grazing exclusion. As many of these species do not have persistent seed banks and have minimal chance for recolonisation due to isolation and fragmentation (Spooner *et al.* 2002), revegetation may be required to help restore these species.

With the species identified as being directly beneficial to *E. t. marnieae,* none were found to significantly increase following grazing exclusion within the first 12 months. *Hymenanthera dentata* may take a number of years to respond while species such as *Senecio pinnatifolius* and *Urtica incisa* which are generally not grazed but damaged extensively through trampling would be expected to recovery much quicker. *Carex appressa* which is grazed extensively also did not vary following grazing exclusion. However, the recovery of individual plants at two sites was encouraging. At site 5 the cover of *Senecio pinnatifolius* was found to decrease following grazing exclusion. This appeared to be in response to an increase in water level rather than a response to grazing exclusion, as *Senecio pinnatifolius* in this section of the transect was generally found at the base of the basalt rocks at the waters edge. During spring 2004 the lake level had encroached over the base of the rocks and was higher than in 2003. This was also evident in the increase percentage water cover recorded within the transect in 2004.

The decrease in ground level vegetation cover in the grazed section of the transects between 2003 and 2004 may have been due to an increase in grazing pressure following the stock exclusion from part of each of the transects. Although grazing was removed from part of each site, stocking rates were not altered. Currently, the decrease in overall vegetation cover appears to be mostly attributed to the decrease in perennial grass cover. The continued impact of this increased grazing pressure, however, will need to be monitored closely so any further degradation to the habitat of *E. t. marnieae* still being grazed can be detected and mitigated.

4.3 Previous management

Since 2003 an extensive effort has been undertaken to implement stock exclusion/regulation fencing at *E. t. marnieae* sites across the species range. With funding from the Australian Federal Government via the Corangamite and Glenelg Hopkins Catchment Management Authorities and EnviroFund, and in conjunction with groups such as Greening Australia's *Borrell-a-kandelop* project, Lismore Land Protection Group, Trust for Nature, West Lake Colac Action Group and Watershed 2000, approximately 22km of fencing has been erected to protect the habitat of *E. t. marnieae* across 16 sites. At all sites the fencing was undertaken with the full agreement, support and enthusiasm of, and at no or minimal costs to, the land owner/manager.

The fencing provided was generally to the landholder's specifications and would run linear to the water body 10m-15m back from the edge of the water itself. This width generally encompassed the entire width of the *E. t. marnieae* habitat.

Even though the specific response of *E. t. marnieae* and its habitat to grazing exclusion had yet to been defined back in 2003, it was considered by the Corangamite Water Skink

National Recovery Team to be appropriate to undertake the fencing program considering the funding available and potential benefits to *E. t. marnieae*, its habitat and the entire wetland community.

Along with protecting the terrestrial and semi-aquatic wetland vegetation of *E. t. marnieae*'s habitat, the stock exclusion fencing would also restrict stock access to the wetland itself. This would reduce the potential for pugging of the wetland, grazing of the aquatic vegetation, reduce turbidity, and reduce nutrification from defecation and thus improve water quality. Exclusion from the wetland would also reduce the chance of stock getting stuck in the mud especially during summer when the water level recedes. If the fencing of a wetland used as a watering point for stock was desired, funding to establish alternative watering points away from the wetland were available.

At all sites, the area occupied by *E. t. marnieae* does not exceed 2% of the landowner's property and in most cases is even less. Combined with the fact that the skink and its rocky habitat is confined to the verges of wetlands, it lent itself to being an ideal species for landholders to retain and preserve on their properties with no net loss in agricultural productivity. Engendering a sense of ownership of this animal on the landholder also helped in encouraging them to conserve the skink and its habitat.

Along with fencing, other management options that have been developed and implemented for *E. t. marnieae* and its habitat include enhancement of rock habitat, weed control, predator control, recreational regulation and water management (Robertson and Peterson 2000).

4.4 Future directions and management options

The results of the short term response of *E. t. marnieae* to grazing exclusion in this study were generally inconclusive. Continued population monitoring is required to ascertain how the populations respond to grazing exclusion long term, as transitions and changes within the habitat and vegetation community takes place. Already additional monitoring has been undertaken over the 2005/06 field season and with this additional data Dr Michael Scroggie will undertake a formal statistical analysis in late 2006. This will be coupled with an assessment of capture-mark-recapture data taken from three of the grazing exclusion sites for 5 years prior to grazing exclusion and two years post exclusion. Currently the population monitoring will continue until at least 2008.

Regarding vegetation response, further long term assessment is planned in spring 2006 and spring 2008. This will be undertaken to track the continued response and successional change within the vegetation community following grazing exclusion, as well as allow potential threats to be identified and appropriate alternative management to be implemented. Continued monitoring will also be conducted at all sites where fencing to exclude stock has been undertaken and if required, alternative management will be implemented.

Alternative or complementary management options could include crash grazing at sites to reduce the cover of perennials such as *Phalaris* or Kikuyu. To date appropriate timing for crash grazing in these habitats is yet to be defined. Obviously spring is not appropriate due to most species flowering and setting seed. Mid summer has been suggested for grassy ecosystems (Dorrough *et al.* 2004). However, this is the period just after *E. t. marnieae* give birth and when neonates are occupying the heavily vegetated microhabitats. Weed control through spraying is another option, especially at sites where crash grazing cannot be conducted or where woody weeds in particular Boxthorn is present. The potential impacts of herbicide sprays on *E. t. marnieae,* however, are currently unknown and any application needs to be timed when *E. t. marnieae* activity is low.

As grazing exclusion alone is unlikely to restore the original vegetation composition of *E. t. marnieae*'s habitat, revegetation with native species could be undertaken. The planting of species that have been identified as beneficial to *E. t. marnieae* such as *Hymenanthera dentata* and *Carex appressa,* as well as other species that were part of the original vegetation community will help to enhance the entire ecosystem. Generally, only native grasses, sedges and shrubs should be planted at *E. t. marnieae* sites. Tree planting is discouraged, as the trees may shade the rocky habitat once mature and reduce moisture levels within the skinks rocky crevice refuges. If both revegetation and crash grazing is required, smaller fenced plots of approximately 2m x 2m can be established within the fenced area and planted with appropriate natives.

5 CONCLUSIONS

This study has demonstrated that long term monitoring is essential to accurately assess the response of *Eulamprus tympanum marnieae* and its habitat to the exclusion of livestock grazing. Although the fencing did not lead to any obvious short term enhancement of *E. t. marnieae* populations or its habitat, it may still facilitate longer term benefits and also prevent long term habitat degradation.

Any significant change in adult abundances in response to grazing exclusion may take numerous years to detect, with the rate of recovery potentially linked to the response and rate of change of the vegetation and other habitat features in the absence of grazing. The change and recovery of some vegetation life-forms, however, can be rapid as seen in this study with perennial grasses, while response from other vegetation life-forms may take longer to manifest. Factors that are likely to influence rate of change and recovery of the vegetation include degree of disturbance and degradation, species composition, habitat structure, and seed bank persistence.

The recovery and spread of undesirables such as *Phalaris* in the absence of grazing can negate or impede the benefits of the stock exclusion and hence will require monitoring and possible mitigating. Supplementary plantings of indigenous natives may also be required at some sites to help re-establish the original vegetation composition.

Management options, including those suggested and discussed above, are to be refined and developed further with DPI Agriculture staff, land-holders and other key stakeholders in the final phase of the project. As *E. t. marnieae* and its rocky habitat is localised along the verges of wetlands, threat abatement can be achieved with no net loss in agricultural productivity. Fencing of *E. t. marnieae* habitat to exclude livestock appears the easiest and most feasible method of threat mitigation. However, long term monitoring will be required, as may additional management actions. The process of conserving *E. t. marnieae* is, therefore more than just a case of 'fence and forget'.

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APPENDIX I

Table 1. Pooled mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of the five transects in spring 2003 and 2004.

Table 1. Continue. Pooled mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of the five transects in spring 2003 and 2004.

Table 2. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 1 in spring 2003 and 2004.

Table 2. Continue. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 1 in spring 2003 and 2004.

Table 3. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 5 in spring 2003 and 2004.

Table 3. Continue. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 5 in spring 2003 and 2004.

Table 4. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 11 in spring 2003 and 2004.

Table 4. Continue. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 11 in spring 2003 and 2004.

Table 5. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 15 in spring 2003 and 2004.

Table 5. Continue. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 15 in spring 2003 and 2004.

Table 6. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 16 in spring 2003 and 2004.

Table 6. Continue. Mean percentage cover and standard errors of vegetation and habitat attributes in grazed and un-grazed sections of Transect 16 in spring 2003 and 2004.

Table 7. Mean percentage cover and standard errors of the 10 dominant species in grazed and un-grazed sections of Transect 1 in spring 2003 and 2004. # indicates an indigenous native species. Other species are introduced.

Table 8. Mean percentage cover and standard errors of the 10 dominant species in grazed and un-grazed sections of Transect 5 in spring 2003 and 2004. # indicates an indigenous native species. Other species are introduced.

Table 9. Mean percentage cover and standard errors of the 10 dominant species in grazed and un-grazed sections of Transect 11 in spring 2003 and 2004. # indicates an indigenous native species. Other species are introduced.

Table 10. Mean percentage cover and standard errors of the 10 dominant species in grazed and un-grazed sections of Transect 15 in spring 2003 and 2004. # indicates an indigenous native species. Other species are introduced.

Table 11. Mean percentage cover and standard errors of the 10 dominant species in grazed and un-grazed sections of Transect 16 in spring 2003 and 2004. # indicates an indigenous native species. Other species are introduced.

APPENDIX II

An assessment of monitoring requirements for the Corangamite Water Skink, *Eulamprus tympanum marnieae*.

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Background

Monitoring data collected from 13 populations of the Corangamite Water Skink, *Eulamprus tympanum marnieae* between 1997 and 2002 was analysed with a view to making recommendations for a robust monitoring program for these populations. The data consist of multiple visual line-transect counts at each site. Not all sites have been sampled for the entire study period. The frequency and timing of sampling has also varied somewhat between sites.

The intention of this study is to determine the size and distribution of the sampling error associated with the measure of abundance used. Using this knowledge it is possible to infer the amount of sampling necessary in order to assess abundance with a given degree of precision, and to detect changes in abundance of a specified magnitude. The measure of abundance used in this study is the number of adult skinks sighted during a visual line-transect count. This index cannot be considered to approximate absolute abundance, as an unknown proportion of individuals in the population may not be sighted during a particular transect count. Nevertheless, it is reasonable to assume that the numbers seen during a transect count would be indicative of absolute abundance, and hence of value as an index of absolute abundance for monitoring purposes.

Methods

Ideally, in order to assess the sampling error associated with transect counts, repeated counts should be made at the same locations, during a relatively short period of time, during which it is reasonable to assume that the absolute abundance of skinks has not changed due to mortality, recruitment or migration. The observed variability of repeated counts could then be used to estimate the size and distribution of sampling errors associated with the line-transect method of assessing abundance. In the present case, the data were collected over a long period of time, during which it is not reasonable to assume that the absolute abundance of skinks has remained constant. Indeed, it must be assumed that considerable changes in absolute abundance may have occurred over the course of study at at least some of the sites.

In order to estimate the sampling error of the line-transect method, it is necessary to simultaneously assess any temporal changes in abundance that may have occurred over the course of the sampling program. This aim can be accomplished by fitting a regression equation to the observed relationship between time and relative abundance, and considering the distribution of residuals of this regression to approximate the distribution of sampling errors associated with the abundance estimator. This approach may tend to overestimate the sampling error associated with the line-transect sampling method, as a proportion of the total error associated with the regression line will be a consequence of real deviations in absolute abundance from the fitted regression equation. (i.e. the total error includes both process error and sampling error, Hilborn and Mangel 1997). Nevertheless, this approach will at least provide an upper limit for the sampling error of the line-transect monitoring method, and gives a framework for providing recommendations on sampling methodologies. Due to strong seasonality in the occurrence of juvenile skinks on the transects, only counts of adult skinks were included in the analyses.

The regression equations fitted to the relative abundance time-series data were exponential population growth equations with the following formulae:

Where N_t is the observed abundance at time t, N_0 is the abundance at time zero (taken as January 1, 1997), r is the population's intrinsic rate of increase, t is time in years, and e is Euler's number. The parsimony exponential regression models for each site as compared to a null model of constant abundance through time was assessed using the corrected Akaike's Information Criterion (AICc, Burnham and Anderson, 1998). The relative support for each of these two alternative models was assessed based on their relative Akaike weights. Where support for one model over the other were equivocal, model averaging of regression coefficients was conducted based on their relative Akaike weights, to account for the uncertainty in model-selection (see Burnham and Anderson, 1998, for a description of these methods).

The regression equations were fitted to the time-abundance data from each site using a generalized linear model with negative binomial error-structure, and logarithmic link-function. The negative binomial distribution is considered to be an appropriate model for the sampling errors of wildlife count data, as it is constrained to take only positive, integer values, and can have any positive mean and variance. In the cases of sites 4, 6, and 11 there was no evidence of over-dispersion, therefore, a poisson model with logarithmic link was considered more parsimonious. The variances of the poisson and negative binomial distributions with mean value μ are given by the following equations (Venables & Ripley, 2002):

(poisson)

(negative binomial)

Therefore, the negative binomial distribution converges to a poisson distribution as $k \rightarrow \infty$.

Results

The regression coefficients N_0 and r were estimated from the data simultaneously with the dispersion parameter of the negative binomial error distribution, k, (where appropriate). The resulting fitted values of are presented in Table 1. Where the Akaike weights did not provide unequivocal support for one of the two alternative models over the other model-averaged coefficient estimates are reported, and were used for all later analyses.

Table 1. Estimates of regression model coefficients, and relevant model details for each of the populations.

Site	Error structure	Model used	Akaike weight (exponential	Akaike weight (constant	N_0	r	k

	1		model)	model)			
1	NB	Exponential	1	0	27.854	0.096	8.029
2	NB	Exponential	1	0	2.519	-0.614	1.206
3	NB	Exponential	1	0	12.528	-0.305	12.743
4	P	Exponential	1	0	21.595	-0.340	-
5	NB	Exponential	1	0	46.065	0.118	21.418
6	P	Exponential	0.999	0.001	11.702	0.272	-
7	NB	Exponential	1	0	31.579	0.165	15.800
8	NB	Exponential	1	0	32.187	-0.017	14.811
11	P	Averaged[2]	0.072	0.928	17.291	0.0131	-
12	NB	Exponential	1	0	4.005	0.338	20.715
14	NB	Exponential	0.999	0.001	43.864	-0.070	5.954
15	NB	Averaged[2]	0.783	0.217	10.375	0.121	2.807
16	NB	Exponential	0.986	0.014	28.841	-0.021	12.011

1. NB = negative binomial, P = poisson.
2. Regression coefficients (N_0, r) and dispersion factor k estimated by weighted averaging of estimates from the exponential and constant models, with weighting based on the relative Akaike weights for each model.

Requirements for future monitoring

The implications of various sampling intensities for future monitoring of the Corangamite Water Skink were assessed using Monte Carlo simulation. The process of line-transect sampling at each site was simulated by drawing random variates from a negative binomial distribution with mean equal to the estimated abundance at the site at the time of the last survey, and dispersion parameter k, determined from the previous monitoring work at each site (values in Table 1). For sites where the errors in transect counts had previously been modelled as poisson distributed, the poisson distribution was used for simulation instead of the negative binomial.

Where multiple transects are conducted during a single season, multiple independent random variates were drawn from the same sampling distribution, and the mean of these variates was taken as the relative abundance estimate for the season. The adequacy of undertaking various numbers of replicate surveys each year at each site was assessed by simulating the sampling program a large number of times (10,000). For the populations where the estimate of the rate-of-increase r was less than zero, a more conservative approach was adopted. Simulation was based on the same dispersion parameter value, but the mean of the relevant sampling distribution was reduced by 20% to allow for any ongoing population decline prior to future data collection, and to provide a margin for error.

A particular sampling program could be considered adequate, if the confidence interval for the estimates of abundance were smaller than a specified level of change in abundance which was considered to be biologically significant. In this case I somewhat arbitrarily chose a decline in abundance of 50% as a reasonable threshold decline in abundance which it would be desirable to detect. Whilst this may seem to be an unreasonably large change in abundance to be acceptable to detect, it should be noted that this reflects a change in abundance of 50% over a single year, and the certainty of detecting changes in abundance over longer periods of time will be considerably greater.

The sampling intensities considered for each site were between one and seven transect counts per year. The 95% confidence intervals for the abundance estimate (i.e., the mean count of transects within a season) are presented in Table 2. On this (admittedly arbitary) basis, the minimum number of annual transect counts required at each sites was determined. These results are presented in Table 3. For two sites (sites 2 and 3), no reasonable levels of effort would provide adequate sampling to detect a 50% decline in relative abundance: for these sites the required effort was given as “>7”. The required sampling regime at these sites would depend on the management purposes for which monitoring data was required. It is already apparent that these populations are small and rapidly declining: further monitoring is not necessary to reinforce these facts. However, if management actions were taken at these sites with the intention of improving survival or recruitment, monitoring programs would be necessary to assess their efficacy. Design of an appropriate management program would depend on the expected responses to the management efforts on relative abundance, and the timescale over which such responses were expected.

The total required annual effort (assuming an effort of 7 transects at sites two and three), was 41 transects per year. Assuming that on average 3 transects could be carried out on an average working day, approximately 14 person/days would be necessary to carry out the recommended sampling program. If additional time were available, additional transect counts should be allocated to the populations which from the present analysis have been found to be small and/or declining, and where the present estimates of r are least precise (Figure 1).

Table 2. Ninety-five percent confidence intervals of the abundance estimates at each of the sites under various sampling intensities. Intervals in boldface are the miminum sampling efforts whose 95% confidence intervals do not contain 50% of the present estimated abundance.

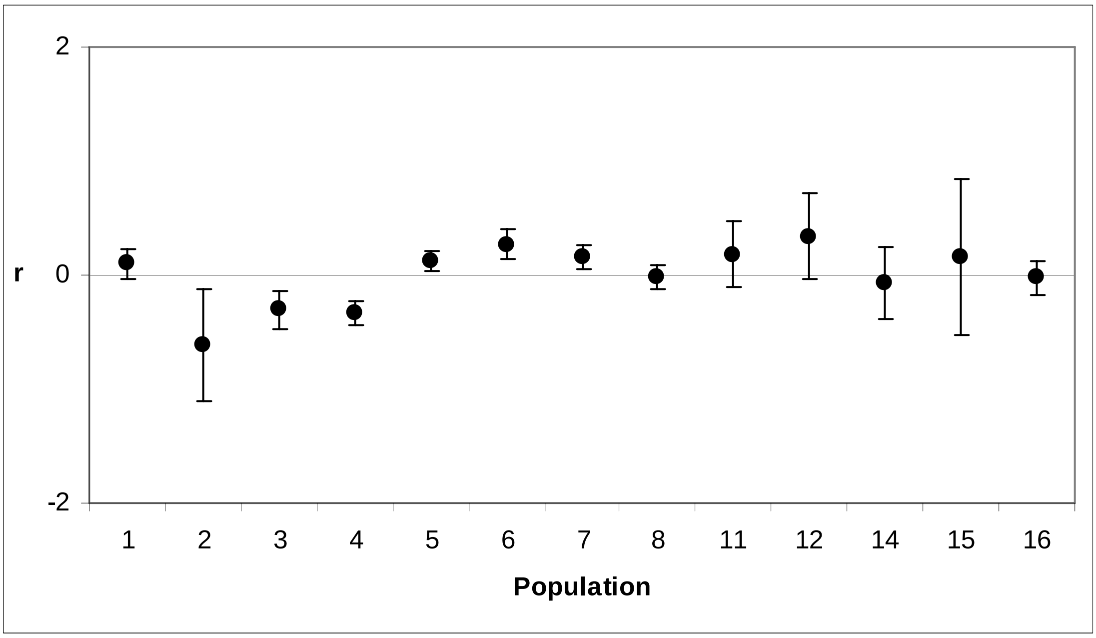
Site	N_{last}	$N_{caution}$	*k*	One	Two	Three	Four	Five	Six	Seven
1	46.594		8.029	18, 88	**25.0, 73.5**	28.33, 68.67	30.75, 65	32, 63	33.33, 62.00	34.28, 60.86
2		0.078	1.206	0, 1	0, 0.5	0, 0.67	0, 0.5	0, 0.4	0, 0.33	0, 0.29
3		1.961	12.743	0, 7	0, 5	0.33, 4.33	0.50, 4.25	0.6, 3.8	0.67, 3.67	0.71, 3.57
4		2.426	poisson	0, 6	0.5, 5.0	1, 4.33	1, 4	1.2, 3.8	**1.33, 3.67**	1.29, 3.71
5	86.287		21.418	**50, 131**	60.5, 117.0	63.33, 111.33	67, 108.01	68.8 105.0	70.0 103.5	71.57, 102.43
6	44.837		poisson	**33, 58**	36, 54	37.67, 52.67	38.50 51.25	39.0 50.8	39.5, 50.17	39.86, 49.86
7	74.934		15.800	**40, 121**	48.49, 106	53.33, 100	55.25, 96.75	57.8, 94.4	59.17, 93	60.14, 90.57
8		23.573	14.811	10, 41	13.5, 35	**15.33, 33**	16.25, 32	17.2, 31	17.67, 30.17	18, 29.71
11	18.449		poisson	**11, 27**	12.5, 24.5	13.67, 23.33	14.50, 22.75	14.8, 22.2	15.17, 22	15.29, 21.71
12	21.320		20.715	10, 36	**13, 31**	14.33, 29.33	15.25, 28	15.8, 27.4	16.5, 26.83	16.71, 26.28
14		23.567	5.954	7, 49	11, 40.5	**12.67, 36.67**	14, 35.25	15.2, 33.4	15.5, 32.67	16.29, 32.43
15	18.552		2.807	3, 48	5.5, 38	7.67, 33.67	8.75, 31.75	**9.4, 30.6**	10.17, 29.34	10.71, 28.57
16		20.403	12.011	8, 37	**11, 32**	12.67, 29.33	13.75, 28.00	14.4, 27.2	14.83, 26.67	15.29, 26.29

Table 3.

Minimum required annual sampling effort at each site

Site	transects
1	2
2	>7
3	>7
4	6
5	1
6	1
7	1
8	3
11	1
12	2
14	3
15	5
16	2

Figure 2. Estimates of *r* for each of the populations and their 95% confidence intervals.



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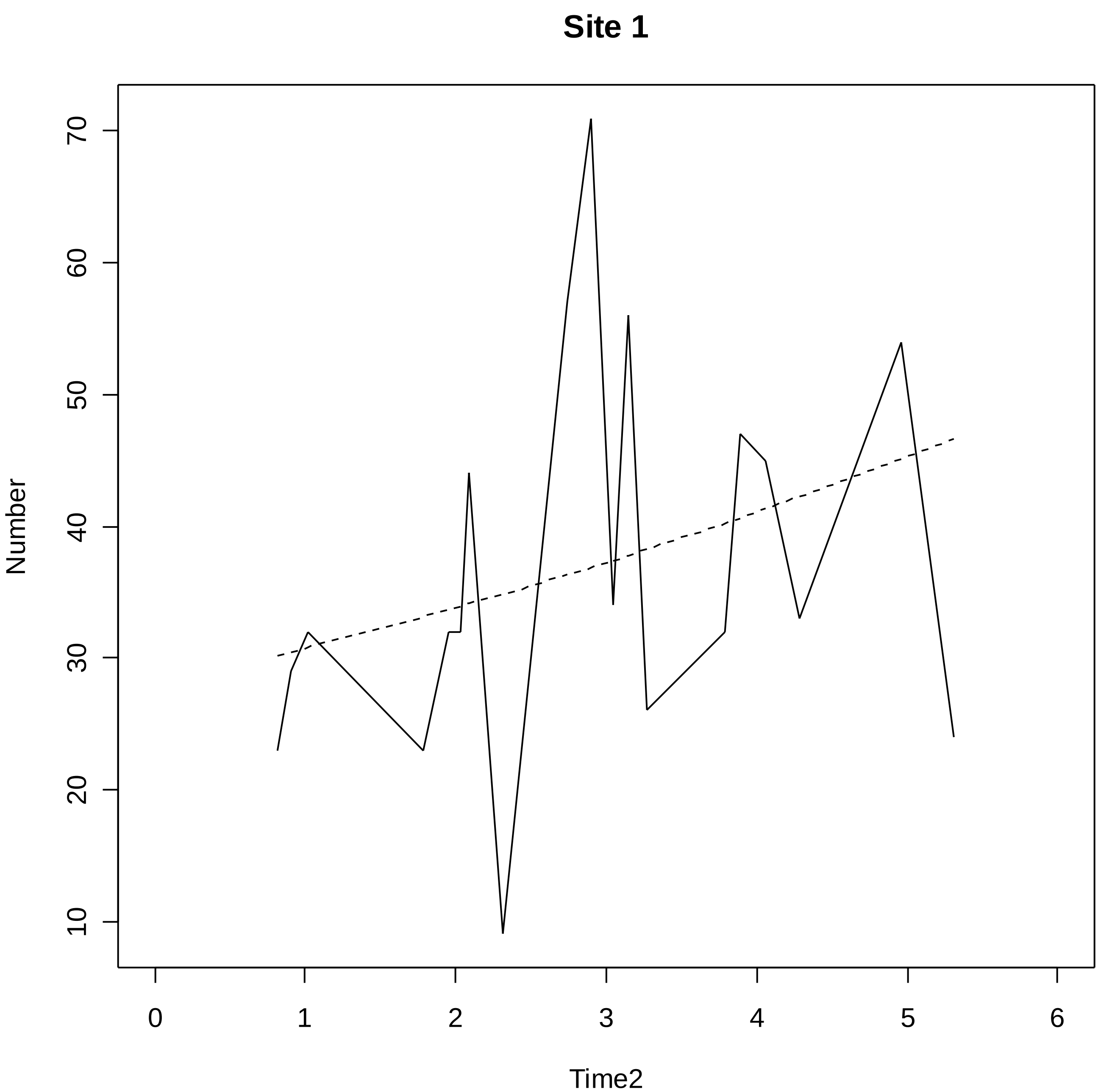
Site One.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.327	0.096	8.02861	-6282.089	1.000	
Null	3.61092	-	7.236358	-6120.818	0.000	0.198

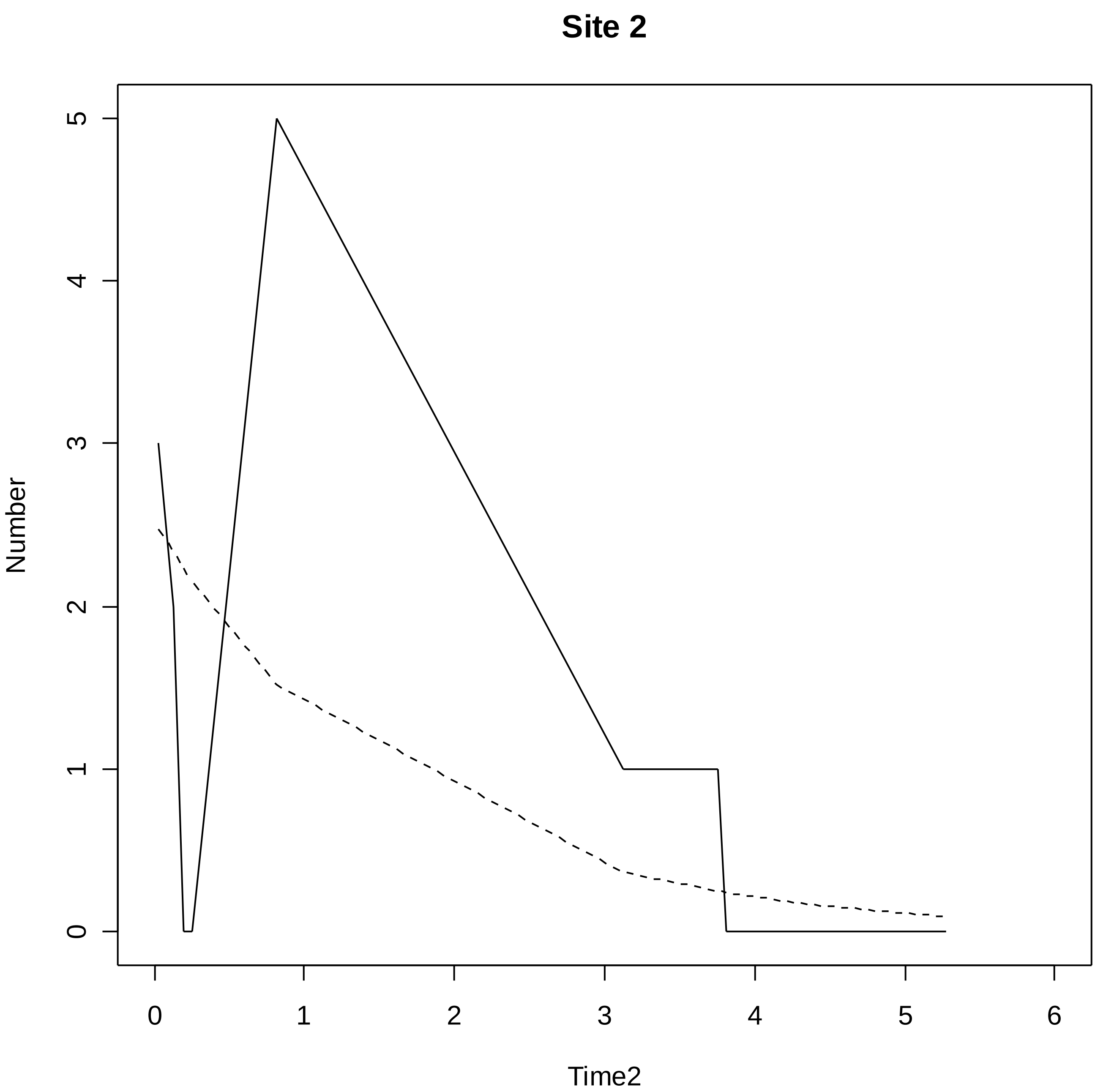


Site Two.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	0.924	-0.614	1.206	-5.96	1.000	
Null	-0.154	-	0.396	47.21	0.000	0.018



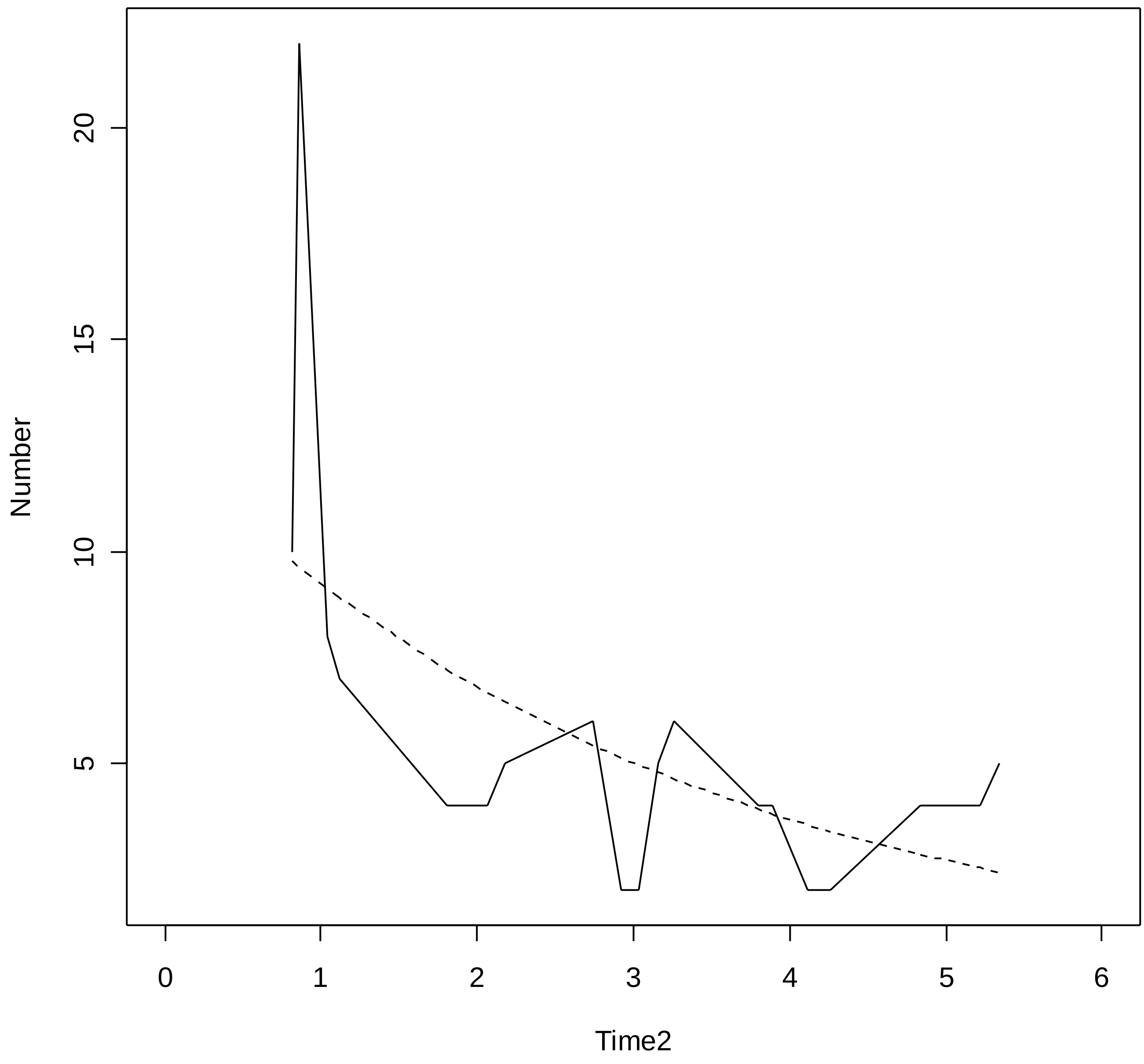
Site Three.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	2.52795	-0.30539	12.74386	-1928.1036	1.000	
Null	1.7190	-	17.69083	-704.4095	0.000	0.001

Site 3

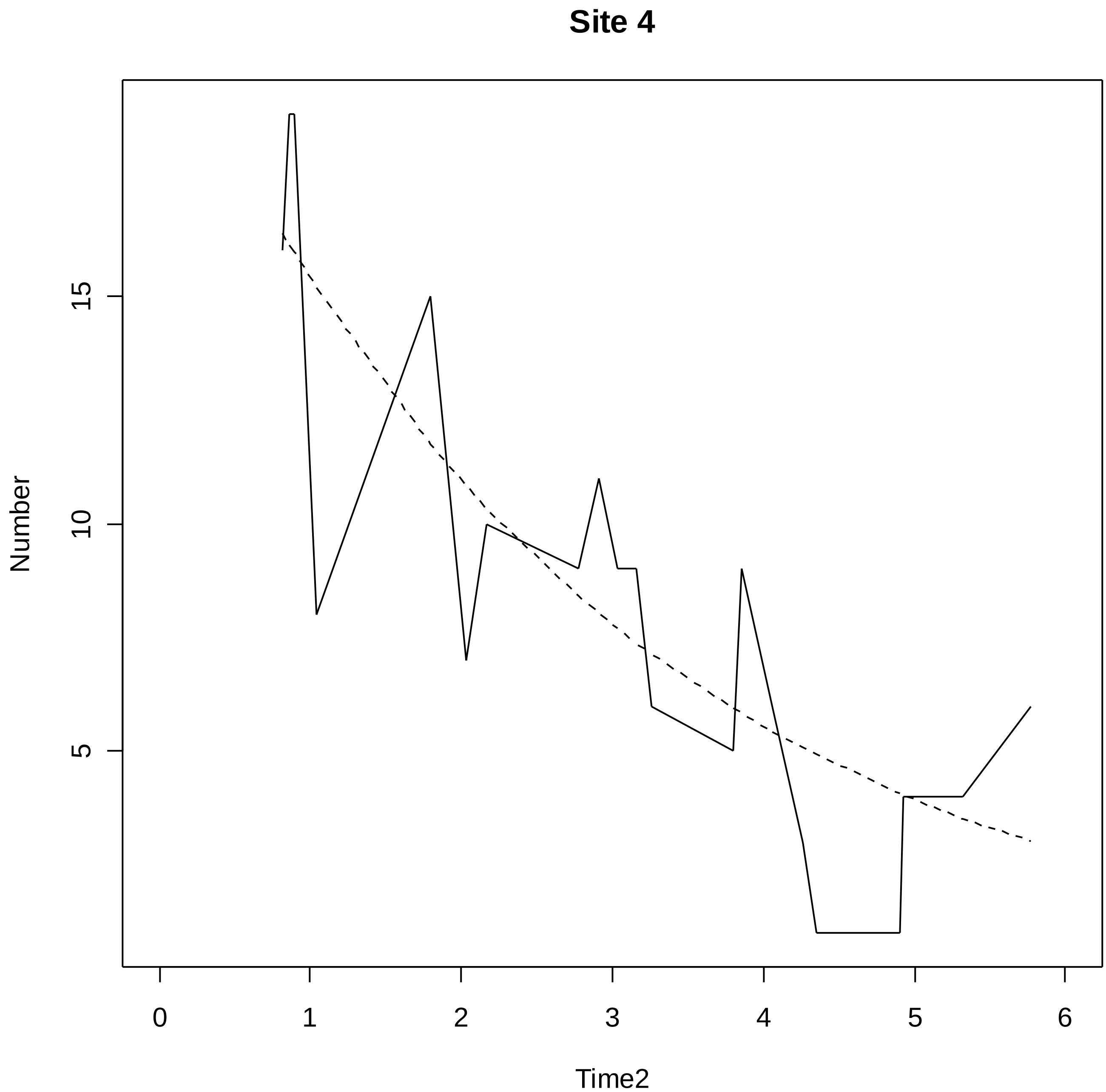


Site Four.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Data at this site not overdispersed, so poisson glm's fitted to models, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.07247	-0.33999	-	106.1010	1.000	
Null	2.13163	-	-	147.3501	0.000	<0.0001

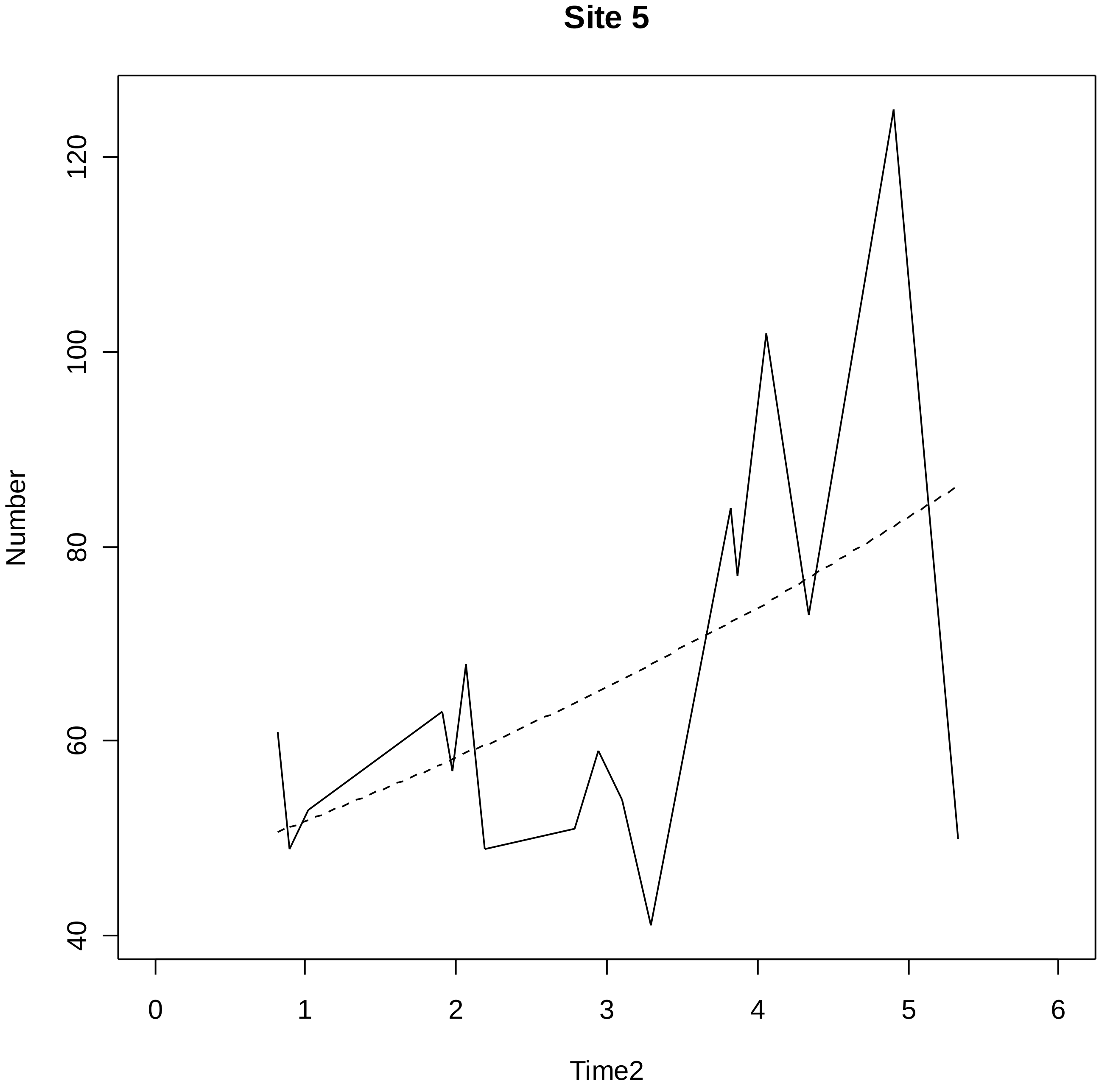


Site Five.

Null model $N_t=N_0$ (constant abundance)
Full model $N_t=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.83005	0.11774	21.41780	-13039.50	1.000	
Null	4.18429	-	14.13638	-11644.78	0.000	0.0158

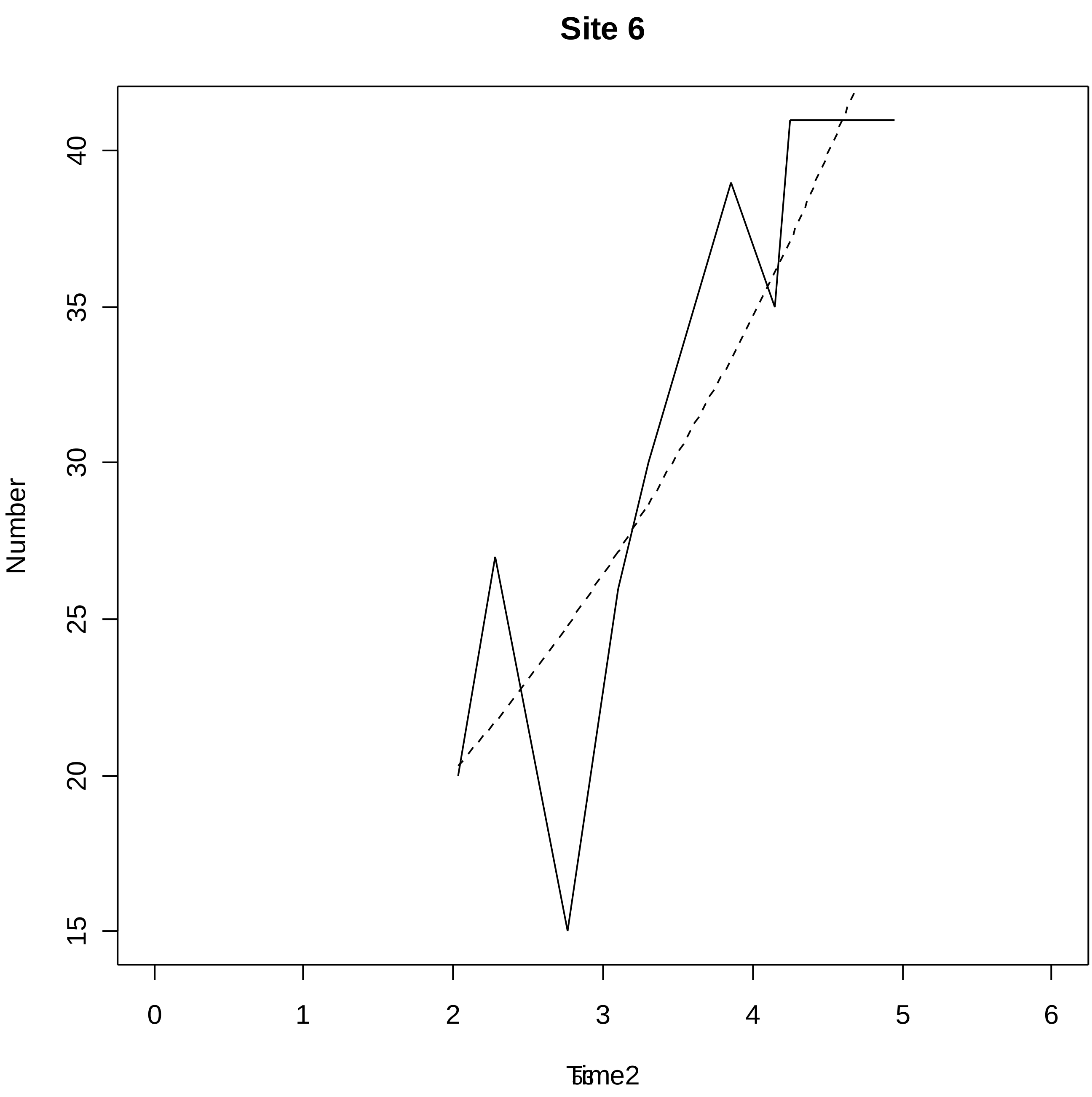


Site Six.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Data at this site not overdispersed, so poisson glm's fitted to models, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	2.45978	0.27151	-	60.41598	0.998731 791	
Null	3.41590	-	-	73.75374	0.001268 209	4.228e-05

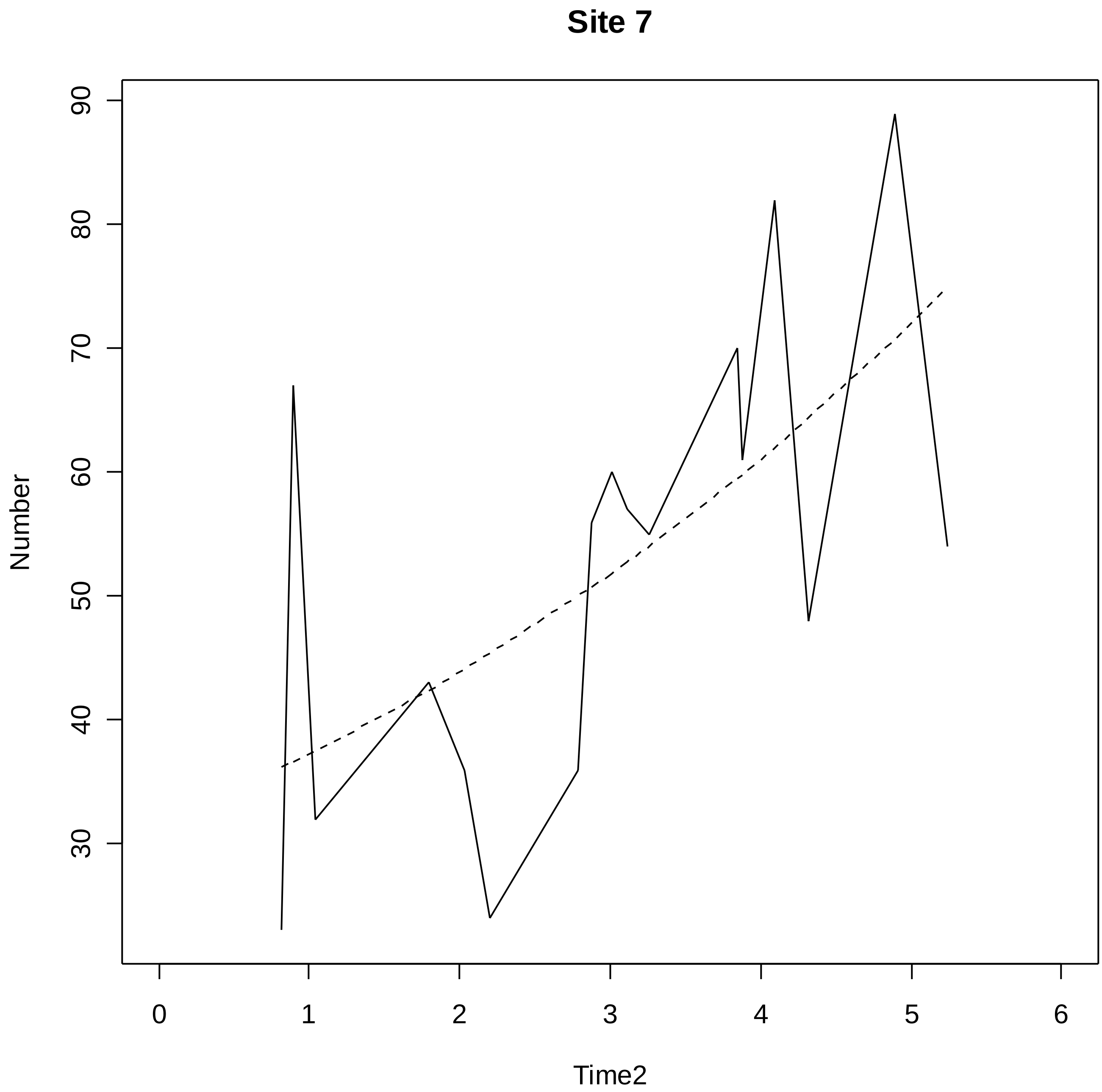


Site Seven.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.45248	0.16482	15.79989	-9648.34	1.000	
Null	3.96137	-	9.06894	-8387.42	0.000	0.006

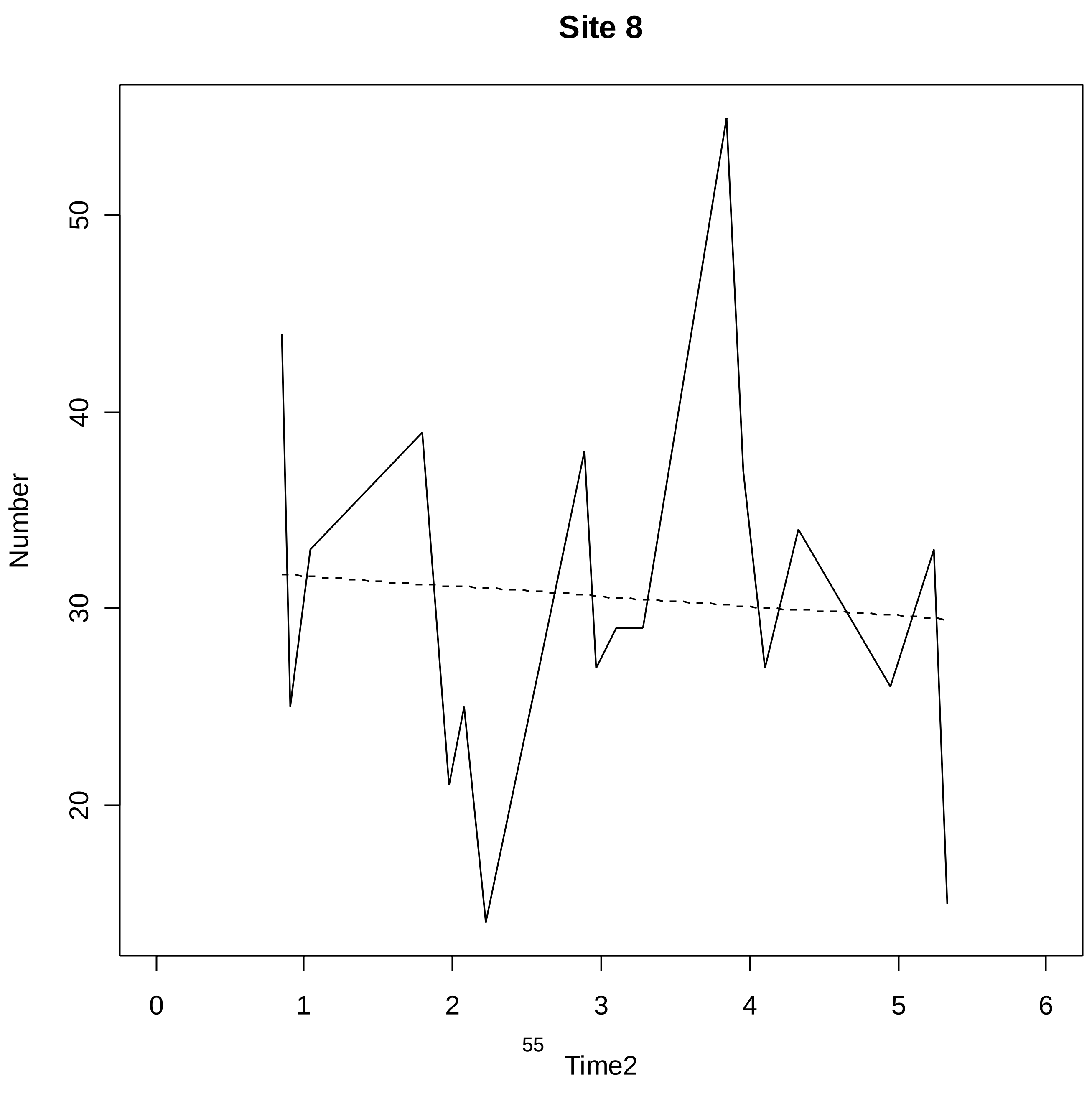


Site Eight.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.47155	-0.01656	14.81141	-6066.849	1.000	
Null	3.4214	-	14.69474	-6048.589	0.000	0.756886 1

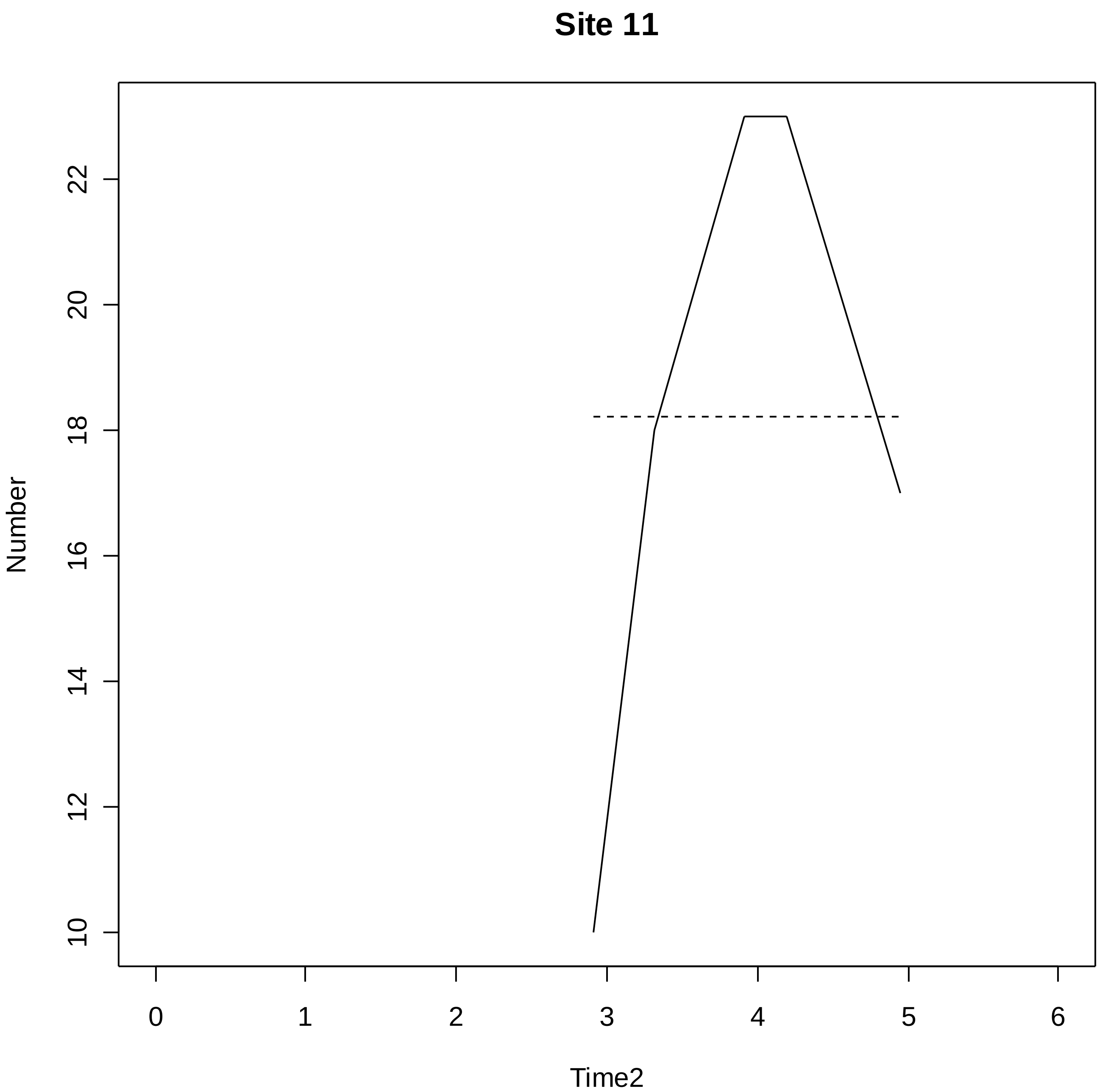


Site Eleven.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Data at this site not overdispersed, so poisson glm's fitted to models, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.
In this case constant abundance is clearly the better model, so prediction was based on this model (model averageing would also be possible: see model-averaged coefficients below).

Model	Intercept	r	k	AICc	weight	LR test
Full	2.1875	0.1830	-	38.83074	0.0717719	
Null	2.9014	-	-	33.71117	0.9282281	0.2136
Averaged	2.8502	0.0131	-			

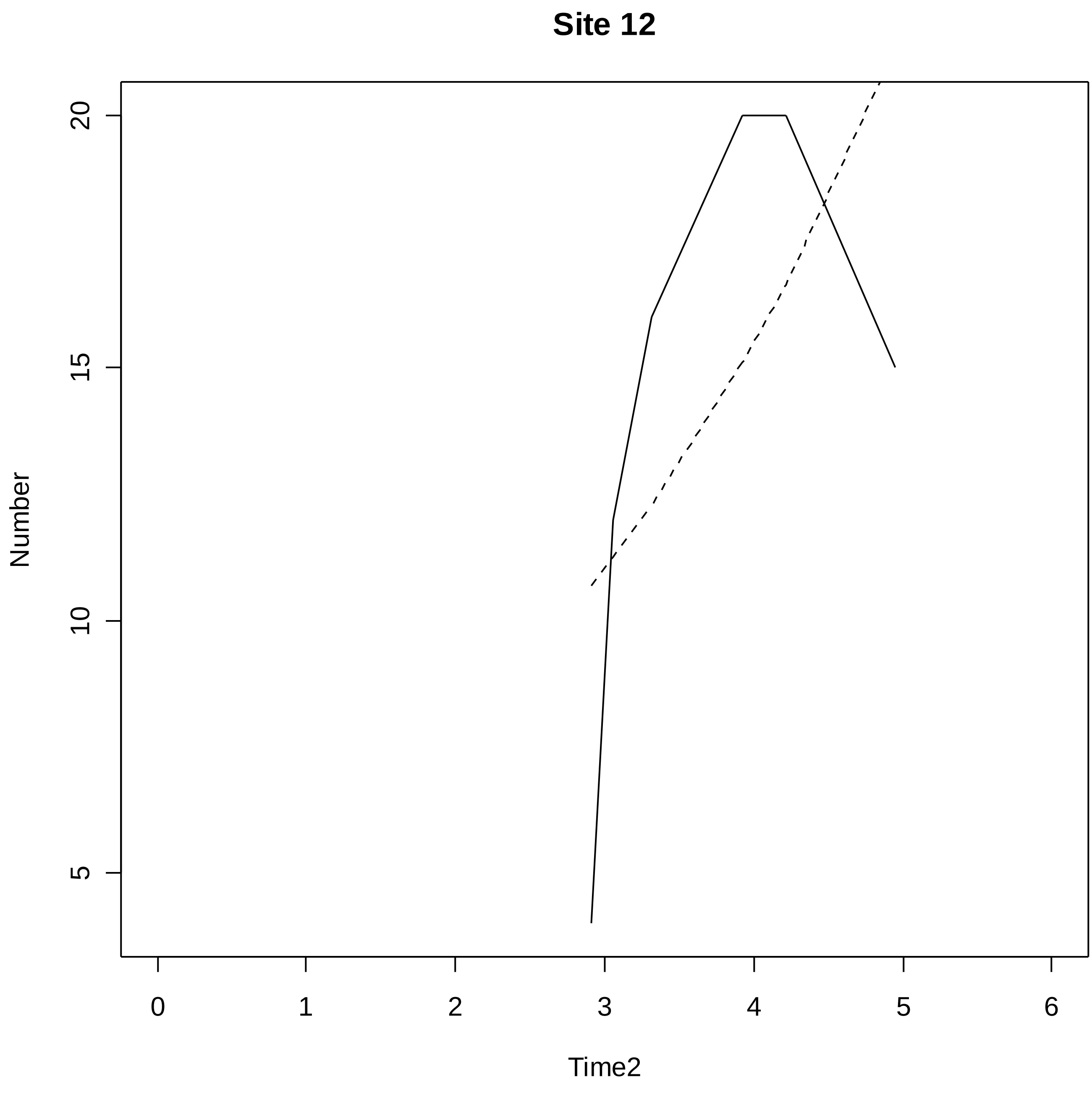


Site Twelve.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	1.3876	0.3380	20.71513	-1459.1917	1.000	
Null	2.6741	-	8.977802	-839.0218	0.000	0.1303807

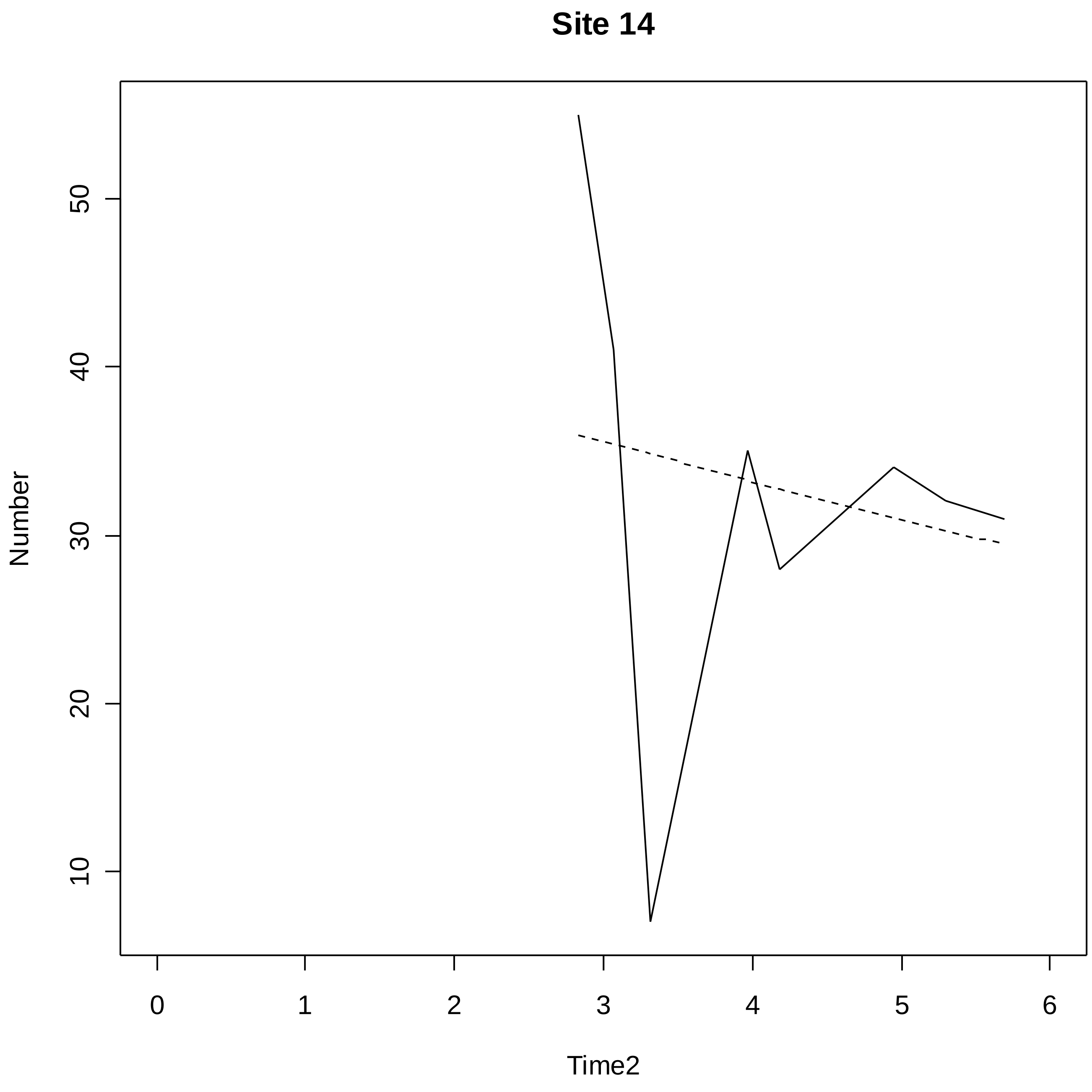


Site Fourteen.

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.78109	-0.06991	5.95431	-2167.763	0.999	
Null	3.4927	-	5.753931	-2153.782	0.001	0.6472123

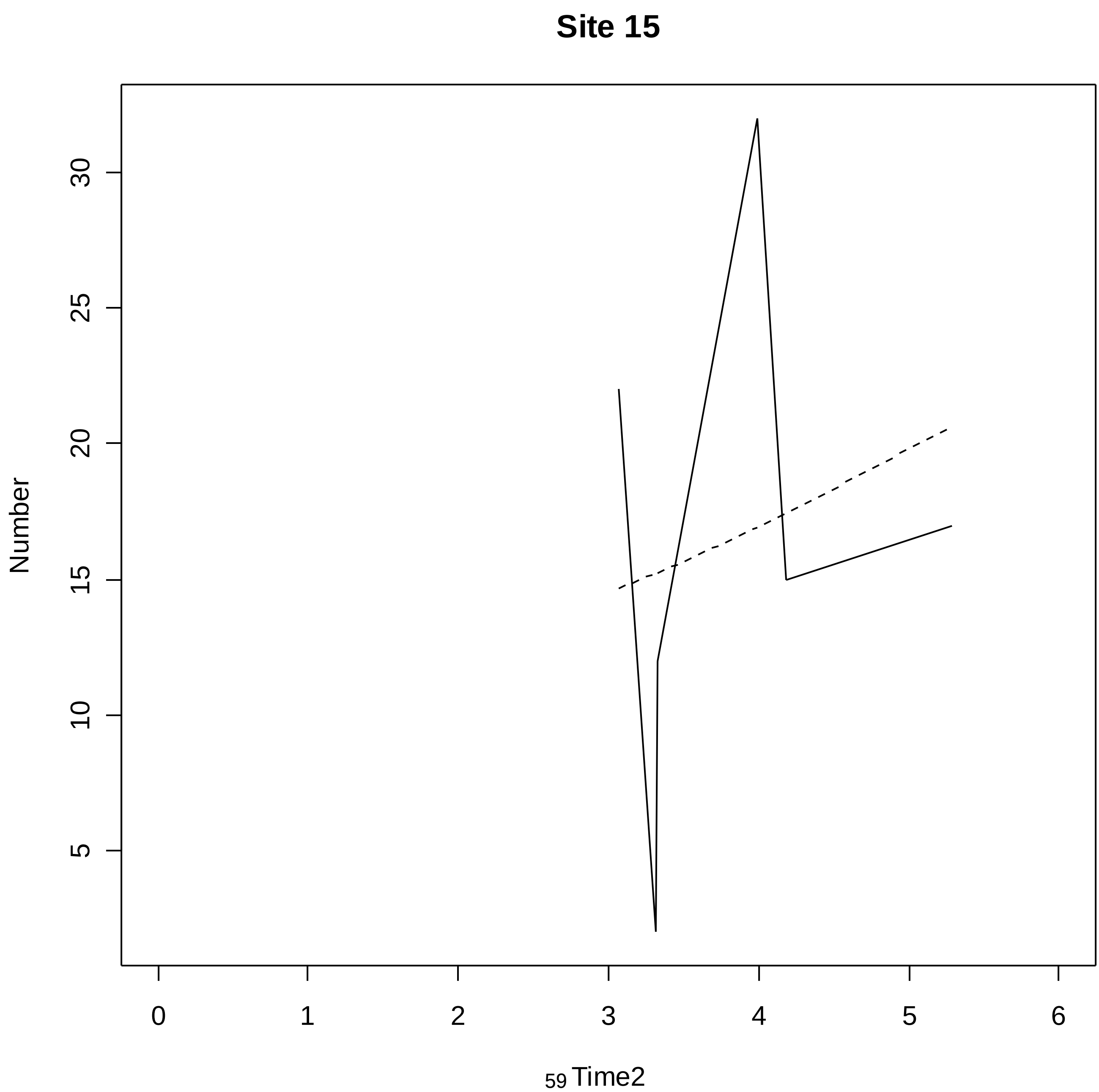


Site Fifteen

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests. Prediction based on full model, although model averaging would also be an appropriate means of prediction.

Model	Intercept	r	k	AICc	weight	LR test
Full	2.2080	0.1551	2.830210	-616.8375	0.7830018	
Null	2.8134	-	2.725539	-614.2710	0.2169982	0.671474
Averaged	2.3393694	0.1214663	2.807497			



Site Sixteen

Null model $Nt=N_0$ (constant abundance)
Full model $Nt=N_0e^{rt}$

Negative binomial glm's fitted to models using glm.nb, most parsimonious model chosen using AICc, models compared using likelihood ratio tests.

Model	Intercept	r	k	AICc	weight	LR test
Full	3.36180	-0.02132	12.01117	-4391.490	0.986	
Null	3.28887	-	11.93895	-4382.919	0.014	0.794