

# **Lantana management and its impacts on reptile assemblages and habitat quality within a wet-sclerophyll forest in south-east Queensland**

By

**Diana Angelique Virkki, B.Sc.**

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## **Statement of Originality**

The material presented in this thesis has not been previously submitted for a degree or diploma in any university, and to the best of my knowledge contains no material previously published or written by another person except where due acknowledgment is made in the thesis itself.

Signed: Diana A. Virkki

Date: 13th January 2009

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## Abstract

The impact of weed invasions can be considered one of the world's most significant environmental issues. Weeds can significantly affect biodiversity through suppression and alteration of natural environments, and can alter the functioning of natural ecosystems by competing with native flora and suppression or enhancement of fauna. The subsequent management of weeds may also have similar impacts on wildlife, however when undertaking such practices, the impacts on fauna are rarely considered. Reptiles are likely to be affected due to their reliance on microhabitat structure for foraging, basking and refuges. This study investigated the effects of *Lantana camara* on reptile assemblages and two integrated approaches to manage and control lantana by (i) herbicide spraying and manually clearing and (ii) herbicide spraying followed by prescribed burning in a densely infested wet-sclerophyll forest, located within the Australian Wildlife Conservancy's Curramore Sanctuary. Two types of experimental controls, (i) undisturbed wet-sclerophyll forest sites and (ii) untreated lantana infested sites were compared to management treatments. To explore causal factors for the impacts of lantana and its treatment, habitat attributes and reptile food availability (invertebrate composition and biomass) were compared across the treatments and their influence on the reptile communities was determined.

Five surveys each consisting of (i) three pitfall trapping nights and (ii) 60 person minutes of time-constrained searches were undertaken at a total of 24 study sites (six in each treatment and control site). Quadrat, line intercept and random point-specific measurements of habitat attributes were also recorded at each site. The effects of

treatment type on reptiles were analysed using ANOVA and ANOSIM with significant associations between reptiles and habitat attributes investigated using BIOENV. Similar tests were used on habitat attributes and invertebrate composition.

Distinct reptile assemblages and habitat characteristics were detected at each of the treatments and controls. The differences in habitat attributes among the treatments were likely determinants of reptile composition. Common *Lampropholis* skinks were abundant in the manually cleared sites, which contained less understorey vegetation cover, as well as lantana infested sites, which had an open upper canopy. *Lampropholis* skinks have a preference for cleared areas and canopy gaps which may explain their prevalence within these sites. The open canopy and high amount of cleared areas may also be the reason for the exclusion of other habitat specialists from these sites. Herbicide sprayed and prescribed burnt sites contained a more diverse habitat structure, and subsequently supported a greater diversity of reptiles. Undisturbed forest sites were found to contain a low abundance of reptiles as fewer *Lampropholis* species utilised these habitats likely due to the closed nature of the forest. However, the highest numbers of rainforest species, habitat specialists such as *Eulamprus* species, were found in these sites. Rare species, particularly *Saproscincus rosei*, were recorded at all treatments and *S. rosei* were common within lantana infested sites.

Reptile communities were mainly influenced by a subset of habitat features measured in this study, specifically (i) palm frond litter and (ii) silt content in soil; which differed among the treatment types. The importance of palm litter highlights the use of heterogeneous ground cover, which is important for many reptile species. Silt may represent differences in soil compaction, which is important for burrowing reptiles, and

may also be a surrogate for different vegetation structures and canopy cover which is likely to influence reptiles. Invertebrate composition was not found to be affected by treatment and is therefore likely not to be a limiting factor for reptiles at Curramore Sanctuary. The invertebrate groups found to be most important for reptiles were spiders and pillbugs.

The ecological effects caused by certain treatment strategies and the utilisation of lantana patches as habitat by a number of species, including rare species, both highlight the importance of utilising a patch mosaic lantana management strategy, especially at the landscape scale. This is particularly important in the context of reptile conservation as reptiles often have smaller home ranges and are incapable of moving large distances onto more suitable habitat after the clearing of lantana. The use of herbicide and prescribed fire was shown to be an ideal method to manage lantana due to the accelerated regrowth of vegetation. The consideration of faunal communities in land management is increasingly important and monitoring the outcomes of weed management in the future will be a valuable tool for conservation, particularly for reptile communities.

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# **1 Introduction**

The invasion by exotic weed species is considered to be one of the most globally significant environmental issues affecting the biodiversity of natural ecosystems (Fensham *et al.* 1994; Williams and West 2000; Sharma *et al.* 2005; Sinden and Griffith 2007). Exotic species can alter the functioning of natural ecosystems by modifying habitats and competing with native flora while suppressing native fauna; adding pressure on already vulnerable species (Groves *et al.* 2003). Certain faunal groups can be particularly susceptible to the impacts of weeds, where some studies have shown negative associations of weeds with butterflies (Samways 1996) and birds (Samways 1996; Smith *et al.* 1998; DECC 2008). However, few studies have considered the impacts of weeds on ground-dwelling fauna such as reptiles. Reptile assemblages may be significantly affected by non-native vegetation due to their ground-dwelling habits and reliance on microhabitat structure for foraging, basking and refuges (Hadden and Westbrooke 1996; Singh *et al.* 2002; Fischer *et al.* 2005). Weeds are known to substantially alter the structure of habitats (Groves *et al.* 2003), potentially altering or even removing desirable reptile microhabitats. However, broad-scale weed removal can leave a habitat bare (Swarbrick *et al.* 1995), and this can also affect faunal assemblages, particularly reptiles (Valentine and Schwarzkopf 2008). Therefore it is important to consider the outcomes of weed management strategies before undertaking large scale clearing or management.

### ***1.1 Impact of weeds in Australia***

Weeds are a major threat to Australian biota due to their high degree of endemism, increasing the susceptibility of these ecosystems to invasive species (Williams and West 2000). Over one thousand special conservation areas are threatened by weeds in Australia, while economic impacts of weeds has been touted as the foremost problem for land use and resource management (Sinden *et al.* 2004). Naturalised invasive plants in Australia now comprise about 10% of the total floristic species richness (Groves *et al.* 2003). However, they are still a poorly understood threat to native plants and associated wildlife (Csurhes and Edwards 1998; Groves *et al.* 2003). Nonetheless, these invasive plants are significantly impacting on the structure and diversity of eucalypt forests and rainforests in eastern Australia (Gentle and Duggin 1997).

### ***1.2 Weeds and fauna***

Invasive plants have been found to alter wildlife community structure and composition, leading to impacts on invertebrate and vertebrate abundance and species richness (Vranjic 2000; Toft 2001; Houston and Duivenvoorden 2002; Walden 2002; Valentine *et al.* 2007). For example, Braithwaite *et al.* (1989) investigated the effects of *Mimosa pigra*, which impact on the habitats and faunal assemblages of wetlands in northern Australia (Walden 2002), finding that waterbirds were negatively affected by *Mimosa* presence, mainly through habitat loss. Furthermore, small mammals were found to favour the density of *Mimosa* stands, while reptiles were rarely found in *Mimosa* dominated areas (Braithwaite *et al.* 1989). Different faunal groups display variable responses to weed species and controlling weeds can have varied impacts. Therefore,

studying the specific impacts of weeds and treatment regimes is important for maintaining ecosystem diversity.

Variable impacts can also be displayed by specific species, as displayed by reptiles. For the control of rubber vine (*Cryptostegia grandiflora*) by planned burning, a weed that is avoided by Australian reptiles, Valentine and Schwarzkopf (2008) found that a number of species were negatively affected by burnt treatments, including *Carlia pectoralis* (open litter rainbow skink) and *Heteronotia binoei* (Bynoe's gecko). A different species, *Carlia munda* (striped rainbow skink), was more abundant in the burnt treatments, however (Valentine and Schwarzkopf 2008). These responses were caused by differing microhabitat requirements of the species (Valentine and Schwarzkopf 2008).

As shown in past studies, not all faunal impacts are negative, as invasive plants may also provide suitable habitat for native fauna, especially in the absence of natural vegetation, and the removal of an invasive plant without restoring native vegetation may leave fauna without adequate cover or food (Zavaleta *et al.* 2001; DECC 2008), resulting in local population declines. Despite the potential for such consequences actual examples of weed eradication leading to declines have not been well recognised or studied (Zavaleta *et al.* 2001).

### ***1.3 Lantana camara***

*Lantana camara* (L. Verbenaceae), native to central America, is a noxious invader of many tropical and subtropical regions and is considered to be a major weed species in over 70 countries or island groups (Sharma *et al.* 2005; Zalucki *et al.* 2007). *Lantana* has been present in the Australian landscape since 1841 when the species was first recorded,

and as a result of its longevity in Australia, is widely considered to be a naturalised species (Fensham *et al.* 1994; Sharma *et al.* 2005). Lantana covers approximately 4 million hectares or about 5.1% of the Australian landscape (Walter 1999; Sinden *et al.* 2004; Turner *et al.* 2008) (Figure 1.1). Despite the invasive capacity of lantana there are few detailed studies on its biology and ecology (Fensham *et al.* 1994; Day *et al.* 2003b; Stock 2004; Sharma *et al.* 2005), particularly those examining faunal responses to lantana and its management.



**Figure 1.1.** Current distribution of *Lantana camara* in Australia (Pestinfo 2008).

Lantana thrives in a number of east Australian forests (Walter 1999; Day *et al.* 2003b; Sharma *et al.* 2007) and threatens more than 1386 native plant and animal species, including more than 300 of conservation significance such as high-priority plants and animals like the endangered shrub native justicia (*Harnieria hygrophiloides*) and

endangered mahogany glider (*Petaurus gracilis*) (Groves *et al.* 2003; Turner *et al.* 2007; DECC 2008; Turner *et al.* 2008).

Lantana is classified as a *Weed of National Significance* due to its invasive capacity and threats to biodiversity (NHT 2003; DECC 2008) and is declared as a Class 3 weed under the Land Protection (Pest and Stock Route Management) Act 2002 (Qld) (DPI 2008). Lantana is also listed as a key threatening process under the Threatened Species Conservation Act 1995 (NSW) (NHT 2003; DECC 2008). The impacts of lantana are not confined to natural ecosystems where significant agricultural impacts can be found on pasture land, livestock, and timber plantations (Swarbrick *et al.* 1995; Day *et al.* 2003a; Day *et al.* 2003b; DECC 2008). However, this research will be focusing on natural ecosystems and in particular impacts on faunal communities.

### **1.3.1 Reproductive biology of lantana**

Lantana can form dense thickets over large areas reaching heights of 5 m (Totland *et al.* 2005; Sharma *et al.* 2007), but has the ability to climb and grow over surrounding trees up to 15 m in height (Day *et al.* 2003b; Sharma *et al.* 2005; Totland *et al.* 2005). The variety of growth forms of lantana, ranging from small singular shrubs to dense monospecific thickets, and climbing plants (Day *et al.* 2003b; Sharma *et al.* 2005), its ability to be present within diverse habitats and different soil types, has assisted the widespread establishment of the weed and complicated its successful management. Lantana grows most successfully in unshaded areas such as the edges of tropical and subtropical forests, beachfronts, warm temperate forests, forests recovering from logging or wildfire, degraded land and pasture (Sharma *et al.* 2005).

A number of biological characteristics have lead to lantana's success as an invasive species; the most important being its genetic variation, where there are currently 29 morphologically defined variants present in Australia (Day *et al.* 2003a; DECC 2008). Other traits which have assisted lantana include: ability to adapt its development and growth in response to environmental changes; fitness homeostasis<sup>1</sup>; widespread geographic range; dispersal that benefits from destructive foraging activities; fire tolerance; vegetative reproduction; allelopathy<sup>2</sup>; and superior competitive ability compared to native species (Sharma *et al.* 2005; Totland *et al.* 2005). Zoochory also facilitates the spread of lantana, where seeds are dispersed primarily by birds, but also by livestock (sheep, cattle and goats), foxes, kangaroos, bearded dragons and some rodents (Fensham *et al.* 1994; Sharma *et al.* 2005).

### **1.3.2 Lantana invasion**

There are a number of suggestions for the proximate causes of lantana invasions in any particular habitat. The foundations for lantana establishment begin with some form of disturbance which increases light availability, enabling weed encroachment (Fensham *et al.* 1994; Swarbrick *et al.* 1995; Gentle and Duggin 1998; Day *et al.* 2003b; DECC 2008). Lantana is usually limited to forest gaps caused by disturbance and does not grow successfully when shaded, making light availability one of the most important limiting factors for lantana (Gentle and Duggin 1998; Stock 2004). Lantana can alter the natural fire regimes of forests (Unwin *et al.* 1985; Swarbrick *et al.* 1995) which can increase the

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<sup>1</sup> The ability of a species to remain at a constant fitness over a broad range of environmental conditions (Sharma *et al.* 2005).

<sup>2</sup> The release of toxins by plants, their seeds, and residues, which inhibit other species' growth (Sharma *et al.* 2005).

chance of wildfires reaching forest canopies, and is ultimately highly detrimental for most ecosystems (Hiremath and Sundaram 2005). This is a particularly threatening process in rainforests, where high intensity fires are fueled by lantana leading to the increased mortality of canopy species, further exacerbating lantana domination of the site (Fensham *et al.* 1994).

Other land management techniques, such as logging, can also facilitate the spread of lantana (Day *et al.* 2003b; Sharma *et al.* 2005). Lantana infestations have been shown to completely hinder forest regeneration for up to three decades (Day *et al.* 2003b), sometimes even longer, and therefore limiting the amount of disturbance is important for reducing lantana invasion. Despite the widespread occurrence of lantana, the ecological significance of lantana invasions, especially impacts on native biodiversity, is not well understood and further studies are required (Sharma *et al.* 2005).

### **1.3.3 The impacts of lantana on native vegetation**

With an invader as persistent as lantana, inevitably, the native flora will be affected. Lantana has been found to inhibit the growth of native vegetation (Sharma *et al.* 1988), obstruct natural succession in plant communities, and displace native species, leading to a reduction in native flora biodiversity (Day *et al.* 2003b) and sometimes the extinction of species (Mauchamp *et al.* 1998). Lantana forms monospecific thickets that can exclude native herbs, shrubs, and seedlings of trees and climbers (Swarbrick *et al.* 1995; Day *et al.* 2003b; Sharma *et al.* 2007).

The weed has had a significant impact on floral communities within Australia. Eucalypt seedlings cannot establish under lantana and therefore the structure of woodlands are

significantly changed over time following the weeds' establishment (Swarbrick *et al.* 1995). A recent national assessment found that 1246 native Australian plants are at risk from lantana invasions (Turner and Downey 2008).

Lantana has contributed to the degradation of native beachfront vegetation in Queensland, from the Gold Coast to Thursday Island (Sharma *et al.* 2007), or approximately 2800 km (Telstra 2005). Invasion of lantana into rainforests and adjacent savanna woodland in Forty Mile Scrub National Park in northern Queensland has caused declines in species richness and increased flammability of the fire-sensitive region (Sharma *et al.* 2007). Controlling the weed therefore needs to be undertaken if natural communities are to be preserved and maintained.

#### **1.3.4 The impacts of lantana on fauna**

Lantana displaces native vegetation, decreasing floral diversity (Sharma *et al.* 1988; Fensham *et al.* 1994), and therefore may potentially affect native wildlife by homogenising the landscape, which further limits available habitat (Day *et al.* 2003b). Given the duration and extent of lantana invasion in eastern Australia (Sharma *et al.* 2005), native wildlife may have adapted to utilising lantana for suitable shelter, habitat and food sources (Day *et al.* 2003b). This is most evident in the avifauna (Day *et al.* 2003b), which assist with the spread of lantana by consuming fruit (Fensham *et al.* 1994). It is therefore important to examine the impacts of lantana, and its subsequent removal, on different faunal groups.

##### *1.3.4.1 Positive impacts*

There have been few studies examining the effects of lantana and its treatment on native

fauna. Lantana provides habitat and vital food sources to a number of native birds including some endangered species (Smith *et al.* 1998; Day *et al.* 2003b). This can be important in disturbed areas such as in lowland forests cleared for agriculture where lantana thrives and there may be limited food sources available (Day *et al.* 2003b). A study by Smith *et al.* (1998) showed that the vulnerable black-breasted buttonquail (*Turnix melanogaster*) frequently uses lantana thickets for feeding and roosting, despite buttonquails preferring microphyll vine forest habitats. There are also many other fauna species that have adapted to using lantana for habitat or as a source of food (Liddy 1985; Swarbrick *et al.* 1995; Turner and Downey 2008), with recent assessments recording 142 native Australian animals, including seven that are of conservation significance (DECC 2008).

#### *1.3.4.2 Negative impacts*

The impacts of lantana are not always positive, and 141 animal species have been identified as at risk from lantana (DECC 2008). Negative impacts on avian communities have been attributed to lantana. For example, in continental areas, lantana fruits are commonly fed on by native bird species, however on a number of islands the seed dispersal is primarily undertaken by exotic birds (Loyn and French 1991; Day *et al.* 2003b). By feeding on lantana, exotic birds can increase lantana's density and distribution while decreasing the native flora diversity by spreading lantana, further displacing native bird species (Day *et al.* 2003b). This may also be occurring in Australia, not only with exotic bird species, but also common generalist birds which may be displacing less common species.

Lantana threatens at least two native species listed as endangered under the Threatened Species Conservation Act 1995 (NSW): the eastern bristlebird (*Dasyornis brachypterus*) and the black grass-dart butterfly (*Ocybadistes knightorum*) (DECC 2008). Declines have been seen in functional groups of ants in lantana-dominated areas (DECC 2008). Lantana has also caused the habitat loss of brush-tailed rock wallabies (*Petrogale penicillata*) and restricts the movement of koalas (*Phascolarctos cinereus*) (DECC 2008). The wide variety of faunal groups affected by lantana highlights the significance of this environmental issue.

#### *1.3.4.3 Lantana and reptiles*

The impacts of lantana on reptiles has not been previously investigated in detail, however they are likely to be affected as they can utilise the plant for protection and shelter (Hadden and Westbrooke 1996). Reptiles also require suitable basking locations in sunlight because of their ectothermic habits (Garden *et al.* 2007a). The density of a lantana patch allows little, if any, direct sunlight to enter (Totland *et al.* 2005), and with the large areas that lantana thickets cover this may cause the exclusion of reptiles from lantana thickets, particularly larger individuals such as snakes and monitors. This suggests a negative association between lantana and reptiles. Despite lantana occurring in forest gaps that confer high light penetration through the canopy, the larger gaps with increased light availability may lead to lantana growing at much greater densities, preventing light reaching the understorey and ground (Totland *et al.* 2005). This impact of lantana, however, is complicated by smaller reptile species which may be able to climb onto the lantana to reach the increased sunlight caused by canopy gaps that lantana occurs in, as shown with the blue periwinkle (*Vinca major*) and a small skink,

*Lampropholis delicata* (eastern grass skink) (Downes and Hoefer 2007). Removing the weed could, therefore, confer both negative and positive impacts on these organisms and this may further impact on the management requirements for controlling lantana. Determining these effects is important for the conservation of native habitats and species in Australia, as well as directing future management strategies for lantana.

The impacts of lantana on fauna are supported by a number of preliminary studies undertaken throughout the range of lantana in Queensland and New South Wales. A number of reptilian species were found to be potentially affected both positively and negatively by lantana presence (DECC 2008). Four species were found to be under high priority threat from lantana, including the elf skink (*Erotoscincus graciloides*) (rare in Qld), the Nangur skink (*Nangura spinosa*) (vulnerable in NSW), the white crowned snake (*Cacophis harriettae*) (rare in Qld) and Stephen's banded snake (*Hoplocephalus stephensii*) (rare in Qld). Seven reptiles species were found to be positively influenced by lantana, including two rare species (*Saproscincus rosei* and *S. spectabilis*) (DECC 2008), highlighting the specificity of impacts on different species.

#### ***1.4 Weed management***

Widespread impacts of invasive species in Australia have lead to weed management becoming an integral part of natural resource management. However, little consideration has previously been given to the long-term benefits or the ecological consequences of weed management practices (Williams and West 2000; Zavaleta *et al.* 2001; Valentine and Schwarzkopf 2008). A paradigm shift is required away from undertaking basic weed control (ie. removal) to implementing wider ecosystem restoration goals that consider

associated impacts caused by treating invasive species, where focus is placed on the affected ecosystem and not simply the invading species (Hobbs and Humphries 1995; Zavaleta *et al.* 2001). The historical knee-jerk reactions have focused primarily on chemical or manual removal of weeds, however contemporary weed management strategies now focus on an integrated approach where ecological impacts and consequences to invasive species management are increasingly becoming more important (Williams and West 2000). Land managers adopting these integrated approaches can use a combination of treatment techniques. These can include incorporating follow-up treatments, to result in little to no weed re-establishment and provide landscape conditions that encourage native plant regeneration (Williams and West 2000). This approach will encourage holistic assessments of threatened ecosystems and lead to more effective management of harmful weed species.

### ***1.5 Management of lantana in Australia***

Past treatment strategies for lantana have focussed primarily on biological control (Broughton 2000; Day *et al.* 2003b). Since 1914, a total of 30 agents have been released into Australia, the highest released into any one country (Day *et al.* 2003a; Zalucki *et al.* 2007; DECC 2008). These included mainly leaf- and flower-feeding insects (Zalucki *et al.* 2007). Despite these efforts, lantana is still widespread and therefore not under adequate control (Day *et al.* 2003a; Zalucki *et al.* 2007; DECC 2008). The poor success of these biological control methods has been attributed to a number of factors including insufficient climate matching and the high genetic diversity of lantana, leading to unknown host specificity (Stock 2004; Zalucki *et al.* 2007). The widespread nature of lantana occurring in a number of ecoclimatic areas has complicated its control as few

biological control agents are able to migrate across these diverse zones, in order to aggregate into large and damaging populations (Zalucki *et al.* 2007). The diverse landscapes where lantana occurs also contributes to its continuing persistence, as some invertebrates have quite specific climatic needs. In Australia, lantana is only seasonally damaged in a few locations and no agent has caused substantial widespread impact (Zalucki *et al.* 2007).

Despite previous concentration on biological control methods, a number of feasible alternatives are available for controlling lantana (Day *et al.* 2003b; Sharma *et al.* 2005). The most conventional methods include (i) chemical control, (ii) manual removal, (iii) control by fire, or (iv) an integrated approach involving a number of these methods (Day *et al.* 2003b; NHT 2003).

### **1.5.1 Chemical control of lantana**

Herbicide application is an effective method of removing lantana and there are a number of herbicides registered for its removal (Day *et al.* 2003b; NHT 2003). Smaller plants are usually more effectively controlled by 2,4-D and Torfon (picloram + 2,4-D) or fosamine, whereas glyphosate (Roundup<sup>®</sup>) has been shown to control larger plants successfully (Day *et al.* 2003b; NHT 2003). Considerable non-target effects on native flora may occur with the use of herbicides, however with the use of glyphosate at low concentrations (1:100) the non-target vegetation remains relatively undamaged while still effectively killing the lantana plants (Day *et al.* 2003b). Foliar spraying could also lead to an increase in surface and exhausted fuels, which require additional management, as well as representing a fire-risk. Although successful in controlling many weeds, including

lantana, the use of herbicides is expensive (Day *et al.* 2003b; Sharma *et al.* 2005) and follow-up treatment is often required (Day *et al.* 2003b; NHT 2003).

### **1.5.2 Manual removal of lantana**

Lantana can be controlled through manual or physical removal, which minimises disturbance to native species (Day *et al.* 2003b) but is slow, highly labour intensive, expensive and also requires additional follow-up treatment (Day *et al.* 2003b; NHT 2003; Totland *et al.* 2005). Follow-up treatment is typically in the form of herbicide spraying or further physical removal (Day *et al.* 2003b). Physical removal of lantana often leaves the ground bare (Swarbrick *et al.* 1995) and does not mimic natural or native succession that may occur with other treatment techniques.

### **1.5.3 Control of lantana by planned burning**

A more cost effective, and potentially more ecologically sensitive, option for lantana removal is the use of planned fire (Day *et al.* 2003b; NHT 2003). Fire can be effective in treating large, dense infestations, which are not easily controlled by other treatment options. Historically, studies have shown that mature lantana is quite fire tolerant with planned burns of low–moderate intensity fires not recommended for its removal (Fensham *et al.* 1994; Gentle and Duggin 1997; Day *et al.* 2003b; Sharma *et al.* 2005). Typically this was the application of fire without any other treatment technique. Under these burning parameters, regrowth often occurs as the weed is competitively superior allowing the formation of thickets and the persistence of lantana (Gentle and Duggin 1997; Day *et al.* 2003b; Sharma *et al.* 2005). Using prescribed fire is therefore most effective under the right conditions, when fires are usually of higher intensities and when

lantana is actively growing, but is also commonly used prior to or as a follow-up to manual or chemical control (Day *et al.* 2003b; NHT 2003; Sharma *et al.* 2005).

The role of fire in controlling lantana is not fully understood, yet the use of fire as a management tool may be important in Australia because of the significance of fire as an ecological process in many forests, including both wet and dry eucalypt (Williams and West 2000; Singh *et al.* 2002; Tran 2007). On the other hand, anecdotal observations (Daniel Stock, pers. comm.) have shown that even after the use of high-intensity fires, lantana has responded aggressively and recovered. Furthermore, the effect of fire (especially high intensity fire) on native flora and fauna needs to be considered before its implementation. Inappropriate fire regimes can have negative impacts on flora or fauna diversity (Whelan 1996; Elliot *et al.* 1999; Andersen *et al.* 2005; Bradstock *et al.* 2005; Whelan *et al.* 2006).

#### **1.5.4 Integrated weed control**

A combination of suitable treatment methods applied in an integrated approach has been suggested as more effective for the treatment of exotic species, including lantana (Kogan 1998; Williams and West 2000; DECC 2008), than simply the use of biological agents or other methods in isolation (Gentle and Duggin 1997; Ghisalberti 2000; NHT 2003; Stock 2004). The use of integrated approaches has been implemented in a number of areas in Australia (Buckley *et al.* 2004; DECC 2008). Research into integrated approaches for lantana removal is ongoing and new combinations are being tested (Day *et al.* 2003b; DECC 2008).

### ***1.6 Reptile communities and habitat management***

Research on the ecology and conservation of reptiles in disturbed and fragmented habitats is important to quantify their vulnerability to disturbance (Gardner *et al.* 2007). The common ground-dwelling habits of reptiles suggests that they will often be exposed to most forest management practices (Singh *et al.* 2002), particularly those involving the control of invasive vegetation (Valentine and Schwarzkopf 2008). However, the impact of a number of management practices, such as those discussed previously, is relatively unknown. Reptiles and their response to forest management practices is therefore an important topic for study.

The effects that disturbances have on reptiles have only been studied in recent years where it has been found that reptiles are vulnerable to decline after disturbances or habitat loss and fragmentation, including even low intensity disturbances, depending on the loss of particular habitat features (Singh *et al.* 2002; Driscoll 2004; Fischer *et al.* 2005; Gardner *et al.* 2007). This demonstrates that reptiles may be useful indicators of the extent of disturbances in forest ecosystems.

Although the effect of lantana on reptiles has not been investigated in detail, several studies have determined the general effects of weeds on reptile community composition. Native vegetation has been found to be important for maintaining reptile species richness, and increases in the ratio of exotic plants to native plant species, result in declines in reptile species richness (Jellinek *et al.* 2004). Hadden and Westbrooke (1996) also found negative correlations with reptile species richness and weed density.

Therefore, it is believed that if exotic plants continue to spread into native forests, reptiles which depend on native plant species diversity will decrease in distribution, becoming increasingly susceptible to local extinctions (Jellinek *et al.* 2004). This is particularly important in the case of lantana because of its monospecific nature, as it can dominate a site (Sharma *et al.* 2005), leading to high ratios of lantana compared to native vegetation. Reptiles may not be able to determine the difference between exotic and native plants, but rather that areas of lower weed cover are indicative of a less disturbed site (Hadden and Westbrooke 1996). The disturbances caused by lantana in forests may therefore lead to negative impacts on reptilian assemblages.

#### **1.6.1 Habitat attributes and reptile communities**

Vegetation structure is the single most important feature determining reptile habitat preferences and it is vital to determine the effects of habitat disturbances on reptiles through associated changes in shelter, food availability and microclimate (Hadden and Westbrooke 1996; Singh *et al.* 2002; Fischer *et al.* 2005). Disturbances such as weed infestations and the disturbance associated with management practices such as weed control can have major influences on the vegetation structure and microhabitat quality of a site (Hobbs and Humphries 1995; Zavaleta *et al.* 2001; Stewart 2003; Valentine and Schwarzkopf 2008).

The relationship between structural heterogeneity and biodiversity suggests that the simplification of habitats will lead to an overall decline in biodiversity through a loss of exploitable opportunities and the exposure of individuals to a greater amount of interspecific interactions, such as predation (Mac Nally *et al.* 2001). Forests can become

simplified by reductions in floral diversity, as well as the variety of age-structures and seral stages (Mac Nally *et al.* 2001) or alternatively through impacts associated with lantana invasions (Day *et al.* 2003b).

In order for reptiles to undertake normal activities such as feeding, breeding and sheltering from predators, a number of structural vegetation characteristics are required (Hadden and Westbrooke 1996; Jellinek *et al.* 2004; Garden *et al.* 2007a). To maximise the diversity of reptiles, it is necessary to maintain habitat heterogeneity at both microhabitat and landscape scales (Fischer *et al.* 2005).

Shrub complexity, including species richness, cover and mean vertical height, provides important microhabitat for reptiles (Hadden and Westbrooke 1996). Furthermore, high shrub diversity will likely lead to an increased diversity of invertebrate populations through increased utilisation of the shrubs, which, in turn provides a greater range of food available for reptiles (Hadden and Westbrooke 1996). The presence of lantana alters the forest structure, which may lead to less diversity of invertebrate food sources, and a lower diversity of reptiles.

Previous research has demonstrated significant positive relationships between reptiles and areas with abundant vertebrate groups such as ants, springtails and spiders (Fischer *et al.* 2005). Therefore, it is important that land managers do not encourage disturbances to the landscape that facilitate an overall reduction in the abundance of invertebrates, which may indirectly result from the removal of shrubs, native or exotic (Hadden and Westbrooke 1996; Toft 2001).

Tree canopy cover is another habitat variable that is important for some species of

reptiles (Kanowski *et al.* 2006). Canopy cover can have direct effects on habitat quality for reptiles, influencing light availability and thermal environments on the forest understorey, and is also correlated with other attributes that influence reptiles such as shrub cover (Kanowski *et al.* 2006). Other habitat characteristics that may affect reptile presence and abundance include leaf litter cover, coarse woody debris (CWD) presence and abundance, disturbance history (including fire history), rainfall, soil type, rock cover, geology and aspect (Hadden and Westbrooke 1996; Mac Nally *et al.* 2001; Fischer *et al.* 2004; Jellinek *et al.* 2004; Garden *et al.* 2007a). The thermal properties of microhabitats are also important because of the ectothermic traits of reptiles which may affect their use as refuge sites (Fischer and Lindenmayer 2005). In subtropical Australia thermal heterogeneity of ground cover vegetation, even in relatively small areas such as tens of metres, may affect the habitat use by reptiles (Fischer and Lindenmayer 2005). An invasive plant such as lantana which can form monospecific thickets over very large areas may then modify or ameliorate the thermal properties over much larger areas which can then exert a significant influence on habitat usage of certain reptile species across the landscape.

### **1.6.2 Fire and reptile communities**

Prescribed burning is often used in forest management (McLeod and Gates 1998; Singh *et al.* 2002), including for the treatment of weeds (Day *et al.* 2003b; NHT 2003; Valentine and Schwarzkopf 2008). The use of fire has been shown to alter reptile assemblages by modifying forest habitats, altering vegetation structure and composition (McLeod and Gates 1998; Valentine and Schwarzkopf 2008). However, the impacts of fire on faunal communities, like other disturbances, are varied, with some studies

demonstrating no long-term impacts (Andersen *et al.* 2005) while others have shown species-specific responses dependent upon individual habitat requirements (Singh *et al.* 2002; Faria *et al.* 2004; Valentine and Schwarzkopf 2008). This is especially important in the Australian context, which has a number of fire regime dependent ecosystems (Singh *et al.* 2002).

For many species of reptiles, fire can cause an increase in food availability in the short term, or also reduce the amount of vegetation shading which may improve habitat quality by increasing the amount of available basking sites, resulting in particular reptiles showing an increase in fitness (Fenner and Bull 2007). Food availability for reptiles may be increased by fires driving the succession of a diverse range of floral species, which can lead to an increased diversity of invertebrates utilising the sites (Hadden and Westbrooke 1996).

Fire, applied appropriately, may therefore be an appropriate and useful tool in forest management for maintaining reptile communities (Fenner and Bull 2007). However, fire can also cause the removal of preferred microhabitats and reductions in ground cover and litter cover. As a result of repeated (seven year burning cycle) low intensity burns, decreases in the abundance of the skink species, *Carlia vivax* (lively skink), can occur because of their preference for these habitat attributes (Singh *et al.* 2002). This effect has also been shown in a study testing the effect of burning for weed treatment (Valentine and Schwarzkopf 2008), however other studies have found minimal effects on reptile species richness and abundance as a result of reductions in leaf litter (Ford *et al.* 1999; Fenner and Bull 2007).

### **1.6.3 Weed treatment and reptile communities**

Manual removal and the use of herbicides are commonly used and widely applied methods of invasive species removal (Day *et al.* 2003b; NHT 2003). However, few have examined the impacts of such treatment and the associated effects on reptilian communities. Manual removal and herbicide application have directed impacts on vegetation and therefore may be less detrimental to reptile communities (Day *et al.* 2003b). However, these treatments can also leave the ground bare after weed removal (Swarbrick *et al.* 1995) and this may have significant impacts on reptile microhabitats.

### ***1.7 Aims and objectives of this study***

This study aimed to determine the effects of *Lantana camara* and its removal on reptile assemblages in a wet-sclerophyll forest in south-east Queensland, focusing on five key questions:

- 1) By comparing lantana infested and non-lantana infested sites, are differences in reptile abundance and composition detectable?
- 2) What are the effects of two weed management strategies, in particular the integrated approaches of (i) manual clearing and herbicide spraying and (ii) herbicide spraying and prescribed burning, on reptile abundance and composition?

As a result of the underlying importance and role of habitat attributes for reptile occupancy (Hadden and Westbrooke 1996; Singh *et al.* 2002), a number of site

characteristics were measured and compared between treatment types. These included understorey plant and litter cover, canopy cover and light availability above (2 m) and below shrubs, coarse woody debris (CWD) presence, rock cover, distance to nearest lantana patch, soil characteristics (colour, clay and silt content), aspect and elevation. This then provides the context for the third question in this study:

- 3) Does lantana and its subsequent management influence habitat attributes, which may then influence reptile assemblages?

The results of this question will assist in determining the possible reasons why certain effects are associated with lantana, and why different lantana treatment methods may have variable impacts on reptiles. In order to determine if these habitat attributes are causal factors, the fourth question will be:

- 4) Which habitat attributes significantly influence reptile abundance and composition?

Finally, to investigate other causal factors of the effects of lantana and treatments, the last question will be:

- 5) Is reptile prey availability affected by lantana and its treatment, and does this influence reptile assemblages in the treatments?

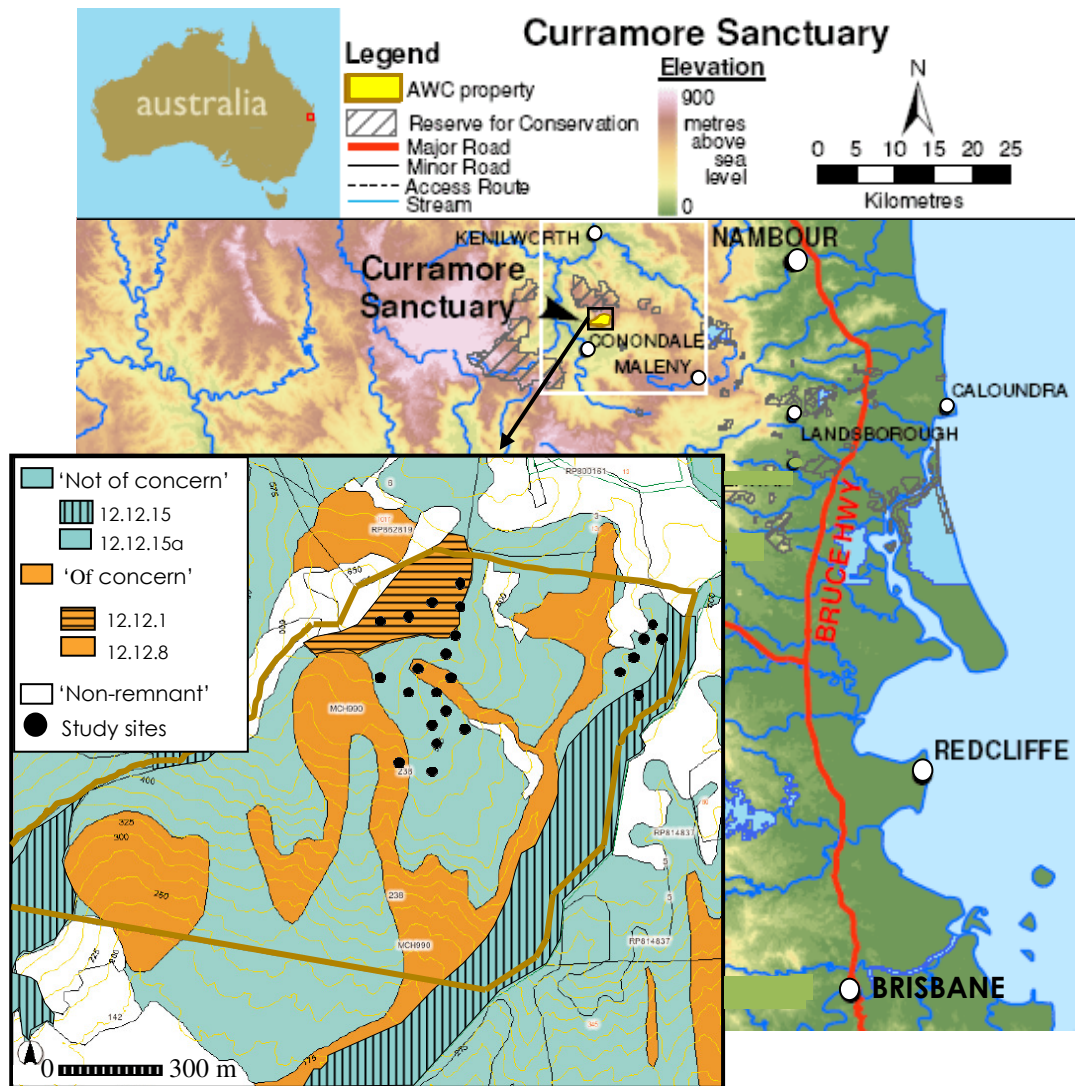
## 2 Methods

### 2.1 Study site

This study was undertaken at the Australian Wildlife Conservancy's (AWC) Curramore Sanctuary on the Sunshine Coast, a refuge for wildlife in south-eastern Queensland (AWC n.d.). Curramore (29°40'56"S 152°44'23"E) is a 175 ha forest mosaic reserve located on the western edge of the Blackall Ranges, reaching 661 m elevation (Figure 2.1). Curramore is surrounded by the largely transformed habitats of Maleny Plateau and as such it is an important remnant bushland region. However, Curramore is also an important part of a corridor of native bushland, with the Conondale Ranges, Kondalilla National Park, and Imbil, Jimna and Walli State Forests (soon to be gazetted as National Parks) within close vicinity.

Curramore has high rainfall (~1800 mm per year recorded at nearest Bureau of Meteorology automatic weather station in Crohamhurst, 30 km south of Curramore) (BOM 2008), and contains diverse geology and topographical variation that allow a diversity of ecosystems to occur. Four regional ecosystems (RE) have been recorded within Curramore including two classified as 'of concern' (EPA 2008a) (Figure 2.1), however, a more detailed study of the site by Stanton (2004) found twenty-nine distinct habitat types. The RE's of the site consist of:

RE 12.12.1 - Simple notophyll vine forest usually with abundant *Archontophoenix cunninghamiana* (gully vine forest); considered 'of concern' (OC); covering ~30% of the property (SCRC 2008); dense vegetation incorporating *Lophostemon confertus* closed forest and dominated by the plant families Lauraceae, Myrtaceae



**Figure 2.1.** Map displaying the location of Curramore Sanctuary on the Sunshine Coast and regional ecosystems (RE) across the site along with study sites. Adapted from AWC (n.d.) and SCRC (2008).

and Elaeocarpaceae; in gullies on Mesozoic to Proterozoic igneous rocks (EPA 2008a).

RE 12.8.8 - *Eucalyptus saligna* or *E. grandis* tall open forest (wet-sclerophyll); 'of concern' (OC); covering ~10% of Curramore (SCRC 2008); mid-dense with other species including *E. microcorys*, *E. acmenoides*, *L. confertus*, *Syncarpia*

*glomulifera* subsp. *glomulifera*; on Cainozoic igneous rocks (EPA 2008a).

RE 12.12.15 - *Corymbia citriodora* and *E. crebra* open forest; 'not of concern' (NOC); covering ~20% of the property (SCRC 2008); mid-dense forest with *E. propinqua*, *Corymbia intermedia*, *E. siderophloia*, and sometimes *E. microcorys*, *E. acmenoides*, *L. confertus*, *E. moluccana*, *Angophora subvelutina* with occasional vine forest species and patches of *E. pilularis*; on Mesozoic to Proterozoic igneous rocks (EPA 2008a).

RE 12.12.15a – *Corymbia citriodora* and *E. crebra* open forest; 'not of concern' (NOC); covering ~40% of the property (SCRC 2008); mid-dense consisting of *E. grandis* tall open-forest, sometimes with a vine forest understorey; sometimes also with *E. microcorys*, *E. acmenoides*, *L. confertus*, *E. siderophloia*, *E. propinqua*, *Corymbia intermedia*; occurring in wet gullies on Mesozoic to Proterozoic igneous rocks (EPA 2008a).

Not included in the RE descriptions is the presence of lantana. Within these habitats, however, large lantana thickets occur across the site, typical of other wet-sclerophyll regions. All RE's occurring on this property have been found to be impacted by lantana in other regions and medium to high impacts occur on forest edges and areas of disturbance within the RE's, with RE 12.8.8 (OC) being particularly at risk (DECC 2008).

Curramore has experienced past habitat disturbance across most of the site from historical logging and extensive weed invasion, with lantana infestation being identified as the primary land management issue (AWC n.d.). Lantana has been present at

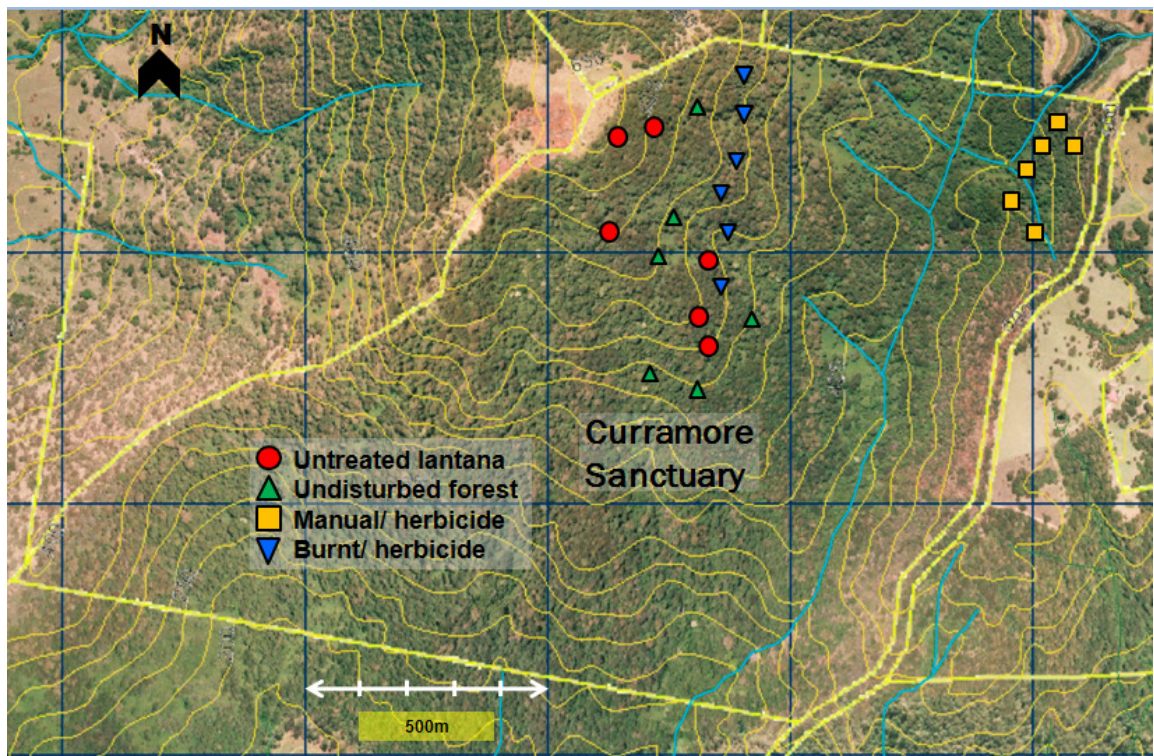
Curramore for over 50 years (Peter Stanton, pers. comm.) and the site was only recently (2003) acquired by the AWC for protection as a private reserve. Since the AWC took over management there have been concerted efforts to re-establish the natural habitats as functional ecosystems at the site, with management focusing mainly on *lantana* eradication (AWC n.d.). A systematic *lantana* control program with the aim of complete eradication commenced in February 2004, where 5 ha of dense infestations were removed with follow-up maintenance and prescribed burning that began in winter 2005 (AWC n.d.). This control program continues through to the present day.

## ***2.2 Treatments***

In total, two *lantana* treatment types and two controls, each with six replicates, were used to determine the effects of *lantana* and its removal on reptile abundance and composition. The four treatments consisted of:

- (1) manually cleared and herbicide sprayed (Treatment 1);
- (2) herbicide sprayed and prescribed burned (Treatment 2); with the controls including,
- (3) existing untreated *lantana* thickets (Control 1); and
- (4) undisturbed wet-sclerophyll forest (Control 2).

Treatment sites were randomly selected, but site selection was constrained by access and ongoing management efforts at the Sanctuary. All sites are located within the wet-sclerophyll eucalypt forest communities on the northern half of the property. As a result of AWC's management at Curramore, the manually cleared sites are spatially isolated from the other treatments (Figure 2.2) as focus for management has been undertaken in



**Figure 2.2.** Topographic map of study site at Curramore Sanctuary, Sunshine Coast, including 24 survey plots, property boundary (thick yellow), contour lines (yellow) and waterways (blue lines).

high priority areas with dense infestations. A number of habitat attributes were measured at all sites, including aspect and elevation, in order to account for any variation in the data that could be attributed to site location and exposure. The design of this study is consistent with space for time substitution research (Pickett 1991; Tyre *et al.* 2000; Krebs 2003).

The sites for treatment and control are approximately  $50 \times 50$  m ( $2500 \text{ m}^2$  or  $\frac{1}{4}$  of a hectare) and each site is located at least 30 m apart, so sites can be treated as independent, to allow for a buffer zone and reduction of scale dependent effects. Most

movement of small lizards occur within <20 m (Fischer *et al.* 2005), however some larger species may have larger home ranges (Turner *et al.* 1969; Schoener and Schoener 1982) (Figure 2.2). Reptile surveys and habitat assessments were undertaken at all sites.

All treatments were applied by the AWC, with prescribed burns taking place in September 2007 and the manual and herbicide clearing work completed between April 2007 and January 2008. There was no baseline monitoring of sites prior to the application of treatments due to the time-constraints of this project and pre-application of treatments. Numerous other studies have been undertaken that compared treatments without available pre-treatment data, including a number of reptile studies (Brown and Nelson 1993; McLeod and Gates 1998; Brown 2001; Letnic *et al.* 2004; Leynaud and Bucher 2005; Kanowski *et al.* 2006; Cunningham *et al.* 2007) . Sites were cleared manually using machetes with the lantana broken down into smaller sections and left on the ground to decompose. Herbicide was applied in larger infestations where manual removal was difficult, using a relatively new technique by application of higher-concentration (10%) glyphosate, known as the “splatter gun” technique. The herbicide was delivered using a modified gas-propelled spray-gun that allows for largescale use and minimal need to construct access tracks. Plants were left to die for a period between 3-6 months, where the plants were then broken up, and left on the ground to decompose. Spot-spraying was used as a follow-up treatment to limit regrowth (Totland *et al.* 2005). To assist with control and to limit regrowth all lantana roots were physically removed (NHT 2003), unless they were too large in which case they were sprayed with glyphosate. In the other spray and burn treatment, the same period of time (between 3-6 months) was allowed to elapse, before the application of a low-intensity fire (conducted

in September, 2007).

### **2.3 Reptile surveys**

The abundance and composition of reptiles was assessed at each site using pitfall traps and time-constrained searches. These survey techniques are regarded as the most widely used and effective survey methods for capturing the reptile species in an area (Read and Moseby 2001; Garden *et al.* 2007b). Both methods were used in order to limit the effect of sampling bias related to each method on the species recorded (Singh *et al.* 2002) where pitfalls under-represent large species (e.g. monitors, dragons and snakes) while searches may overlook small, cryptic or nocturnal species (Thompson and Thompson 2007).

Each site was surveyed five times between April and November 2008 (Table 2.1). Reptile surveys were not undertaken through the cooler months of winter (eight weeks in July–August) as reptiles become less active in cold conditions (Read and Moseby 2001). Each survey consisted of three trap nights and three time-constrained searches per site. Moseby and Read (2001) found that three trap nights allowed for an effective trapping period for various reptile species. Due to the intensive survey effort requirements of pitfall trapping and spatial separation of the sites, all 24 sites were not surveyed simultaneously and therefore the area was split into three sections of eight sites each (*sensu* Fischer *et al.* 2005). Each survey section included a random selection of treatment and control sites. This survey design was used in order to reduce the potential effect of time of month or year by surveying all treatment types at once.

**Table 2.1.** Surveys undertaken for this study and their dates.

Survey	Date
Site set-up	March
1	April
2	May-June
Habitat assessments	July
3	September
4	October
5	November

### **2.3.1 Pitfall trapping**

Pitfall trapping is often used as a method for studying small terrestrial vertebrates (Moseby and Read 2001; Schlesinger 2007). The trapability of a species though is directly affected by reptile activity and behaviour (Schlesinger 2007). Pitfall trap arrays in this study consisted of two 20 L plastic buckets buried until the rim was flush with the surface of the ground, located at both ends of 10 m of drift fence. The drift fence (polythene dampcourse) was supported by galvanised metal rods (5 mm thick) spaced every 1.5 m, with the fence standing at 25 cm height and buried 5 cm below the ground (Singh *et al.* 2002) to reduce the chance of animals crawling under the fence.

The buckets were 40 cm deep and 30 cm in diameter, with six 8 mm drainage holes drilled in the bottom. Relatively large holes were used owing to buckets with smaller holes becoming filled with water during the pilot study. The holes were covered with mesh on the underside to stop any animals from escaping. A polystyrene plate was placed in each trap during wet conditions to prevent the drowning of captured animals (Fischer *et al.* 2005) as well as leaves to allow for shelter from heat and predators (Figure 2.3).



**Figure 2.3.** Pitfall bucket flush with the ground located at one end of 10 m of drift fence, with leaves inside (Photo by D. Virkki).

Pitfall traps were located in the centre of the treated (or untreated) *lantana* areas with the fence roughly following the local contour of the site. In the uncleared *lantana* control sites, it was necessary to construct a 1 m wide track through the centre of the *lantana* thicket to enable the initial setup of the traps and for checking traps for the remainder of the study. Although a potential source of bias, this procedure was required, as it was not possible to install the pitfall array underneath *lantana* thickets, due to the thickness and density of the *lantana* stand. It is acknowledged that this access, in the first instance, opens up the canopy and may bias the results. Subsequent regrowth of *lantana* within this strip and over the track in a short time ( $\leq 1$  month) assisted in re-establishing *lantana* canopy cover that should reduce any subsequent effect from the construction of the access track.

Pitfall traps were checked in the morning on survey days, between 7 am and 12 pm; a time period which has been found to limit capture-related mortality (Longmore and Lee 1981; Hobbs and James 1999). The traps were checked in the same order on consecutive days to allow for each trap to be surveyed for a consistent amount of time (~24 hours). All captures were identified to species level (Wilson 2005) with their snout-vent length (SVL) measured using Vernier calipers to the nearest 0.5 mm. Incidental captures of other fauna, including small mammals and anurans, were also recorded and identified.

### **2.3.2 Time-constrained searches**

Active searches target heliothermic lizards that are active (Brown and Nicholls 1993) and with this in mind only passive searching was undertaken to complement the pitfall trapping. This meant searches were conducted without examining under bark, logs or rocks. The time-constrained searches were undertaken along two ~25 m transects at each site for a set period of 20 person minutes, where reptiles were systematically searched for on the ground. The same transects were used in each repeat survey. Limiting the search time to a standard value (20 minutes) ensured that the effort was comparable for each sampling unit and therefore reduced potential bias (Roughton and Seddon 2006). Survey effort consisted of two searchers, with one of the surveyors remaining constant throughout the survey period, to help reduce surveyor bias.

Searches were conducted between 10 am and 2 pm, as higher daytime temperatures are typically correlated with higher reptile activity (Read and Moseby 2001), particularly during the cooler months. In uncleared lantana sites where walking through uncleared lantana thickets was not possible because of the plants' physiognomy and density,

surveyors searched for reptiles under and in the lantana as far as could be seen from the edge of the tracks cleared for establishing pitfall trap arrays and undertaking habitat surveys.

Each reptile seen was identified from a distance, and if possible a photograph was taken (to assist with identification). This was especially important in the (control) lantana thickets, with the high likelihood that upon sighting the reptile would quickly move into the lantana before a positive identification was made. All observations were identified to species where possible, while other observations were identified to genus, depending on knowledge of species. Unidentified reptiles were grouped into a single non-specific group (Singh *et al.* 2002) in order to compare the relative abundance of reptiles among sites.

The order in which the sites were surveyed took into account the aspect of the land, where sites orientated towards the east were surveyed earlier in the day and sites orientated towards the west were surveyed later in the day in order to maximise the amount of sunlight during the time-constrained searches which is correlated with increased reptile activity (Garden *et al.* 2007a). Within each group of sites, the order of sites being surveyed was rotated throughout each survey day to reduce the effect of time of day that may influence reptile activity. Cloud cover was also recorded before each survey using the Oktas scale (0–8 for cloudiness) (Ehnberg and Bollen 2005). The cloud cover was averaged for the three-day survey period at each site. Average survey rainfall, minimum and maximum temperatures over the three days were calculated from weather data obtained from the Maleny weather station (BOM 2008).

## ***2.4 Habitat Attributes***

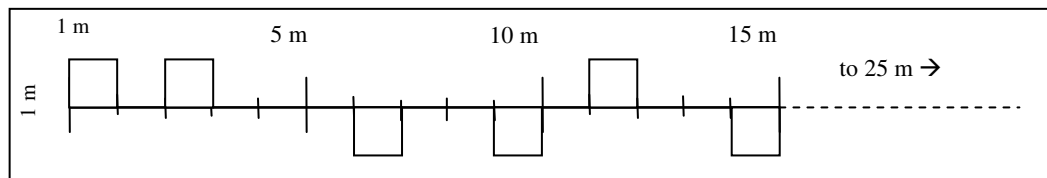
Reptile composition is influenced by a number of habitat characteristics, however, in this study, ten variables were used. Habitat attributes measured using a variety of methods included: (1) understorey plant cover (2) litter cover, (3) presence of coarse woody debris (CWD), (4) rock cover, (5) canopy cover and (6) light availability below and above shrubs (2 m), (7) aspect, (8) elevation, (9) soil characteristics and (10) distance to nearest lantana patch (Fischer *et al.* 2004; Fischer *et al.* 2005; Kanowski *et al.* 2006; Garden *et al.* 2007a). These variables were expected to assist with determining differences in reptile composition between the study sites. Structural variables were chosen rather than plant species composition as the physiognomy is expected to have a greater influence on reptile assemblages than plant species richness (Hadden and Westbrooke 1996; Mac Nally *et al.* 2001; Garden *et al.* 2007a). These attributes were also expected to be the factors that are most likely to be influenced by weed management practices, for example where shrub cover is directly removed.

For the determination of all vegetative variables, a randomised quadrat survey was chosen where quadrats were randomly placed along three 25 m transects. The quadrats (1 m<sup>2</sup>) were spaced at least 1 m apart, following a survey method similar to that outlined by Garden *et al.* (2007a). In addition to the quadrat surveys, a line-intercept method was used to estimate the proportion of CWD and rocks in each site (Greenberg 2001). Accessibility through the highly dense lantana thickets was very low, therefore 1 m wide tracks were created in order to access the lantana to facilitate the placement of quadrats along the transects. Within each site three transects were randomly (using a random number table) placed a minimum of 4 m apart roughly along the contours of the land in

order to reduce the variation in soil characteristics and vegetation assemblages (Costa *et al.* 2005; Magnusson *et al.* 2005). Two of these transects were also used for time-constrained searches and pitfall arrays.

#### 2.4.1 Understorey cover and litter cover

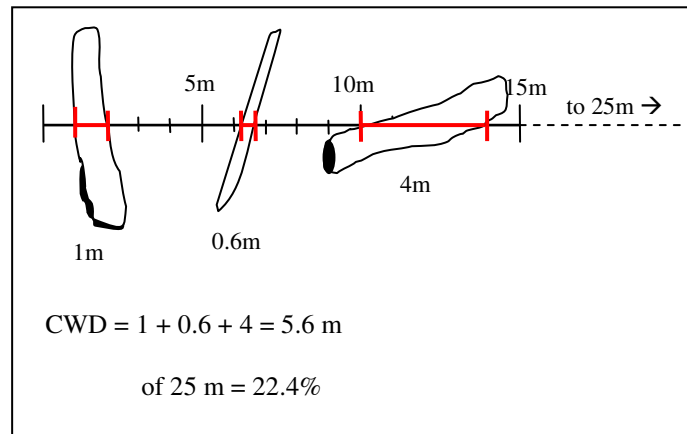
Understorey plant and litter cover were measured using the quadrat survey where understorey plants were defined as all plants <0.5 m in height (McElhinny *et al.* 2006). Ten quadrats were randomly placed along each transect, with five quadrats on either side (Figure 2.4). At each quadrat, percentage cover for both the understorey vegetation and litter was visually estimated, with the mean percentage cover calculated from the ten quadrats along that transect. The composition of the understorey cover was categorised as herb, grass, sedge, shrub, fern, vine, seedling, or lantana and litter was categorised as leaf, bark, twig, palm frond, wooden debris, root or tree trunk cover. A value for percent cover of each category was recorded.



**Figure 2.4.** Representation of 25 m transect used for measuring understorey plant and litter cover, showing random placement of  $1 \times 1 \text{ m}^2$  quadrats on either side of the transect.

### 2.4.2 Coarse Woody Debris (CWD)

Coarse woody debris was defined as woody material that is dead and in various stages of decomposition, that is not self-supporting (Turner 2006). CWD could be used by reptiles as basking or foraging sites and is therefore a useful habitat attribute to quantify (Greenberg 2001; Grove and Meggs 2003). The frequency of CWD was measured along transects, where the length that each piece of debris intercepted the line was measured (Greenberg 2001; Sutherland 2006) (Figure 2.5). This provided the total percentage of the length of transect covered in CWD, giving a percentage cover value (Sutherland 2006).



**Figure 2.5.** Determination of coarse woody debris (CWD) along a 25 m transect, measuring the intercept length of debris on the ground (shown in red) and calculation of total CWD % cover.

### **2.4.3 Rock cover**

Rock cover was measured using a similar method to that of CWD. The length that each rock, > 8 cm intercept length, intercepted the transects was measured. The minimum length of 8 cm was estimated to be a minimum size used as basking rocks for reptiles. This provided the total percentage of the length of the transects that is covered by rocks (Sutherland 2006).

### **2.4.4 Canopy cover and light availability below and above shrubs**

Canopy cover was defined as the percentage cover of the vertical projection of vascular tree crowns (McElhinny *et al.* 2006; Turner 2006). Canopy cover was calculated using nine photographs taken at each site (three for each transect) (Mueller-Dombois and Ellenberg 1974), with the photos taken at a constant height (using a level tripod) and aperture (Turner 2006). The cover provided by shrubs was calculated using photographs taken at 20 cm above the ground facing up. Upper canopy cover was calculated using photographs taken from a level tripod at a height of 2 m. The photos were taken at the start, centre and end of each of the three transects in each site. The photographs were analysed in ImagePro© which creates a binary image after manual correction of light flecks, if required (Paquette *et al.* 2007). The binary images were then analysed using ImagePro©, which found the percentage cover of each photo. An average canopy cover (%) was then calculated for each site. In addition to canopy cover, the amount of light penetrating through was also measured using a light meter at the two heights (I) above and (II) below shrubs. Light penetration was measured at three points along each transect (start, middle and end of transect), with values averaged for both shrub and canopy to find an average of each per site.

#### **2.4.5 Aspect**

The angle of the aspect of the land was measured from two points at each site, using a compass while orientated down hill. The points were 12 m apart, measured from either side of the pitfall trap buckets. The average of the two measurements was calculated to find the average aspect value per site.

#### **2.4.6 Soil characteristics**

Soil attributes were determined for each site by taking three samples per site, with one 120 ml soil sample taken from the centre of each of the three transects previously used. This was done using a spade and removing soil from the top 10 cm. Each soil sample was analysed separately, classified with McDonald *et al.* (1998) using a bolus and ribbon test to find the soil type, determining the percentage of silt and clay content. The mean silt and clay content were determined for each site. Munsell Colour Charts were also used to determine the soil colour and these were categorised on a scale from dark to light (1-7) (Munsell Colour 1977) and averaged for the mean site colour.

#### **2.4.7 Distance to nearest lantana patch**

The distance to the nearest lantana patch from each pitfall trap array was measured using GPS. This measurement was done in order to estimate movements of reptiles away from lantana. For example, if species used lantana patches they may not move long distances away from the patches.

## ***2.5 Food availability***

Food availability may also be an important indicator of reptile composition, especially for insectivorous reptiles. Food availability was measured by analysing the invertebrates found in the pitfall trap arrays used for the primary focus of this study and finding invertebrate composition and biomass. Data from the two traps at each site were initially kept separate in order to test for the effects of the presence of vertebrates on the invertebrate assemblages; however these data were later pooled per site. Both Invertebrates found in the pitfall traps were collected and grouped into broad invertebrate assemblages based on taxonomic classification, including ants and wasps (Order Hymenoptera), arachnids, beetles (Order Coleoptera) and other groups (*sensu* Letnic *et al.* 2004; Fischer *et al.* 2005). The presence of reptiles or other insectivorous fauna (ie. small mammals) were also noted to determine which invertebrate groups are potentially being used as prey items whilst inside the pitfall trap. Pitfall traps without vertebrate species were classified as containing *overall invertebrate composition* whereas traps with vertebrates may contain a subset of all invertebrate species as some predation may have occurred.

After collection, invertebrates were preserved in 70% ethanol (Mackay and Kalff 1969; Gowing and Recher 1984; Majer and Recher 1988; Fairchild 2000). The total biomass of all invertebrates in one trap was then measured to allow for biomass comparisons of food availability between sites (Letnic *et al.* 2004). To do this, the specimens were dried on absorbant tissue for approximately 10 seconds (Mackay and Kalff 1969) and the alcohol wet weight determined using an electrobalance. The wet weight of alcohol-preserved

invertebrates was used as it is approximately the same as the weight of fresh specimens (Mackay and Kalff 1969).

## **2.6 Data Analysis**

### **2.6.1 Reptile communities**

In order to normalise the data, the reptile composition matrices were log-transformed ( $\log(x + 1)$ ). This was mainly due to the high number of trapping events which did not record a sighting, in addition to the dominance of certain reptile species (Zar 1984) (particularly *Lampropholis* species). Furthermore, all unidentified reptiles were removed from these reptile matrices to reduce the variation that this may cause, before further statistical analysis. Unidentified reptiles in the data-set were analysed in comparing overall relative abundances between sites and treatments.

Bray-Curtis distance measures were used to create dissimilarity matrices for the log-reptile composition data over the entire survey period separately for both survey methods – (i) pitfall traps and (ii) time-constrained searches (Primer 2006). The most representative reptile species of the two methods was determined by systematically finding the species that could assist in explaining most of the variation in the composition data using BIOENV (Primer 2006). This method was chosen as it effectively selects species that “best explain” the community pattern by maximising a rank correlation between the resemblance matrices (Primer 2006). With this technique, the first to the fourth species of reptiles recorded in this study that explained the most variation in the reptile matrices were found. The significance of these associations was determined by executing 1000 permutations which tests for a significance value, with

alpha ( $\alpha$ ) set at 0.05. In order to graphically highlight the statistical and ecological patterns found for these species, variations in log-reptile composition between the sites were plotted using non-metric multidimensional scaling (MDS) in two dimensions with 100 iterations. The patterns in the abundances of “important” species at each site were overlaid using bubble plots.

### **2.6.2 The effect of sampling method and treatment**

To increase the robustness of the treatment analyses the reptile abundance, species richness data and reptile matrices were pooled across all survey periods. The raw data for abundance and species richness were examined using Q-Q and box-plots in order to test the assumptions of the analyses, including a normal distribution and homogeneity of variance. Repeated measures ANOVA analyses were undertaken in order to compare reptile abundance and species richness between treatment and method type (the repeated measure), with alpha ( $\alpha$ ) for significant results set at 0.05. In order to graphically display any trends in reptile abundance and species richness between both the methods used and the lantana treatment strategy, standard error plots were used.

Due to a high instance of zero reptile recordings throughout the study, reptile composition data was transformed using a simple log ( $x + 1$ ) function. This was then compared between each method type using ANOSIM, with alpha ( $\alpha$ ) = 0.05. In order to compare the effectiveness of each method, the total survey efficiency per method type and survey efficiency for each treatment per method type were calculated. This was done by dividing the total number of captures over all survey periods by the total survey time (for example hours of time-constrained searches and trap nights for pitfall trapping).

Additionally, the Jaccard Index of community similarity was used to compare the species caught with each method (Mueller-Dombois and Ellenberg 1974). This Index calculates the number of species common to both trap types divided by the total number of species in both trap types, where 0 is the least common and 1 occurs when both methods have the same species composition (Jenkins *et al.* 2003). This index is commonly used in a number of fauna studies (Heck Jr. 1977; Da Silva Jr. and Sites Jr. 2002; Jenkins *et al.* 2003; Krasnov *et al.* 2005).

Preliminary data analyses showed statistically significant differences between reptile composition (after log transformation) between the two method types. With this in mind, the treatment effects were analysed separately between the methods. Reptile composition data from both pitfall trapping and time-constrained searches were log-transformed and compared among treatments using ANOSIM analyses. Additionally, pairwise post-hoc analyses were undertaken in order to determine which of the treatments were statistically different. In order to graphically display these trends in treatments, non-metric MDS was used for both methods, in two-dimensions with 100 iterations. The MDS plots were derived from Bray-Curtis dissimilarity matrices for log-reptile composition from each method.

In order to determine the total effects of treatment type on the species composition, a binary value (0-1) for presence/absence of species was used, pooling both pitfall trapping and time-constrained search data. Transforming these data to a binary value facilitated the determination of total species composition as it reduced the effect of differences in the reptile composition recorded by each method. Preliminary analysis showed the time-

constrained search method resulted in higher abundances but similar species richness to the pitfall trapping method. In order to reduce the skewness of the data, the presence/absence method was undertaken as it reduces the effect of differing abundances (Karakassis and Hatziyanni 2000) and has been shown to eliminate differences in method type in the past (Somerfield and Clarke 1996). This provided a more useful total species composition analysis across the study period. The presence/absence data were compared among treatments using an ANOSIM and pairwise post-hoc analyses. Non-metric MDS plots derived from Bray-Curtis dissimilarity matrices were used to display any trends in reptile presence/absence among treatments.

### **2.6.3 Invertebrate communities**

The invertebrate composition matrices were log-transformed ( $\log(x + 1)$ ) for a similar reason to the reptile data, as certain invertebrate groups dominated the data set (mainly ants), whereas other groups had low abundances (Zar 1984). Invertebrate composition was compared between treatments and survey periods using a two-way ANOSIM with pairwise post-hoc testing. In order to graphically display any trends, a Bray-Curtis dissimilarity matrix was created and a non-metric MDS was used in two-dimensions with 100 iterations. Invertebrate biomass was also compared between time and treatments using one-way ANOVA analyses and LSD post-hoc testing. A standard error plot was used to graphically display any trends in invertebrate biomass.

#### **2.6.4 Patterns in habitat attributes**

Habitat attribute data were normalised (by subtracting means and dividing by standard deviation) prior to analysis as all attributes were not measured in the same units and this allows for meaningful distances between samples (Clarke and Ainsworth 1993; Primer 2006). In order to determine which attributes were correlated with each other, a Pearson correlation matrix was used and a cluster diagram drawn with the group average method in order to visually display the trends in the correlations. Correlated variables were tested by linear regression to determine if the relationships were statistically significant, with alpha ( $\alpha$ ) = 0.05. This test aimed to assist in the understanding of the composition of habitat attributes at the sites. All variables were used in the correlations with reptile assemblages, despite any correlated habitat attributes, in order to pin-point which specific variables were related to reptiles.

The habitat attributes matrix was compared among treatments using an ANOSIM and pairwise post-hoc tests. A non-metric MDS was generated from a Euclidean dissimilarity matrix in order to display any trends in the habitat attributes among treatments. Differences in habitat attributes among treatments were analysed using one-way ANOVA and LSD post-hoc tests. For these tests, standard error bar plots were used on significant results to display the trends found.

A Euclidean distance measure was used to create dissimilarity matrices for habitat attributes at each of the study sites and BIOENV was used to determine the habitat attributes that helped to explain variation in the data matrix (Primer 2006). The first to the fourth habitat attribute that explained the most variation was found and 1000

permutations were run in order to determine the significance of these relationships. Alpha ( $\alpha$ ) was set at 0.05. Patterns in these attributes among sites were shown graphically using non-metric MDS in two dimensions with 100 iterations after compiling a Euclidean dissimilarity matrix on the habitat attributes among the sites. The patterns in the values of the “important” habitat variables were overlaid using bubble plots.

### **2.6.5 Reptile associations with habitat attributes**

In order to determine the habitat characteristics that reptiles were utilising at the sites, meaningful statistical relationships between reptile assemblages and habitat attributes were explored using BIOENV. The BIOENV function was used as it determines which habitat attributes explain the variation (Clarke and Ainsworth 1993) in the reptile composition, a technique commonly used to relate environmental variables to faunal data matrices (Norkko *et al.* 2000; Kelmo *et al.* 2004; Spear *et al.* 2005; Urbina-Cardona *et al.* 2006). The best combination of habitat attributes (normalised) that correlated significantly ( $\alpha = 0.05$ ) with reptile composition (log transformed) from pitfall traps and time-constrained searches (methods analysed separately and pooled over time) was found by running 1000 permutations. The habitat attributes found to be correlated with reptile composition are potentially the most important attributes used by reptiles across the site. The manner of the data obtained, having two matrices of more than one dimension, is the reason this method of multivariate analysis was chosen, as when dealing with community ecology this level of multivariate statistics is required due to the complexity of the data (Gauch Jr. 1982).

### **2.6.6 Weather effects**

Four weather variables were analysed to assess their effects on reptile composition from time constrained searches. The abundance of four reptile species that were found to be “important” using the BIOENV mathematical model using the time-constrained search data from previous analyses were compared with the following weather data (i) rainfall, (ii) cloud cover and (iii) maximum and (iv) minimum temperatures using linear regressions. Reptile composition (log transformed) was compared between cloud cover and rainfall categories using ANOSIM analyses.

### **2.6.7 Reptile and invertebrate correlations**

Invertebrate composition was compared between the pitfall traps found with and without reptiles and other vertebrates in order to determine if there was a predatory factor within the traps impacting on the invertebrate assemblages. The composition of invertebrates was analysed with the presence of different vertebrate groups (mammal, reptile, anuran and snake) and vertebrate presence or absence (0-1) using ANOSIM analyses and pairwise post-hoc tests.

Preliminary analysis of the data showed no significant differences between invertebrate composition regardless of the presence or absence of vertebrates in the pitfall traps. With this in mind, the complete data set for the invertebrates was used in the following analyses.

Relationships between reptile assemblages and invertebrate composition were investigated using BIOENV in order to determine which invertebrate groups were considered important for influencing reptile composition (both log-transformed). Both

data sets were pooled over time in order to increase the robustness of the analyses. The most appropriate correlation of invertebrate groups was found for reptile composition from both survey methods. The relationship between invertebrate biomass and reptile species richness, total abundance and the abundance of single species was determined using linear regressions in order to determine if the amount of invertebrate biomass influenced the diversity or abundances of reptiles and different species.

#### **2.6.8 Invertebrate associations with habitat attributes**

Any potential relationship between invertebrate assemblages (log-transformed and pooled over time) and habitat attributes (normalised) was explored using BIOENV in order to determine those habitat characteristics that are important for invertebrate use. The best combination of habitat attributes that correlated significantly with invertebrate composition was found by running 1000 permutations. Invertebrate biomass was correlated with habitat attributes using linear regression.

### 3 Results

#### 3.1 Descriptive analyses

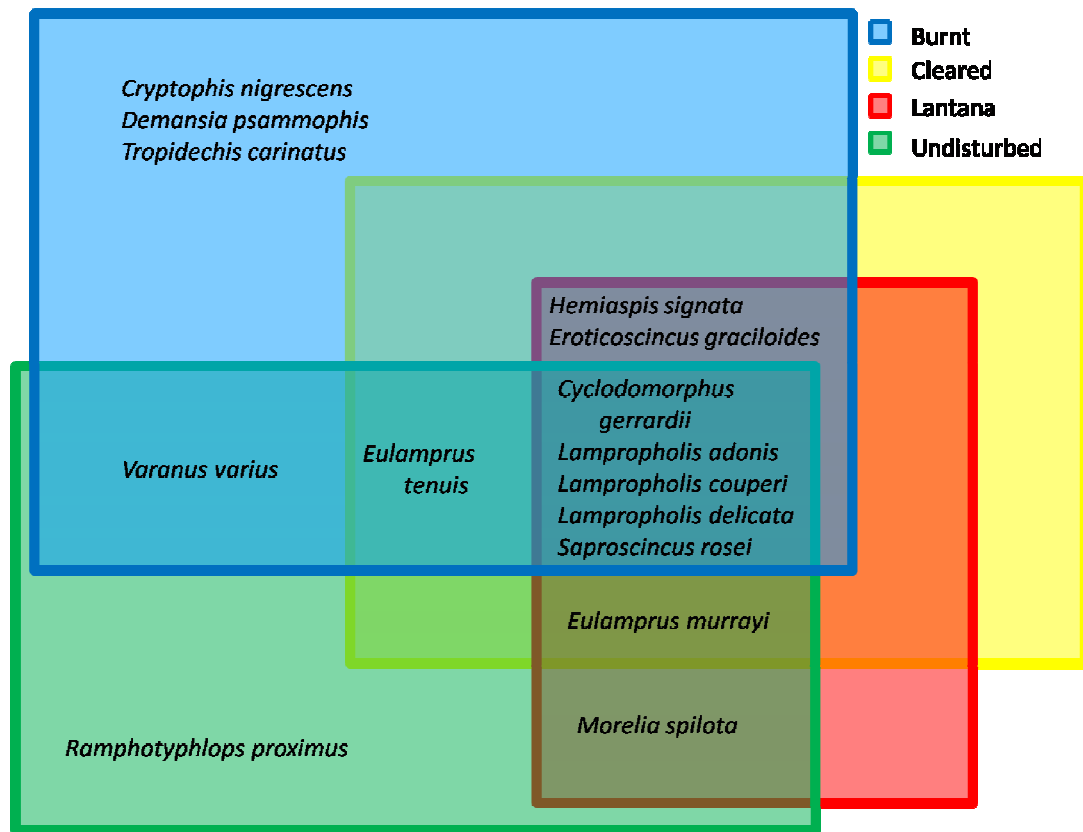
##### 3.1.1 Reptiles

A total of 15 species of reptiles from five families was recorded across the five survey periods during this study (Table 3.1). Unidentified reptiles from time-constrained searches comprised 4.5% of the total observations from this method, where a total of 574 individuals were observed in time-constrained searches and a total of 165 individuals were caught in pitfall traps (see Appendix A: 1 and 2 for raw data). A number of species were only observed by a single method, highlighting the specificity of the survey techniques. Twenty-seven percent (four species) of the species caught were caught only using pitfall traps (*Cryptophis nigrescens* – small eyed snake, *Cyclodomorphus gerrardii* – pink-tongued skink, *Erotioscincus graciloides* – elf skink, and *Ramphotyphlops proximus* – a blind snake). These four species comprised 7% of the total captures from pitfall traps; a small proportion. Twenty-seven percent (four larger reptile species) were only encountered during time-constrained searches (*Demansia psammophis* – yellow-faced whipsnake, *Tropidechis carinatus* – rough-scaled snake, *Morelia spilota* – carpet python, and *Varanus varius* – lace monitor) (Table 3.1), comprising only 3% of the total observations from time-constrained searches.

**Table 3.1.** Reptile species observed across various treatments (L = lantana, B = burnt, C = cleared, U = undisturbed) over the course of this study using two methods (P = pitfall, T = time-constrained search).

Family	Scientific name	Common name	Treatments recorded in	Method of observation
Elapidae	<i>Cryptophis nigrescens</i>	Small-eyed snake	B	P
	<i>Demansia psammophis</i>	Yellow-faced whipsnake	B	T
	<i>Hemiaspis signata</i>	Black-bellied swamp snake	L, B, C	P, T
	<i>Tropidechis carinatus</i>	Rough-scaled snake	B	T
Pythonidae	<i>Morelia spilota</i>	Carpet python	L, U	T
Scincidae	<i>Cyclodomorphus gerrardii</i>	Pink-tongued skink	L, B, C, U	P
	<i>Eroticoscincus graciloides</i>	Elf skink	L, B, C	P
	<i>Eulamprus murrayi</i>	Murray's skink	L, C, U	P, T
	<i>Eulamprus tenuis</i>	Barred-sided skink	B, C, U	P, T
	<i>Lampropholis adonis</i>		L, B, C, U	P, T
	<i>Lampropholis couperi</i>	Couper's skink	L, B, C, U	P, T
	<i>Lampropholis delicata</i>	Eastern grass skink	L, B, C, U	P, T
	<i>Saproscincus rosei</i>	Challenger skink	L, B, C, U	P, T
Typhlopidae	<i>Ramphotyphlops proximus</i>	A blind snake	U	P
Varanidae	<i>Varanus varius</i>	Lace monitor	B, U	T

Descriptively, it is evident that different combinations of species were caught in each treatment (Figure 3.1). Thirty-three percent (five species) of the species caught were found to occur at all sites regardless of treatment type, comprising 93% of the total observations over all surveys. Forty percent (six species) were identified within a combination of treatments making up 6.5% of the total observations and 27% (four species) were found only at an individual treatment type which comprised only 0.5% of the total observations. Furthermore, the herbicide and burnt sites had the highest number of treatment specialist species, with three species that were only observed here (*Cryptophis nigrescens*, *Demansia psammophis*, and *Tropidechis carinatus*). One species, *R. proximus*, was found only in undisturbed wet-sclerophyll sites, however this



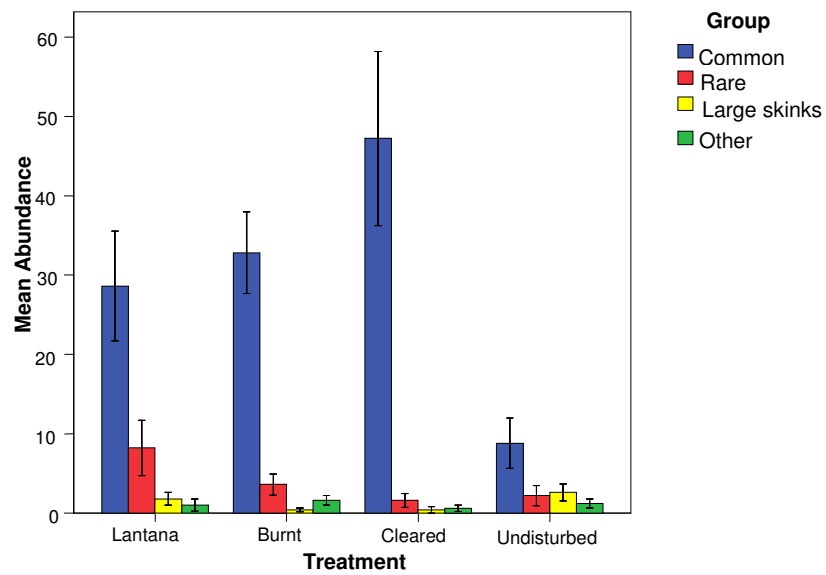
**Figure 3.1.** Representation of species that were common and unique to each treatment type. Overlapping boxes indicate common species.

was only from one observation. A high proportion (40%) of the species were not found in lantana infested sites including *C. nigrescens*, *D. psammophis*, *T. carinatus*, *R. proximus*, *V. varius* and *Eulamprus tenuis* – barred-sided skink) (Figure 3.1), likely representing a negative effect of the weed.

Mean abundance of reptile groups (common skinks, rare species, large skinks and other) (Table 3.2) over the five survey periods shows trends in the composition of reptiles within each treatment (Figure 3.2). The highest abundance was found in the manually cleared sites, which is shown to be mainly attributable to a high number of common

**Table 3.2.** List of reptile species in each reptile group found in this study.

Common skinks	Rare species	Large skinks	Other (monitors and snakes)
<i>Lampropholis adonis</i> <i>L. couperi</i> <i>L. delicata</i>	<i>Eroticoscincus graciloides</i> <i>Saproscincus rosei</i>	<i>Cyclodomorphus gerrardii</i> <i>Eulamprus murrayi</i> <i>E. tenuis</i>	<i>Cryptophis nigrescens</i> <i>Demansia psammophis</i> <i>Hemiaspis signata</i> <i>Morelia spilota</i> <i>Ramphotyphlops proximus</i> <i>Tropidechis carinatus</i> <i>Varanus varius</i>



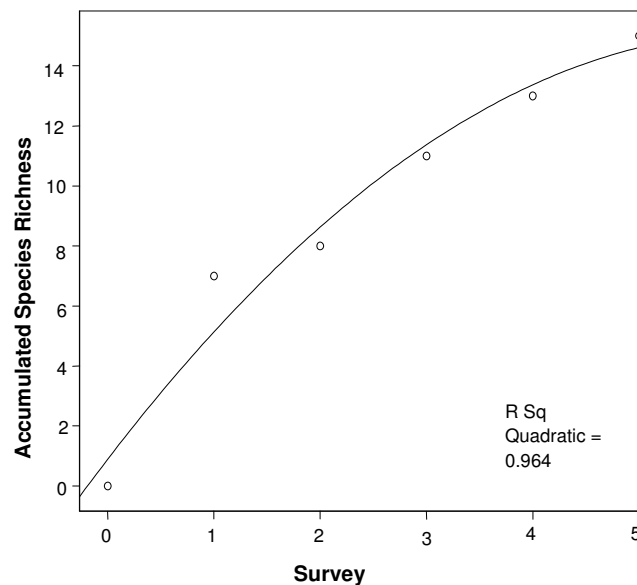
**Figure 3.2.** Mean ( $\pm$  SE) reptile abundance of reptile groups within each treatment type over five survey periods.

species; this treatment has the lowest abundances of all other reptile groups. The undisturbed forest displays a low abundance of reptiles due to much lower numbers of common species compared to all other treatments; however it had the highest abundance of large skinks. Burnt and lantana sites had similar abundances of common skinks, however the composition of the three other species groups differs. Burnt sites had the highest number of other reptiles, including snakes and monitors, highlighting the ability

of these larger species to use this habitat. The two rare species in this study, *Saproscincus rosei* and *Erotoscincus graciloides* (as listed in the Queensland Nature Conservation Act 1992), did not appear to be affected by lantana, and the former appeared to favour it, where lantana had a notably higher abundance of rare species (Figure 3.2).

Fifteen species found in this study represent 35% of the “potentially occurring” reptile species listed for Curramore Sanctuary (Eyre *et al.* 1998; EPA 2008b; AWC n.d.) (Appendix B). Although 29 potential species were not recorded, one new species, *R. proximus*, was recorded in this study.

The accumulation curve for species caught over the five survey periods fits a quadratic curve ( $R^2 = 0.984$ ) (Figure 3.3). In reaching 15 species in the final survey, the curve of the graph appeared to be leveling out, suggesting that the total species richness of the site has almost been reached (Figure 3.3).



**Figure 3.3.** Total species richness accumulation curve recorded over the five survey periods, with a fitted quadratic curve.

A number of other reptile species were also observed opportunistically, i.e. outside survey periods or when walking to/ from sites and included two species of snakes and one skink (Table 3.3). In addition to this, a number of other vertebrate species were observed during survey periods and caught in pitfall traps. This included eight mammal species and eight species of anurans, mostly caught in pitfall traps (Table 3.3).

**Table 3.3.** Opportunistic captures of vertebrates recorded from various treatments (L = lantana, B = burnt, C = cleared, U = undisturbed) over the course of this study using various methods (O = opportunistic, P = pitfall, T = time-constrained search).

Family	Scientific name	Common name	Method	Treatment
<b>Reptiles</b>				
Colubridae	<i>Dendrelaphis punctulata</i>	Common Tree Snake	O	-
Elapidae	<i>Pseudechis porphyriacus</i>	Red-bellied Black Snake	O	B
Scincidae	<i>Ophioscincus truncatus</i>	A limbless skink	Pilot	B
<b>Anurans</b>				
Bufonidae	<i>Bufo marinus</i>	Cane toad	O, P	L, B
Hylidae	<i>Litoria dentata</i>	Bleating tree frog	O	-
	<i>Litoria fallax</i>	Eastern sedge frog	P, T	C
	<i>Litoria gracilentia</i>	Dainty green tree frog	T	C
	<i>Litoria peroni</i>	Emerald-spotted tree frog	P	C
Myobatrachidae	<i>Limnodynastes peronii</i>	Striped marsh frog	P	L, B, C, U
	<i>Mixophyes fasciolatus</i>	Great barred frog	P	C
	<i>Uperoleia fusca</i>	Sandy gungan	P	C
<b>Mammals</b>				
Canidae	<i>Vulpes vulpes</i>	Red fox	O	-
Dasyuridae	<i>Antechinus flavipes</i>	Yellow-footed antechinus	P	U
	<i>Antechinus stuartii</i>	Subtropical antechinus	P	L, C, U
	<i>Planigale maculata</i>	Common planigale	P	C, U
	<i>Melomys cervinipes</i>	Fawn-footed melomys	P	L, B, C, U
Muridae	<i>Mus musculus</i>	House mouse	P	C
	<i>Rattus fuscipes</i>	Bush rat	P	L, B, C, U
Tachyglossidae	<i>Tachyglossus aculeatus</i>	Short-beaked echidna	T	C

### 3.1.2 Invertebrates

Invertebrates were classified into 18 groups consisting of ten groups of Insecta, three groups of Arachnida, two groups of Crustacea, and an “other” group (Table 3.4) (see Appendix C for raw data). Each invertebrate group was recorded in all four treatment types, suggesting there was no difference in the representation of invertebrates at the sites.

**Table 3.4.** Invertebrate groups found in the current study from various treatments (L = lantana, B = burnt, C = cleared, U = undisturbed) and their classification.

Classification		Group	Includes	Treatments recorded in
Class Insecta	Order Hymenoptera	Ant	Ants, wasps, bees	L, B, C, U
	Order Coleoptera	Beetle	Beetles	L, B, C, U
	Order Orthoptera	Cricket	Crickets, grasshoppers	L, B, C, U
	Order Dermaptera	Earwig	Earwigs	L, B, C, U
	Order Blattodea	Cockroach	Cockroaches	L, B, C, U
	Order Mantodea	Mantis	Mantises	L, B, C, U
	Order Hemiptera	Bug	Bugs	L, B, C, U
	Order Neuroptera	Antlion	Antlions	L, B, C, U
	Order Diptera	Fly	Flies	L, B, C, U
	Order Lepidoptera	Caterpillar	Caterpillars	L, B, C, U
Class Arachnida	Order Scorpiones	Scorpion	Scorpions	L, B, C, U
	Orders Aranea, Amblypiga, Phalangida	Spider	Spiders, whip-spiders, harvestmen	L, B, C, U
	Order Acarina	Tick	Ticks, mites	L, B, C, U
Subphylum Crustacea	Class Decapoda	Decapod	Amphipods, other Decapods	L, B, C, U
	Class Malacostraca	Slater	Slaters	L, B, C, U
Superclass Myriapoda	Classes Chilopoda and Diplopoda	Centipede	Centipedes, millipedes	L, B, C, U
Phyla Annelida and Mollusca	Classes Oligochaeta, Hirudinea and Gastropoda	Other	Earthworms, leeches, slugs, snails	L, B, C, U

Measurements of invertebrate biomass yielded low values over the five survey periods, with most measurements <1 g (Table 3.5). Little variation occurred between the treatments or times, except during Survey One, where each treatment had its highest biomass compared to the other survey periods. The highest measurement during Survey One was for lantana, at 1.08 g (Table 3.5). Lantana also had the highest biomass of invertebrates at all other times except Survey Two.

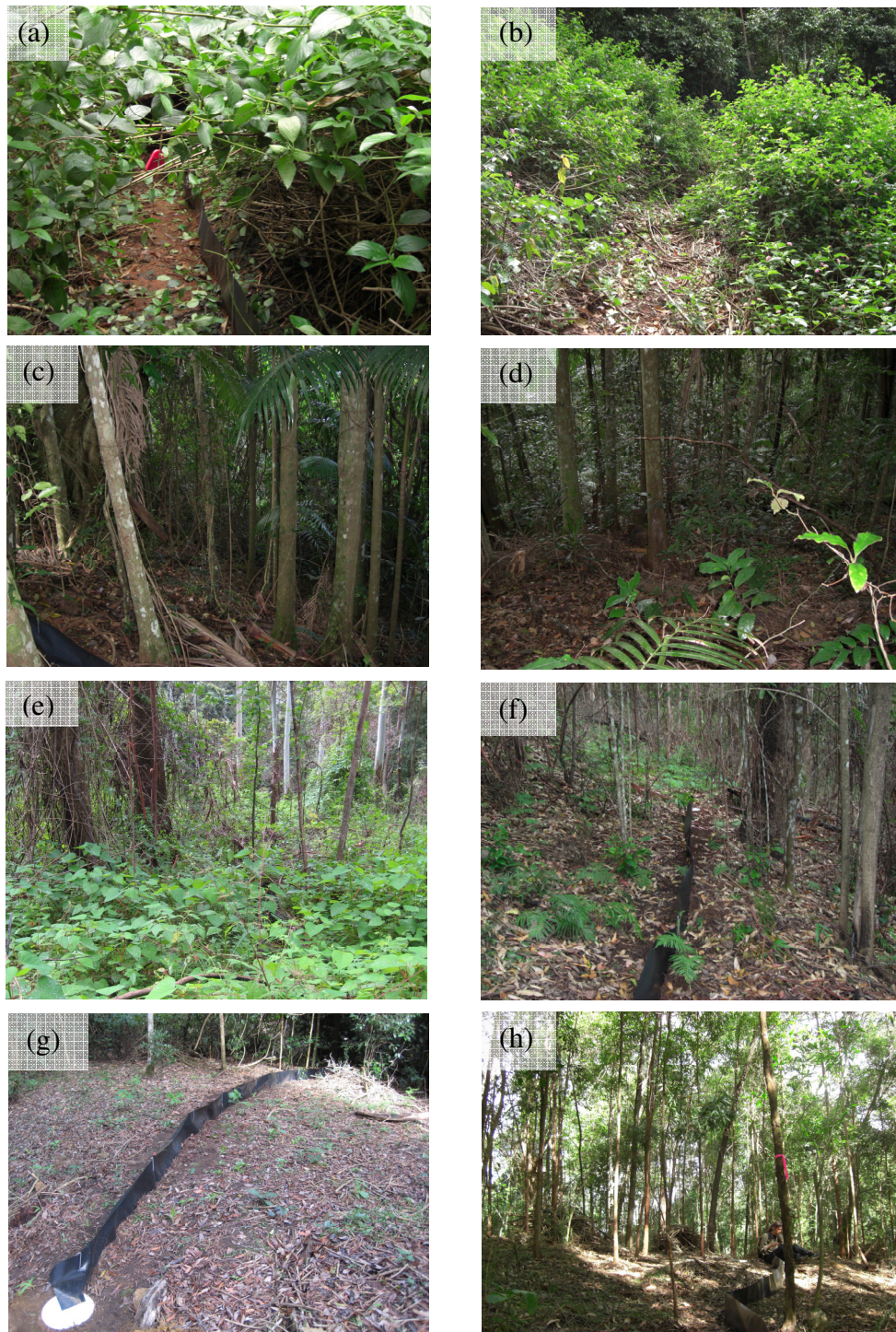
**Table 3.5.** Average invertebrate biomass (g) for each treatment at the five survey periods.

Survey	Treatment			
	Lantana (g)	Burnt (g)	Cleared (g)	Undisturbed (g)
1	1.08	0.69	0.99	1.02
2	0.50	0.59	0.53	0.29
3	0.44	0.27	0.37	0.32
4	0.50	0.42	0.47	0.35
5	0.49	0.33	0.36	0.39

### 3.1.3 Habitat attributes

The four treatment types had distinct habitat structures (Figure 3.4) (see Appendix D: 1-3 for raw data). The most evident differences in habitats among the treatment sites are as follows:

Lantana sites had a highly dense shrub layer of the weed with an open canopy above (Figure 3.4: a, b), also shown by the measurements of canopy cover and light availability above and below shrubs (Table 3.6). Undisturbed sites were very dense and structurally diverse, closed forests (Figure 3.4: c, d). These sites also had the highest canopy cover and lowest light availability, highest litter cover values, and highest percentage of clay and silt content in soil (Table 3.6). The burnt sites generally had a highly dense understorey plant layer due to regenerative growth (Figure 3.4: e), however was less



**Figure 3.4.** Photographic images of each treatment type displaying the habitat structure, of (a) lantana growth over a pitfall trap line (b) lantana patch, (c) – (d) undisturbed forest, (e) pitfall trap line at burnt sites, (f) burnt treatment, (g) – (h) pitfall trap arrays at manually cleared sites (Photos by D. Virkki).

**Table 3.6.** Average values for habitat attributes for each treatment type.

Attribute	Treatment			
	Lantana	Burnt	Cleared	Undisturbed
Altitude (m)	575.67	569.83	478.67	560.33
Aspect (°)	124	90	272	160
Bark litter cover (%)	6.49	6.09	4.45	6.32
% Canopy cover above shrubs (%)	62.49	73.90	77.65	89.64
% Canopy cover below shrubs (%)	86.17	81.91	80.67	90.95
% cClay content in soil (%)	29.86	31.53	22.15	37.36
CWD cover (%)	4.42	5.06	2.85	4.95
Distance to nearest lantana patch (m)	0.00	31.67	78.33	40.00
Fern cover (%)	0.94	4.67	2.81	7.37
Grass cover (%)	0.00	1.86	8.21	0.19
Herb cover (%)	0.66	7.72	3.37	4.14
Lantana cover (%)	46.06	0.14	0.03	0.01
Leaf litter cover (cover (%))	74.05	71.97	59.24	72.61
Light penetration above shrubs (lux)	2211.06	3842.98	3961.31	669.58
Light penetration below shrubs (lux)	11116.54	4632.15	3264.59	495.08
Palm frond litter cover (%)	0.93	0.08	0.03	2.62
Rock cover (%)	3.29	2.17	1.04	1.59
Sedge cover (%)	0.04	0.78	1.80	0.15
Seedling cover (%)	0.44	5.19	3.61	1.45
% Ssilt content in soil (%)	1.39	2.78	1.39	9.03
Small debris cover (%)	0.25	1.04	0.44	0.40
Soil Colour (1 light – 7 dark)	6	7	3	6
Trunk cover (%)	0.12	2.18	1.26	1.83
Twig litter cover (%)	5.91	8.95	8.39	7.35
Vine cover (%)	2.09	8.19	1.32	1.85

dense in sites with higher canopy covers (Figure 3.4: f). The highest average values for percentage of seedlings, herbs and vine was found in the burnt sites (Table 3.6). The manually cleared sites had low understorey growth occurring at all sites (Figure 3.4: g and h). Low cover of vine, fern and herbs occurred here (Table 3.6).

### 3.2 Reptile communities

Reptilian species that explained the most variation in reptile composition among the 24 sites were mainly from the *Lampropholis* genus (sun skinks) (Table 3.7). For both pitfall trapping and time-constrained searches, the models with the best correlations were additive models (1-4 most important reptile species) showing an increase in significant correlation values (Primer – BIOENV;  $P < 0.05$ ) with the addition of another species. Most of the variation in pitfall trap reptile composition was explained by *Lampropholis couperi* (Couper's skink) at 70.2%. Two more *Lampropholis* species were added to this model, and finally *Eulamprus murrayi* (Murray's skink) in the four variable model to explain 94.3% of the variation (Table 3.7).

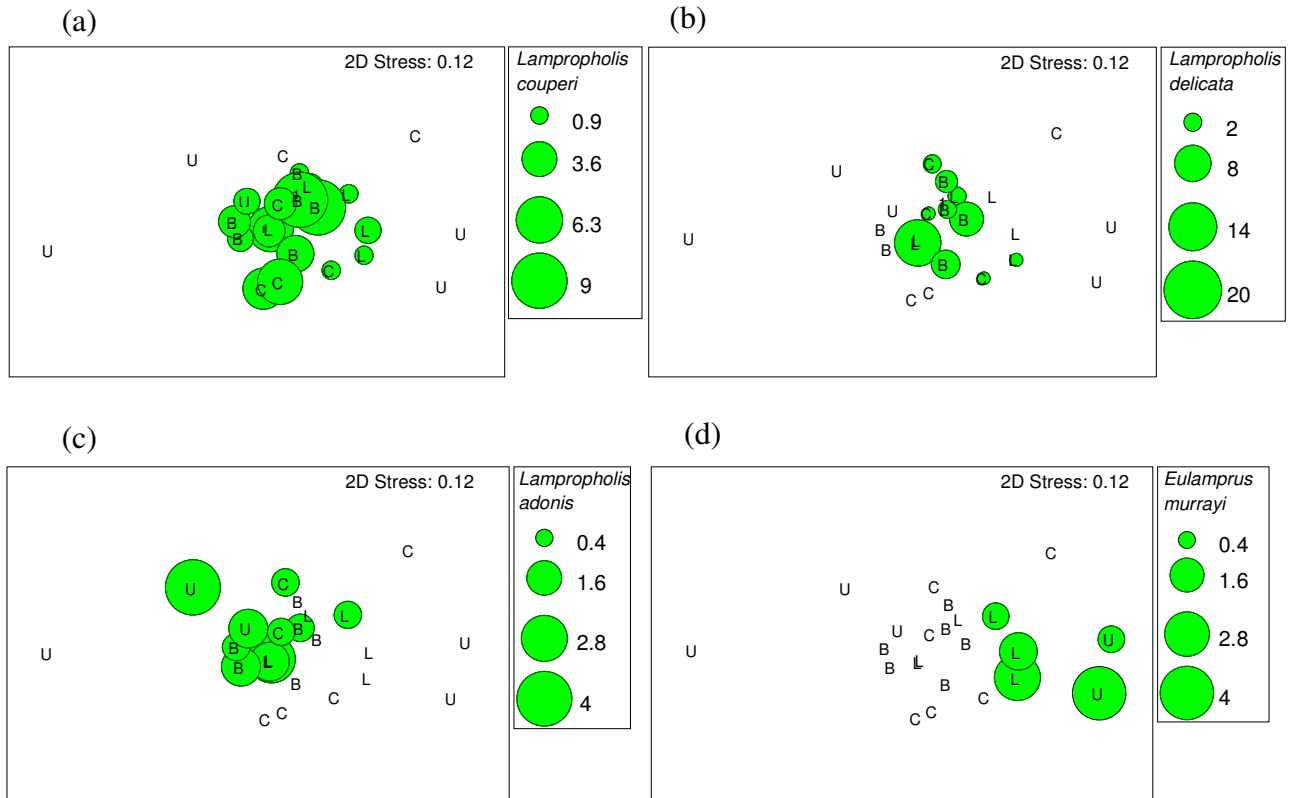
The variation in the time-constrained searches reptile matrix was mostly explained by *Lampropholis* species (only identified to genus), explaining 73.1% of the variation (BIOENV;  $P < 0.05$ ) (Table 3.7). *Saproscincus rosei* was then added to increase the explained variation to 86.4%, and then *Varanus varius* and *L. delicata* (eastern grass skink) to explain 92.4% of the variation in the four species model (Table 3.7).

Due to the significance of all the species found to be important for both methods and high variation explained, they are likely to be significantly related to the overall reptile composition matrices. These species were therefore the most commonly occurring at Curramore and are dominating the data sets. They are also likely to vary somewhat between the sites, and therefore may have specific habitat preferences causing this variation.

**Table 3.7.** Results of BIOENV analyses indicating the reptile species that best explain the variation in reptile composition over all survey periods at the 24 sites surveys (where \* indicates significant result).

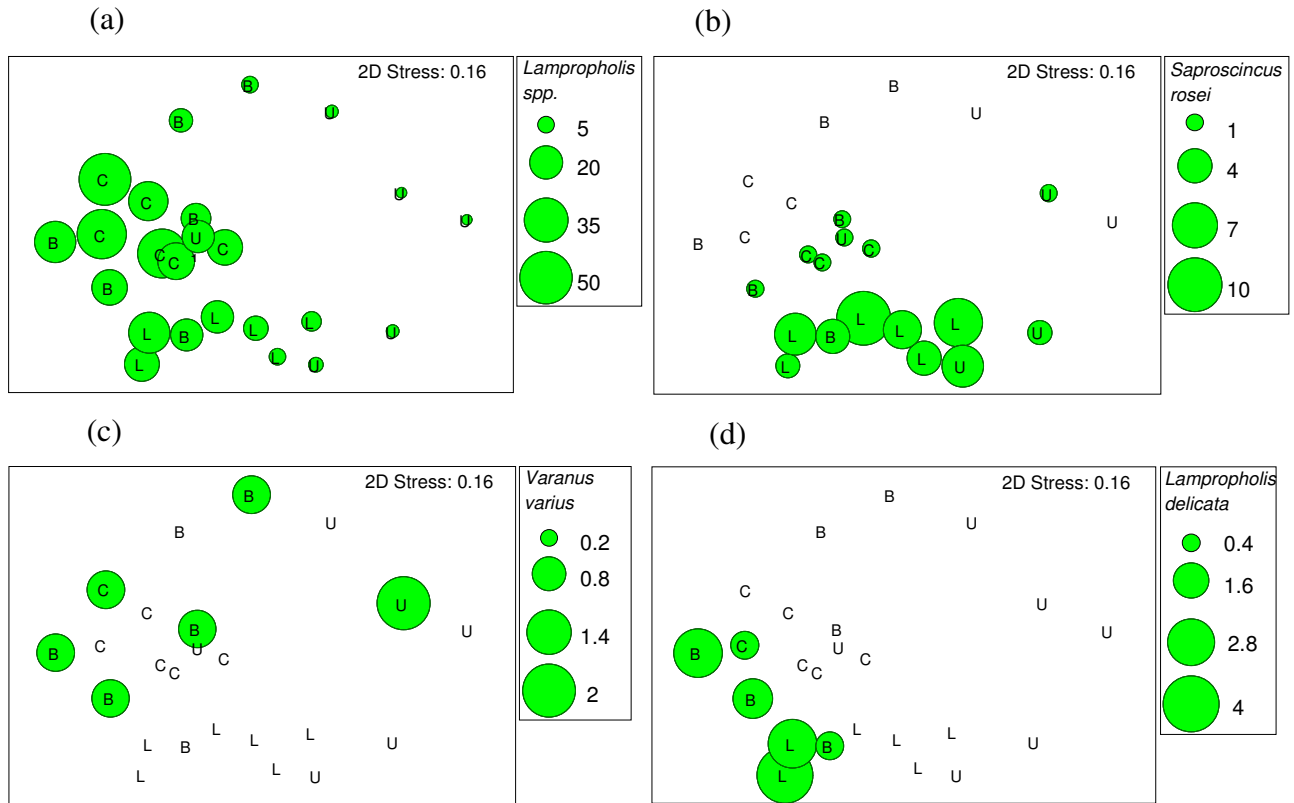
Data type	Maximum number of trial variables	Best correlation	Correlation value	Significance value
Pitfall trap reptile composition	1	<i>Lampropholis couperi</i>	0.702	0.01*
	2	<i>Lampropholis couperi</i> , <i>L. delicata</i>	0.811	0.01*
	3	<i>Lampropholis couperi</i> , <i>L. delicata</i> , <i>L. adonis</i>	0.893	0.01*
	4	<i>Lampropholis couperi</i> , <i>L. delicata</i> , <i>L. adonis</i> , <i>Eulamprus murrayi</i>	0.943	0.01*
Time-constrained search reptile composition	1	<i>Lampropholis</i> spp.	0.731	0.01*
	2	<i>Lampropholis</i> spp., <i>Saproscincus rosei</i>	0.864	0.01*
	3	<i>Lampropholis</i> spp., <i>Saproscincus rosei</i> , <i>Varanus varius</i>	0.894	0.01*
	4	<i>Lampropholis</i> spp., <i>Saproscincus rosei</i> , <i>Varanus varius</i> , <i>L. delicata</i>	0.924	0.01*

The multidimensional scaling (MDS) plots for pitfall trapping showed a single outlier, indicating a site with no observations of reptiles (see graph in Appendix E). This site was omitted from the following graphs to display patterns in reptile abundance over the remaining sites. The MDS plots of the important species found by pitfall trapping (*Lampropholis couperi*, *L. delicata*, *L. adonis* and *E. murrayi*) display clustering patterns of sites containing specific species (Figure 3.5: a-d). Each graph depicts groups of sites dominated by different species, suggesting that these species are the causal reason for the patterns in site separation. The low stress value (0.12) confirms that these species are a useful indication of the ecological responses and patterns shown for reptile composition among sites.



**Figure 3.5.** Two-dimensional MDS representations of reptile composition from pitfall traps over five survey periods (outliers removed), where size of circle indicates the abundance of the four most important species: (a) *Lampropholis couperi*, (b) *L. delicata*, (c) *L. adonis*, and (d) *Eulamprus murrayi*, at each plot. Each point represents a separate site, labelled by treatment (L = lantana, B = burnt, C = cleared, U = undisturbed).

The MDS plots are showing more distinct trends for the important species for time-constrained searches (Figure 3.6: a-d). *Lampropholis* species occurred in each site and there is a distinct decreasing trend at the sites from the left to the right of the MDS, with the most abundant sites mainly consisting of manually cleared and burnt sites (Figure 3.6: a). *Saproscincus rosei* is most abundant at sites to the bottom of the plot, mostly lantana sites (Figure 3.6: b). *Varanus varius* is not showing any distinct patterns, with this species occurring at six sites spread across the upper half of the MDS, mostly of

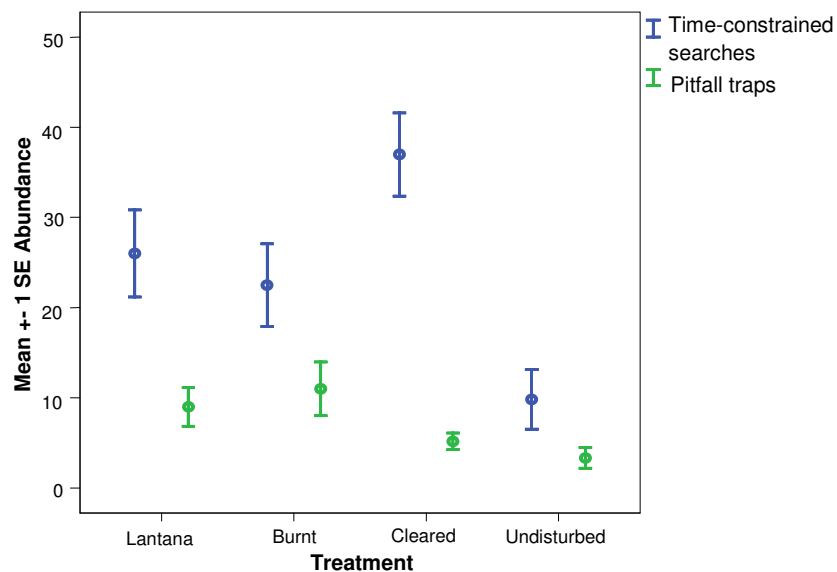


**Figure 3.6.** Two-dimensional MDS representations of reptile composition from time-constrained searches over five survey periods (outliers removed), where size of circle indicates the abundance of the four most important species: (a) *Lampropholis* spp., (b) *Saproscincus rosei*, (c) *Varanus varius*, and (d) *L. delicata* at each plot. Each point represents a separate site at a single time period, labelled by treatment (L = lantana, B = burnt, C = cleared, U = undisturbed).

manually cleared sites (Figure 3.6: c). *L. delicata* is showing a small cluster of sites with this species, consisting of burnt, lantana and cleared sites (Figure 3.6: d). The low stress value (0.12) indicates and the distinct patterns show that these observed patterns in species are a useful indicator of the separation of sites by the composition of the recorded reptiles.

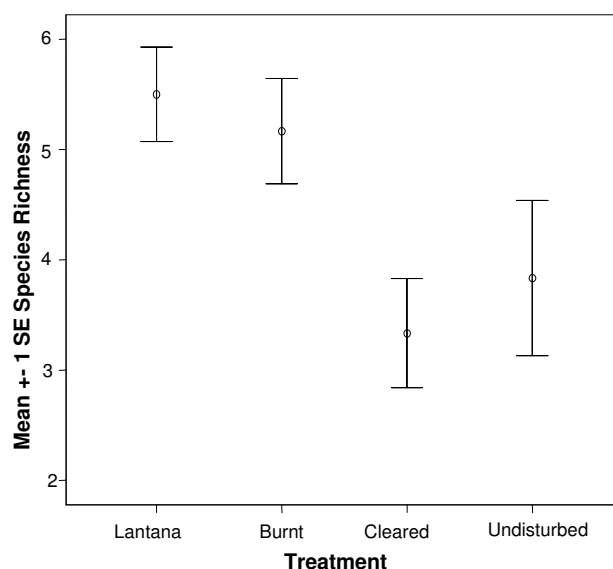
### 3.2.1 The effect of sampling type and treatment

Reptile abundance (pooled over time (see Section 2.6.2)) was significantly different between survey methods and treatment, where an interaction between survey methods and treatment occurred (Repeated Measures ANOVA;  $F=8.436$ ,  $d.f.=3$ ,  $P<0.05$ ). Time-constrained searches had significantly higher abundances at each treatment than pitfall trapping data (Figure 3.7). Using only time-constrained search data the manually cleared sites had significantly higher abundances whereas the lantana and burnt sites had higher abundances using only pitfall data. Both methods show that undisturbed sites had the lowest abundances (Figure 3.7). This suggests that the two methods had variable efficiency depending on the treatment type.



**Figure 3.7.** Mean ( $\pm$  SE) reptile abundance from two survey methods among treatment types.

Species richness (pooled over time (see Section 2.6.2)) was not significantly different between survey method (one-way ANOVA;  $F=3.214$ ,  $d.f.=1$ ,  $P>0.05$ ) and therefore species richness was also pooled between methods. A significant difference between species richness at treatment types was found (one-way ANOVA;  $F=3.760$ ,  $d.f.=3$ ,  $P<0.05$ ). Burnt and lantana treatments had a similar species richness that was higher than the other two treatments, where undisturbed and manually cleared sites also had a similar species richness (Figure 3.8). This was confirmed by post-hoc testing that found the treatments that were significantly different were: lantana and cleared, lantana and undisturbed, and burnt and cleared (LSD post-hoc;  $P<0.05$ ).



**Figure 3.8.** Standard error plot of reptile species richness per site (pooled over time and methods) among treatments.

Reptile composition (pooled over time (see Section 2.6.2)) was significantly different between sampling methods (ANOSIM;  $P < 0.01$ ). This showed that different species were caught by each method. The Jaccard Index of similarity between survey methods resulted in a value of 0.47, indicating that over half of the species caught by each method were different.

Over the survey period, pitfall traps were open for a total of 720 trap nights and time-constrained searches were undertaken for a total of 120 person hours. Time-constrained searches were more efficient than pitfall trapping at detecting reptilian individuals. Importantly, the efficiency of survey techniques varied between treatments (Table 3.8). Time-constrained searches were the most efficient in manually cleared sites. Pitfall traps, however, were the most efficient in burnt sites and both methods had low efficiency within the undisturbed sites (Table 3.8). This suggests that the particular methods may be more efficient in different habitat types, or it may simply be displaying the trends of different abundances found in each treatment.

**Table 3.8.** Calculated reptile survey efficiency for each treatment and total efficiency for two survey techniques.

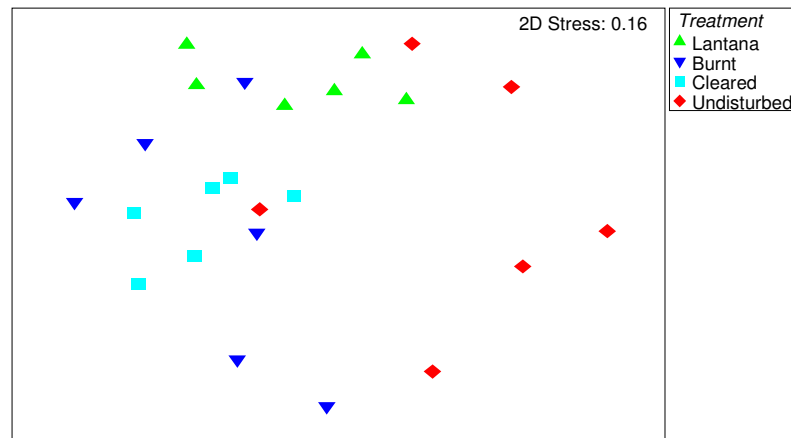
<b>Data type</b>	<b>Treatment</b>	<b>Efficiency</b>	<b>Total efficiency</b>
Time-constrained searches (per hour)	Lantana	5.20	4.78
	Burnt	4.57	
	Manually cleared	7.40	
	Undisturbed	1.97	
Pitfall traps (per night)	Lantana	0.30	0.23
	Burnt	0.36	
	Manually cleared	0.17	
	Undisturbed	0.10	

Reptile composition obtained from time-constrained searches was significantly different among treatments (ANOSIM;  $P < 0.05$ ), however pitfall trap data did not display significant differences among treatments due to insufficient data (ANOSIM;  $P > 0.01$ ) (Table 3.9). The presence/absence of species among sites displayed similar trends as the total reptile composition from time-constrained searches, with a significant difference found among treatments (ANOSIM;  $P < 0.05$ ) (Table 3.9). This difference shows that, as found by time-constrained search data and presence/absence analysis, reptile composition is affected by the treatment types or presence of lantana.

**Table 3.9.** Results of ANOSIM analyses comparing the composition of reptiles among treatment types (where \* indicates significant result).

<b>Data type</b>	<b>Significance value</b>	<b>Post-hoc (significantly different treatments, <math>P &lt; 0.05</math>)</b>
Pitfall trap reptile composition (log-transformed)	0.066	
Time-constrained search reptile composition (log-transformed)	0.001*	All treatments with eachother
Reptile presence/absence (0-1) from both methods	0.01*	Lantana and undisturbed, burnt and undisturbed, cleared and undisturbed

The MDS of time-constrained search reptile composition is displaying a grouping of treatments, particularly of lantana and manually cleared sites which are in tight clusters (Figure 3.9). Undisturbed sites are grouped to the right of the plot, with site one undisturbed site separate and overlapping with other treatments, and the burnt sites also overlapping with other treatment types (Figure 3.9). The pattern evident in the MDS plot supports the significant value found for treatment effects on reptile assemblages (Table 3.9) and the low stress of the graph (0.16) suggests that this pattern is a good

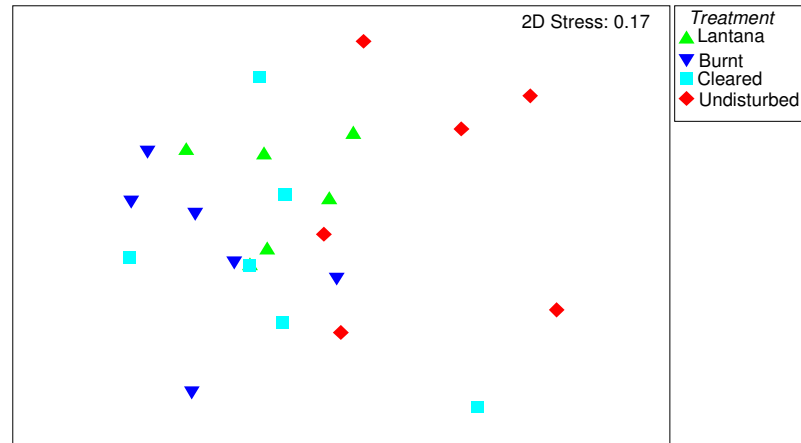


**Figure 3.9.** Two-dimensional MDS plot of reptile composition from time-constrained searches among treatment types. Each point represents a separate site.

representation of the reptile compositions at the treatment sites. The MDS plot shows different compositions of reptiles at the treatments and therefore is indicating an effect of the treatment strategy of lantana impacting on native reptiles.

The MDS plot of reptile presence/absence is showing some overlap with treatment types, where undisturbed forest is separated to the right, consistent with these sites found to be significantly different to all other treatments (pairwise post-hoc;  $P < 0.05$ ) (Figure 3.10).

The patterns shown in these MDS plot and the low stress of the graph (0.17) suggests that this pattern is a good representation of the reptile presence/absence at the study sites. The trend found is also suggesting that the treatment strategies and presence of lantana are influencing the species present at the sites compared to the undisturbed forest control sites. Although showing similar results, there are fewer patterns shown here than for

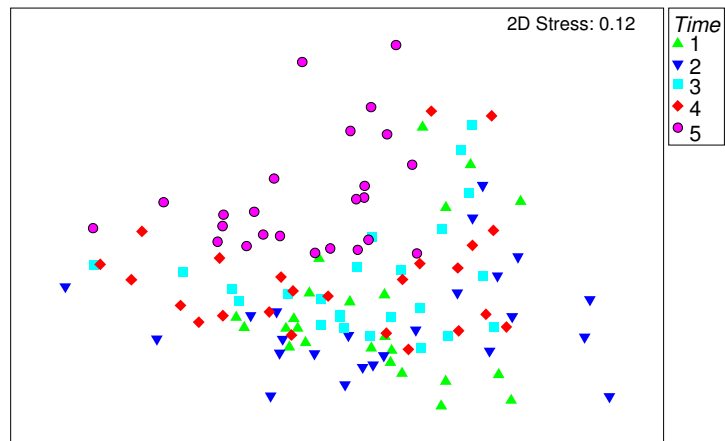


**Figure 3.10.** Two-dimensional MDS plot of reptile presence/absence among treatment types. Each point represents a separate site.

reptile composition, suggesting that the abundances of species are important in determining differences in the reptile assemblages between the sites.

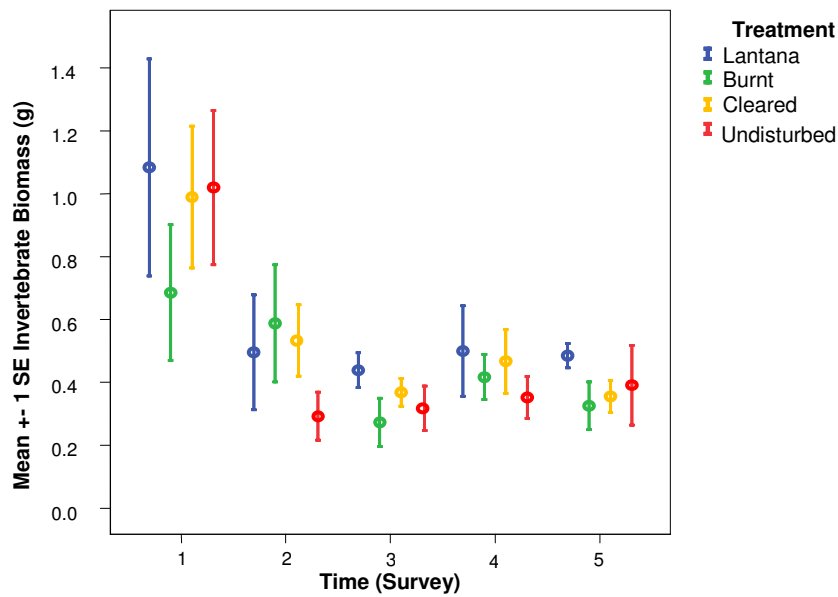
### ***3.3 Invertebrate communities***

Invertebrate composition was not significantly different among treatments (ANOSIM;  $P > 0.05$ ), however differed significantly over time (ANOSIM;  $P < 0.001$ ), clearly indicating a seasonal effect. Survey Five appeared to be very distinct, and pair-wise testing found that Survey Five was significantly different to all other times (pair-wise post-hoc;  $P < 0.001$  for all) (Figure 3.11). This was likely due to differences in overall composition and different abundances of certain invertebrates. This indicated that the composition of insects was not affected by the removal of lantana or lantana presence, however was influenced by time.



**Figure 3.11.** Two-dimensional MDS of invertebrate composition over five survey periods. Each point represents a separate site at a single time period.

Similar patterns were observed with invertebrate biomass data, where they were significantly different over time (ANOVA;  $P < 0.001$ ) but not different among treatments (ANOVA;  $P > 0.05$ ). This shows that the abundance of invertebrates was also not affected by lantana and its treatment. Post-hoc analyses found that Survey One was significantly different to all other times (LSD;  $P < 0.001$  for all). Survey One has a higher biomass than the other times, however at this time there is much higher variation among treatments (Figure 3.12). It can also be seen that the biomass within each time period is not significantly different between any of the control or treatment sites (Figure 3.12).

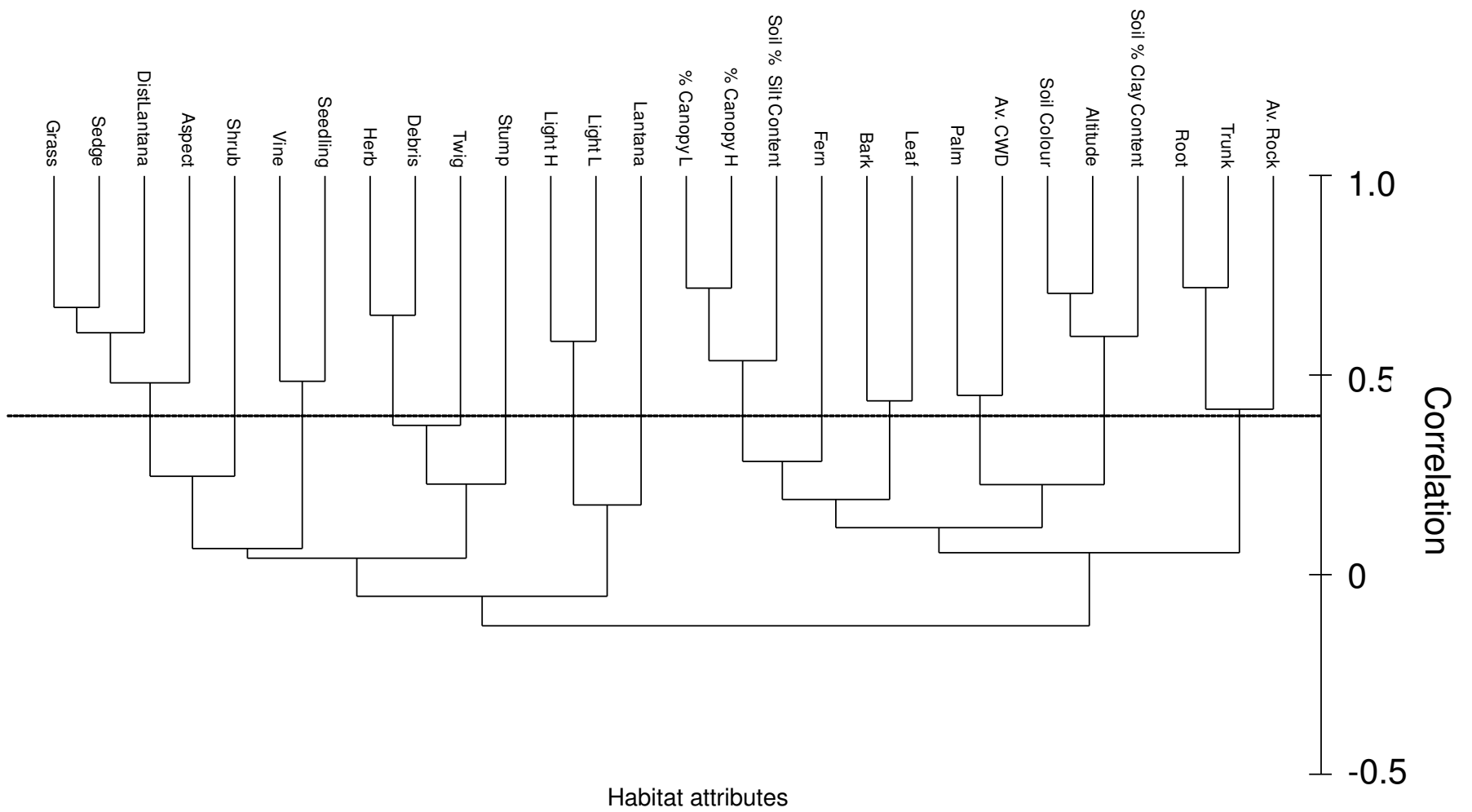


**Figure 3.12.** Mean ( $\pm$  SE) invertebrate biomass among treatments over five survey periods.

### ***3.4 Determinants of reptile and invertebrate communities***

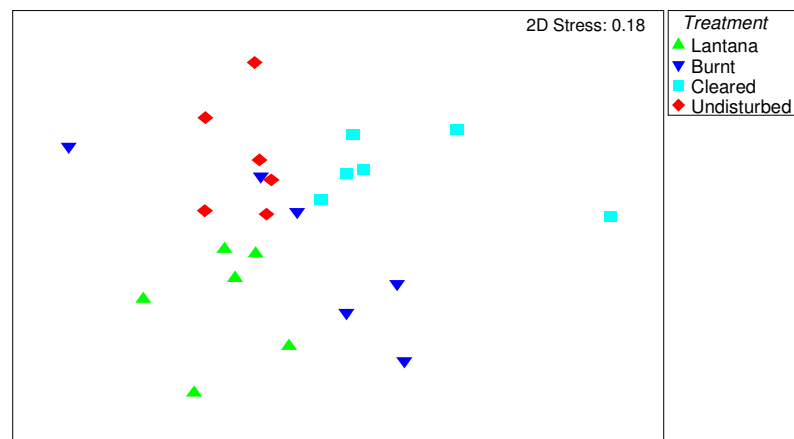
#### **3.4.1 Patterns in habitat attributes**

A number of habitat attributes were significantly correlated with each other. As shown by the line at a correlation of 0.40 on the cluster diagram, all clusters below this line are significantly correlated (linear regression;  $P < 0.05$ ) (Figure 3.13). Also, one correlation above this cutoff is significant; that of lantana cover and light availability above shrubs (linear regression;  $P < 0.05$ ). This shows that attributes within these habitats are related to or influencing other characteristics, in some cases (with altitude) this is related to site location.



**Figure 3.13.** Cluster diagram of normalised habitat attributes. Significant correlation cutoff among attributes is shown at 0.40 correlation.

Habitat attributes were significantly different among treatments (ANOSIM;  $P < 0.01$ ), where pairwise post-hoc testing found that each treatment was significantly different when compared to one another (LSD;  $P < 0.05$ ). The MDS shows a clustering of treatments, with some degree of overlap, where the manually cleared, undisturbed and lantana sites are in distinct clusters (Figure 3.14) indicating distinct habitat types in these treatments. The burnt sites are more dispersed, overlapping with other treatments suggesting that this treatment has a mixed or more heterogenous structure. Further testing found that 12 habitat attributes were significantly different among the treatments (one-way ANOVA;  $P < 0.05$ ) (Table 3.10), confirming the different habitat structures between the treatments.



**Figure 3.14.** Two-dimensional MDS of habitat attributes between treatments. Each point represents a separate site.

**Table 3.10.** Results of one-way ANOVA analyses comparing habitat variables between treatment types (L = lantana, C = manually cleared, B = burnt, U = undisturbed forest) (where \* indicates significant result).

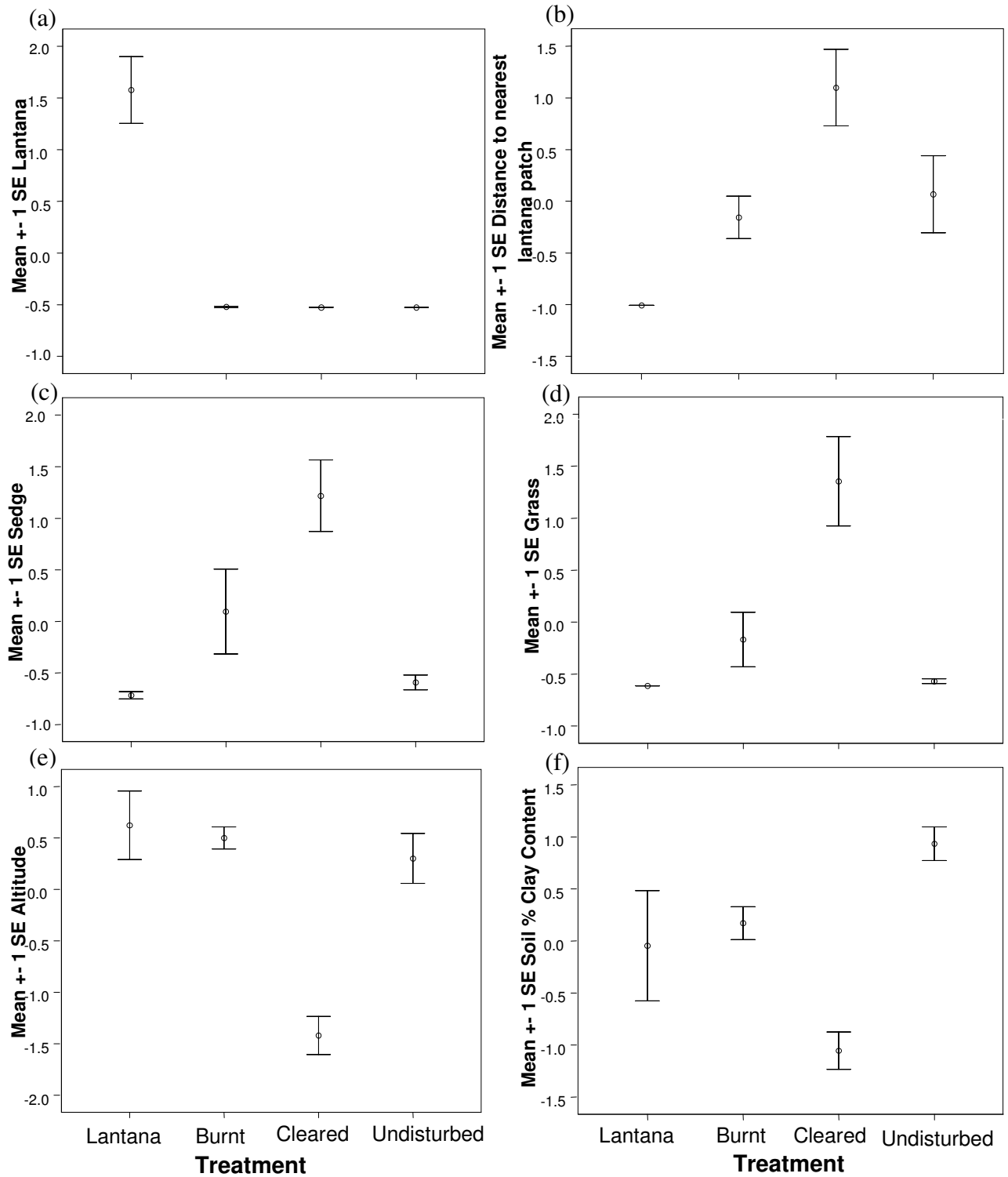
Habitat variable	Significance value	Significantly different treatments
Lantana cover (%)	<0.001*	L + B, L + C, L + U
Distance to nearest lantana (m)	<0.001*	L + B, L + C, L + U, C + B, C + U
Sedge cover (%)	<0.001*	C + L, C + B, C + U, B + U
Grass cover (%)	<0.001*	C + L, C + B, C + U
Altitude (m)	<0.001*	C + L, C + B, C + U
Clay content in soil (%)	0.002*	C + L, C + U, C + B, L + U
Light penetration above shrubs (lux)	0.003*	L + B, L + C, L + U
Soil colour (light 1 – dark 7)	0.012*	C + L, C + B, C + U
Silt content in soil (%)	0.016*	U + L, U + B, U + C
Herb cover (%)	0.022*	B + L, B + C
Canopy cover above shrubs (%)	0.023*	L + U
Aspect (°)	0.033*	C + L, C + B, C + U, B + U

### *Lantana*

Lantana treatment sites, as expected, had a significantly higher cover of lantana than the other treatments (one-way ANOVA;  $P < 0.001$ ) (Figure 3.15: a). No cover of lantana was present at either of the burnt or cleared sites, showing that regrowth of lantana did not occur. Distance to nearest lantana patch was the highest at manually cleared sites due to extensive clearing occurring, with burnt and undisturbed sites of similar distances to lantana patches (Figure 3.15: b).

### *Sedge and grass cover*

Sedge cover and grass cover (indicator of native understorey productivity) displayed similar trends, where they were both significantly higher in manually cleared sites to all other treatments (one-way ANOVA;  $P < 0.001$ ) and very low in lantana and undisturbed forest sites (Figure 3.15: c-d).



**Figure 3.15.** Standard error plots of habitat attributes significantly different between treatment types ( $p < 0.05$ ), with (a) lantana cover, (b) distance to nearest lantana patch, (c) sedge cover, (d) grass cover, (e) altitude, and (f) clay content in soil.

### *Altitude*

The site altitude was significantly lower at the manually cleared sites (one-way ANOVA;  $P < 0.001$ ) and similar between the other three treatments (Figure 3.15: e) due to the spatial separation of the manually cleared sites.

### *Clay content*

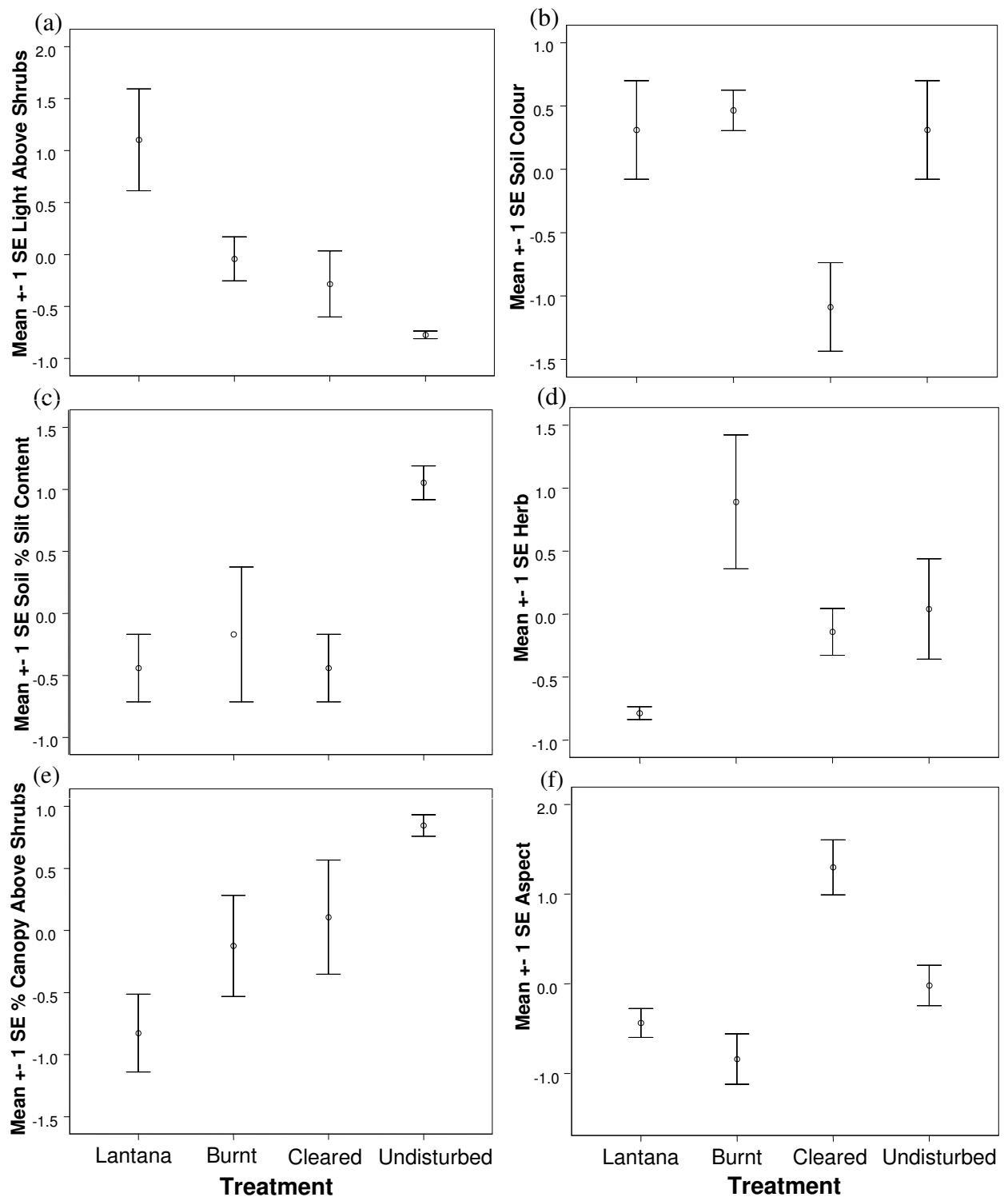
Percentage of clay in soil was significantly higher in undisturbed sites (one-way ANOVA;  $P < 0.01$ ), and lowest at the manually cleared sites (Figure 3.15: f).

### *Light penetration*

Light penetration above shrubs (from 2 m) was significantly higher in lantana sites than all other treatments (one-way ANOVA;  $P < 0.01$ ) with undisturbed sites having the lowest light penetration (Figure 3.16: a), highlighting the different structure of the two control types due to lantana encroaching canopy gaps and the closed nature of the undisturbed forest. Burnt and cleared sites were intermediate between the two controls.

### *Soil colour and silt content*

Soil colour was similar at all treatment sites except manually cleared sites, which had a significantly lower value (one-way ANOVA;  $P < 0.05$ ), therefore a darker coloured soil (Figure 3.16: b), suggesting a different type of soil at these sites and likely a different vegetation type. Percentage of silt in soil was similar at all sites except undisturbed forest, which had significantly higher silt content (one-way ANOVA;  $P < 0.05$ ) (Figure 3.16: c).



**Figure 3.16.** Standard error plots of habitat attributes significantly different between treatment types ( $p < 0.05$ ), with (a) light penetration above shrubs, (b) soil colour, (c) % silt in soil, (d) herb cover, (e) canopy cover above shrubs, and (f) aspect.

### *Herb cover*

Herb cover was the highest at burnt treatment sites, and significantly lower at lantana sites (one-way ANOVA;  $P < 0.05$ ) (Figure 3.16: d), suggesting an increase in native groundcover growth after fires.

### *Canopy cover*

Percent canopy cover above shrubs showed an increasing trend across treatments, with lantana having the lowest cover and undisturbed the highest percentage cover, where these two controls were significantly different (LSD post-hoc;  $P < 0.01$ ) (Figure 3.16: e), similar to light availability.

### *Aspect*

Lastly, aspect was variable between the treatment sites, and significantly higher at manually cleared sites (one-way ANOVA;  $P < 0.05$ ) (Figure 3.16: f), which represents a difference in sunlight availability that may influence reptile assemblages.

The single most influential habitat variable found in this study was canopy cover above shrubs (BIOENV;  $P < 0.05$ ), explaining 50.8 % of the variation in the habitat attribute data between the sites (Table 3.11). The amount and type of canopy seems to significantly influence plant function and production. Canopy cover below shrubs was the second most influential variable, highlighting the significance of canopy cover, with 62.8 % of the variation explained when canopy cover below shrubs was added to the model (BIOENV;  $P < 0.05$ ). Rock cover was variable between the sites and therefore, when added to the model, 71.6 % of the variation in the habitat attributes was explained

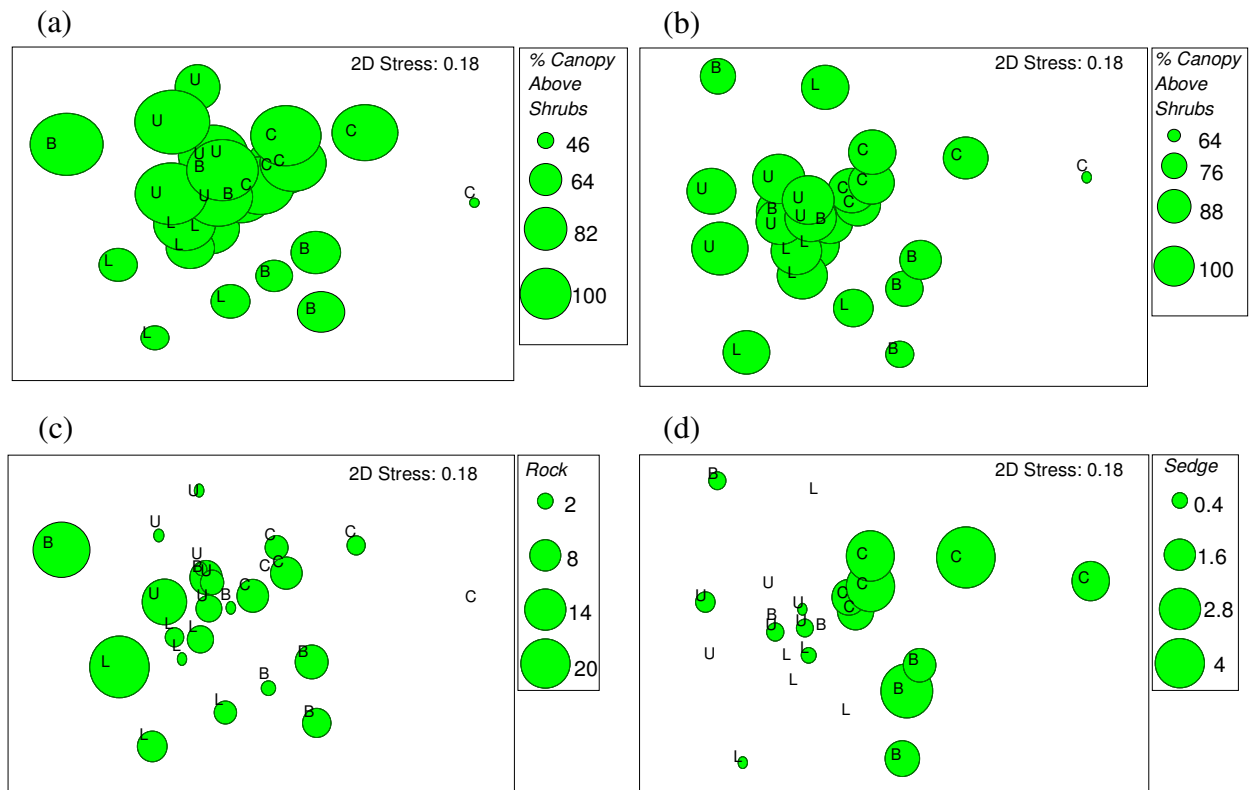
**Table 3.11.** BIOENV results indicating the habitat attributes that best explain the habitat structure at the 24 study sites (where \* indicates significant result).

Maximum number of trial variables	Best correlation	Correlation value	Significance value
1	Canopy cover above shrubs	0.508	0.04*
2	Canopy cover above shrubs and canopy cover below shrubs	0.628	0.01*
3	Canopy cover above shrubs, canopy cover below shrubs and rock cover	0.716	0.01*
4	Canopy cover above shrubs, canopy cover below shrubs, rock cover and sedge cover	0.789	0.01*

(BIOENV;  $P < 0.05$ ). The four variable model incorporated sedge cover as an indicator of understorey productivity, to explain 78.9 % of the variation (BIOENV;  $P < 0.05$ ).

MDS bubble plots of the significant habitat variables revealed distinct patterns among the sites (Figure 3.17: a-d). Canopy cover above and below shrubs display similar patterns, where both plots display sites of lower covers to the bottom of the plot, mainly lantana and burnt sites (Figure 3.17: a-b), showing at these sites there is more light available for reptiles to utilise. The sites with higher canopy cover above shrubs were mainly undisturbed forest and cleared sites and the sites with higher canopy cover below shrubs were mainly undisturbed, lantana and burnt sites.

The rock cover MDS is showing a pattern of decreasing cover from left to right on the MDS across the array of sites (Figure 3.17: c) and sedge cover is showing an opposite trend from right to left with high values occurring mainly in manually cleared sites (Figure 3.17: d). This figure shows that the sites with low rock cover had higher sedge cover. The low stress of 0.18 and distinct patterns suggest that these attributes are



**Figure 3.17.** Two-dimensional MDS plots of the habitat attributes explaining the variation in the 24 study sites, where circle size is indicating the value of the attribute, including (a) canopy cover above shrubs, (b) canopy cover below shrubs, (c) rock cover and (d) sedge cover. Each point represents a separate site, labelled by treatment (L = lantana, B = burnt, C = cleared, U = undisturbed).

significantly related to the differences in habitat structure of the sites and likely influencing other variables as well as reptile communities.

### 3.4.2 Reptile associations with habitat attributes

The habitat attributes found to best explain reptile composition was similar between the method types (Table 3.12). The variables *palm frond litter* and *percentage of silt in soil* appeared in both of the correlations. Both of these variables had higher values in the undisturbed forest sites, although non-significant for palm frond litter (see Section 3.1.3

and 3.4.1), suggesting an effect of this habitat type. Pitfalling trapping reptile composition was additionally correlated with *shrub cover*. Correlations were slightly stronger when explaining the data from pitfall trapping, explaining 54.5% of the variation in the reptile data, and both correlations were significant (BIOENV;  $P < 0.01$ ) (Table 3.12).

**Table 3.12.** Results of BIOENV analyses indicating the habitat variables that are best correlated with reptile composition over the five survey periods (where \* indicates significant result).

Data type	Best correlation	Correlation value	Significance value
Pitfall trap reptile composition (log-transformed)	Palm frond litter, silt content in soil, shrub cover	0.545	0.001*
Time-constrained search reptile composition (log-transformed)	Palm frond litter, silt content in soil	0.511	0.001*

### 3.4.3 Weather effects

Variable responses of reptile composition to weather parameters (including rainfall, cloud cover, maximum and minimum temperature) were found with low correlations and high variation. These effects were masked by other habitat structure effects and weather was not considered in further detail.

### 3.4.4 Reptile and invertebrate correlations

The composition of invertebrates found in the pitfall traps was not significantly affected by the presence of vertebrates within the traps (ANOSIM;  $P > 0.05$ ). Similar outcomes were reached when comparing the invertebrate matrix to the presence of different vertebrate groups (reptile, mammal or anuran) or vertebrate presence/absence. As a result of this, it was assumed that vertebrates were not affecting the invertebrate composition or

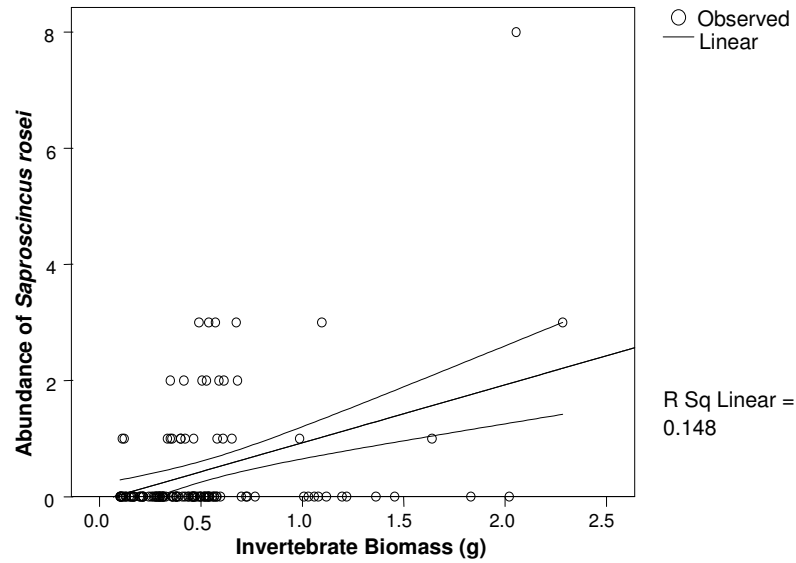
preying on invertebrates in the traps and therefore the invertebrate composition from all pitfall traps (regardless of vertebrate presence) were used in subsequent analyses.

Invertebrate groups found to be the most influential on reptile composition were variable between reptile survey methods. The correlation values found for combinations of invertebrate groups explained a low proportion of the variation in the pitfall trap reptile data (28.8%), but was much higher for time-constrained searches, explaining 46.5% (Table 3.13). These were both significant correlations between invertebrate and reptile composition (BIOENV;  $P < 0.05$ ). Spiders and pillbugs appeared in both of the correlations with reptile composition.

**Table 3.13.** Results of BIOENV analyses indicating the invertebrate groups which are best correlated with reptile composition (where \* indicates significant result).

Data type	Best correlation	Correlation value	Significance value
Pitfall trap reptile composition (log-transformed)	Spider, pillbug, earwig, centipede, other	0.288	0.001*
Time-constrained search reptile composition (log-transformed)	Spider, cricket, pillbug, bug	0.465	0.012*

Invertebrate biomass was not significantly correlated with reptile species richness or abundance from both survey methods (linear regression;  $P > 0.05$ ). However, it was significantly positively correlated with one species (*Saproscincus rosei*) (linear regression;  $P < 0.001$ ). The correlation had high variation and an outlier point (Figure 3.18).



**Figure 3.18.** Linear regression of abundance of *Saproscincus rosei* by invertebrate biomass with 95% confidence interval.

### 3.4.5 Invertebrate associations with habitat attributes

The habitat variables that best explain the invertebrate composition (log-transformed) in a significant correlation were *rock cover*, *coarse woody debris presence*, *light availability above shrubs (2 m)*, *palm frond litter* and *sedge cover* (BIOENV;  $P < 0.01$ ). This combination of habitat attributes explained 60.3% of the variation in the invertebrate data. Due to the high proportion of variation that was explained, it is clear that invertebrates require presence of plants and available light and microhabitats for high abundances, but in general occur across the site regardless of prior or ongoing treatment regimes.

## 4 Discussion

### 4.1 Reptile communities

The faunal communities present at Curramore Sanctuary have not been rigorously investigated previously. A total of 15 species of reptiles was recorded in this study using a combination of two survey methods: pitfall trapping and time-constrained searches. By comparison, a similar study in subtropical rainforests found slightly more species (19) (Kanowski *et al.* 2006) with a similar composition of reptiles. The species at Curramore and those found by Kanowski *et al.* (2006) in south-east Queensland mainly comprised skinks (Scincidae), however both also included members of three other reptile families (Elapidae, Pythonidae and Varanidae). The assemblage recorded by Kanowski *et al.* (2006) was also dominated by *Lampropholis* species.

A list of potential species based on species occurring in the surrounding Connondale and Blackall Ranges (AWC n.d.), with additional records from Eyre *et al.* (1998) and EPA (2008b) (within a 20 km radius of Curramore), listed 29 potential species that were not recorded in this study possibly due to habitat preference (Borsboom *et al.* 2002) (Appendix B), however this research recorded one additional species, *Ramphotyphlops proximus* (a blind snake).

The inability to record a considerable number of potential reptiles from AWC (n.d.) despite substantial survey effort could mean that they do not occur at this Sanctuary. The “potential” species not observed may have also occurred in habitats at Curramore not surveyed in this study. A number of species that were not recorded use margins of waterways, woodlands, dry sclerophyll, brigalow, rainforests and rock outcrops as their

primary habitat, and these types of ecosystems were not extensively surveyed during this study, potentially explaining why these species were not encountered.

Twenty-three of the potentially occurring species at Curramore based on the three regional data sets are associated with wet-sclerophyll or similar habitat types (Wilson 2005), and sixteen of these species were recorded. The “missing” species may be cryptic species or occur in very low numbers at Curramore and could therefore represent false absences of species. The detectability of many reptiles (eg. fossorial skinks, blind snakes and many other snakes) is low throughout the year (Eyre *et al.* 1998; Borsboom *et al.* 2002) and these species may require a considerably higher survey effort for detection (Ryan *et al.* 2002; Roughton and Seddon 2006).

The most abundant species, mainly from the genus *Lampropholis* (sun skinks), including *L. couperi* (Couper’s skink), *L. delicata* (eastern grass skink) and *L. adonis*, explained the greatest amount of variation in the reptile composition data. These species were the most commonly occurring individuals at Curramore and created differences in the reptile assemblages across the sites. These species’ preference for cleared areas and forest edges (Wilson 2005; Kanowski *et al.* 2006) supports their abundance at the treatment and lantana sites given the suitability of these disturbed habitats. The management undertaken at Curramore, particularly at the manually cleared sites, also favours those species that prefer open areas and this is reflected in the high abundance of *Lampropholis* species in these cleared areas.

Interestingly, another species considered to be rare in Queensland was also found to be important in the reptile assemblages. *Saproscincus rosei* was part of the two-species

model explaining 86.4% of the variation in time-constrained search composition. This species has a restricted range occurring in rainforests and adjacent wet-sclerophyll forests (Wilson 2005), however in this study was encountered in high abundances.

Past studies that have assessed reptile communities in similar habitats have found that *L. delicata* is highly dominating in the community structure, occurring at much greater abundances than all other reptile species including *L. couperi* (McFarland 1998; Borsboom et al. 2002; Kanowski et al. 2006). In a fauna study undertaken in south-east Queensland by McFarland (1998) however, *S. rosei* was not found to be a dominant species and was in the bottom 50% of most commonly recorded species. The trends observed with *Lampropholis* species in this study are consistent with past research, however *S. rosei* was found to be more common than in other areas and may therefore have a more abundant, thriving population at Curramore compared to other regions. The occurrence of *S. rosei* across the Sanctuary in all treatment types may be explained by the low microhabitat requirements of *Saproscincus* species which only necessitate litter (Kanowski et al. 2006).

Two larger species were also significant in the reptile matrices, *Eulamprus murrayi* (Murray's skink) as the fourth most important species for pitfall trapping and *Varanus varius* (lace monitor) as the third most important species for time-constrained searches. *E. murrayi* is typically found in subtropical forests preferring rotting logs and crevices as microhabitat (Greer 2005; Wilson 2005), and has also been recorded in average abundances in other studies in south-east Queensland (McFarland 1998; Kanowski et al. 2006). At Curramore, this species was only recorded by pitfall trapping at five sites of

the lantana and undisturbed controls (it was also observed at a manually cleared site in a log during a time-constrained search). Kanowski *et al.* (2006) found that the abundances of *E. murrayi* varied significantly between study sites and was only observed in forest reference sites and not in any disturbed sites (ie. plantation and regrowth) due to its microhabitat requirements. This was also the case at Curramore, with the species utilising undisturbed forest, however was also present in a number of lantana sites. The lantana may have provided a structural microhabitat preferred by *E. murrayi*, with enclosed crevices under the plant and leaf litter, as well as a number of logs occurring within the lantana.

*V. varius* occurs in a variety of well-timbered areas in eastern Australia (Wilson 2005), and is often recorded in south-east Queensland (McFarland 1998; Kanowski *et al.* 2006), and was therefore expected to be found across Curramore Sanctuary. This species was not found in lantana sites, and the significance of this will be discussed in section 4.1.2.

A number of species (*Cyclodomorphus gerrardii* – pink-tongued skink, *Eroticoscincus graciloides* – elf skink and *Eulamprus tenuis* – bar-sided skink) were encountered in low numbers during the survey periods while many snakes (*Cryptophis nigrescens* – small-eyed snake, *Demansia psammophis* – yellow-faced whipsnake, *Tropidechis carinatus* – rough-scaled snake and *Ramphotyphlops proximus* – a blind snake) were only seen once during the entire survey period. Snakes often have low trapability or detection (Eyre *et al.* 1998; Borsboom *et al.* 2002) and this is highlighted in the current study. The burrowing blind snake, *R. proximus*, is very cryptic and therefore may have needed an increased survey effort to increase the species' capture rate. Increasing the survey effort

and areas surveyed for future studies may allow for increasing the number of observations of these snakes and other species (Ryan *et al.* 2002; Roughton and Seddon 2006).

Overall, the larger skinks were found in low numbers. These larger species of skinks are likely to have larger home ranges than the smaller, more locally abundant skinks (Turner *et al.* 1969; Schoener and Schoener 1982) (ie. *Lampropholis* species). The larger species therefore may have a lower encounter rate due to this, particularly with the use of a small survey grid at each site.

The larger body sizes of these skinks may have meant their activity is highly weather dependent, requiring plenty of light for basking (Roughton and Seddon 2006). This effect has been noted for the detectability of a large skink *Oligosoma ottagense* (the Otago skink) which only emerges under optimal conditions (Roughton and Seddon 2006). This may be a similar trend for the larger skinks in this study, which may have limited their emergence due to the variable weather conditions, particularly because the larger skinks more commonly occurred in the undisturbed forest where less light penetrated.

Some reptile species were not observed during the survey periods, only being noted opportunistically. This included two snake species (*Dendrelaphis punctulata* – common tree snake and *Pseudechis porphyriacus* – red-bellied black snake) and a legless skink (*Ophioscincus truncatus*). Both these snakes are commonly occurring in south-east Queensland, however *P. porphyriacus* are more likely to occur by riverbanks and swamps (Wilson 2005), and these habitats were not surveyed during this study. Consequently, *P. porphyriacus* was only opportunistically sighted once, however *D.*

*punctulata* individuals were seen a number of times across the site. It is unknown why these species were not observed during the survey times as they are common in a variety of habitats (Wilson 2005). All observations of *D. punctulata* occurred on tracks at Curramore, and therefore the high observations outside the survey periods may have been attributable to this, as the species were utilising the increased sunlight along the tracks and were therefore more active within these areas compared to the trap sites.

*O. truncatus* was only caught once during the pilot study for this study. This species is a cryptic, burrowing skink (Wilson 2005) and it is therefore expected that low captures of this species will occur.

#### **4.1.1 The effect of sampling type**

The two survey methods used in the study found different compositions of reptiles, with a Jaccard Index of 47% (see Section 3.1.1). Although the time-constrained searches were more efficient and encountered much higher abundances of reptiles than the use of pitfall traps, it is likely to be more effective to use a combination of methods, in order to maximise the species surveyed from within the whole reptile community (Garden *et al.* 2007b). While time-constrained searches were found to be more efficient, they are also known to under-represent the occurrence of certain species, particularly on overcast or cool days (Singh *et al.* 2002). Individuals that are not active and basking during the survey period will be well hidden and would appear to be absent from the site (Roughton and Seddon 2006). This may have confounded the results, however the multiple survey periods coupled with two survey methods reduced this effect as much as practicable. In addition, survey effort was similar between treatment types.

It is known that different survey types target certain species more effectively (Roughton and Seddon 2006; Thompson and Thompson 2007), and this study confirmed this trend. Two species caught only in pitfall traps (*Erotoscincus graciloides* and *R. proximus*) are considered very secretive (Wilson 2005), suggesting that it is unlikely that these species would be observed during a search. This highlights the importance of pitfall trapping for capturing certain cryptic species (Thompson and Thompson 2007).

Those species encountered only during time-constrained searches were three species of snake (*Demansia psammophis*, *Tropidechis carinatus* and *Morelia spilota*) and the lace monitor (*Varanus varius*), all larger reptiles that are known to be undersampled by pitfall trapping as it is unlikely for them to be caught (Thompson and Thompson 2007). It is therefore important to utilise time-constrained searches in order to account for larger species such as these (Thompson and Thompson 2007).

The low survey efficiency in this study could have lead to false absences of reptiles occurring which may have confounded the final results, and may have resulted in high variation in the patterns seen within the results. False absences can occur when there is limited sampling effort, individuals are difficult to sample or the population is of small size (Gu and Swihart 2004). This can impact on the statistical power of tests through limiting the number of captures (Gu and Swihart 2004; Garden *et al.* 2007a). However, in this study, a large number of sites were surveyed over five survey periods, which is considered to be an optimal number of surveys for detecting reptile occupancy (Roughton and Seddon 2006).

#### **4.1.2 The effect of treatment type**

The effect of treatment was quite variable between survey methods, where significant results were consistent with higher captures and observations of reptiles. Pitfall trap data did not show any significant differences of reptile composition among treatment types. This is likely due to the low capture rates that occurred, and therefore there was likely not enough statistical power to allow for robust analyses (Gu and Swihart 2004; Garden *et al.* 2007a). Significant differences were found for time-constrained search reptile composition among treatments and higher abundances of reptiles were observed by this method.

##### *4.1.2.1 Trends within the manually cleared sites*

Reptile abundances from time-constrained searches were highest in manually cleared sites, mainly consisting of common species, such as *Lampropholis* species. These patterns may have been associated with the change in habitat caused by the manual clearing of these sites. It was seen that understorey vegetative cover was significantly reduced after lantana removal by this method. This lead to an increased amount of sunlight available at ground level, and increases in utilisation for reptile basking. *Lampropholis* species are known to prefer cleared and disturbed sites (Greer 2005; Wilson 2005; Kanowski *et al.* 2006) and therefore they may have been attracted to these manually cleared areas. A number of mixed-habitat species, including *Lampropholis*, have been shown to prefer areas in forests with open canopy gaps in a number of regions around the world in order to utilise the increased sunlight for basking (Brown and Nelson 1993; Greenberg 2001; Leynaud and Bucher 2005; Kanowski *et al.* 2006)

The low understorey cover may have also caused the exclusion of habitat specialists that prefer a more closed or complex habitat structure (Kanowski *et al.* 2006). A number of reptiles are reliant on shrub complexity and cover, as shown in a range of past studies (Hadden and Westbrooke 1996; Brown 2001; Singh *et al.* 2002; Jellinek *et al.* 2004; Letnic *et al.* 2004; Fischer and Lindenmayer 2005; Fischer *et al.* 2005) and this study supports these findings where cover was an important habitat variable. The importance of vegetation cover is important to note when utilising a variety of management strategies, as the use of manual clearing and herbicide may cause impacts on reptile assemblages.

A number of habitat attributes appeared to be different in manually cleared sites compared with other treatments, where the difference in location of these treatment sites lead to a different aspect, altitude, clay content in soil and soil colour to the other sites. Although none of these variables were correlated with reptile composition, they may have influenced the high abundances at manually cleared sites indirectly, particularly aspect or altitude. The difference in aspect may have affected the sunlight availability for reptiles for basking sites. Also, a study by Fischer and Lindenmayer (2005) suggests that reptiles are highly sensitive to elevation and changes as small as 50 m can alter the composition of reptiles.

Detection probability of reptiles within each treatment may have influenced the detected abundances of reptiles. With less understorey cover present, there may have been increased visibility at manually cleared sites. This may have influenced the results by allowing observers to more easily locate and identify reptiles at the manually cleared sites. This notion is supported by the survey efficiency, which was highest for time-

constrained searches in manually cleared sites. Detection probability is an issue that affects population estimates of several vertebrate fauna groups due to a number of factors such as conspicuousness of species, space or area surveyed including microhabitats in the survey area, stochastic weather fluctuations, observer bias and method type, where these effects have been shown for birds (Nichols *et al.* 2000; Conway and Simon 2003), anurans (Frei and Schär 2001; MacKenzie *et al.* 2002) and mammals (MacKenzie and Kendall 2002). Other studies have been affected by differing detection probabilities of certain reptile species between different vegetation forms, where reptiles are more likely encountered on bare ground or logs (Brown and Nelson 1993; Cunningham *et al.* 2007).

#### *4.1.2.2 Trends within the lantana-infested sites*

The high abundances of reptiles that occurred at the lantana sites were mostly the common species such as *Lampropholis*. As mentioned earlier, these species are widespread generalists which are able to utilise disturbed habitats effectively (Wilson 2005; Kanowski *et al.* 2006). These species may have been attracted to the lantana thickets due to the low canopy cover (2 m), as lantana thrives in canopy gaps with increased sunlight (Fensham *et al.* 1994; Swarbrick *et al.* 1995; Gentle and Duggin 1998; Day *et al.* 2003b; DECC 2008), which was shown in this study by the comparison of canopy cover among the treatment types. This may also confer an increase in sunlight for use by basking reptiles, either in small gaps in the lantana or for reptiles to climb the plants.

Climbing behaviour has been noted in a number of reptile studies (Chapple and Swain 2002; Irschick *et al.* 2005; Downes and Hoefer 2007; Goodman and Isaac 2008), and a

study by Downes and Hoefer (2007) experimentally compared the climbing of reptiles in weed infested and non-weed infested treatments. Climbing behaviours of *L. delicata* (eastern grass skink), found in high abundances in this study, was substantially greater in areas heavily infested with blue periwinkle (*Vinca major*), a dense, sprawling plant much like lantana. This was shown to be a phenotypic response, where skinks with longer legs and lighter body mass were recorded in the weedy sites, which resulted in a shift in basking strategies so skinks climbed the plants to reach sunlight (Downes and Hoefer 2007). This response to dense weed infestations may also be occurring at Curramore Sanctuary, allowing for small skink species (such as *Lampropholis* species) to survive within the dense lantana patches.

A total of six species in this study were not found in lantana sites. Four of these species were only observed once during this study (*Cryptophis nigriscens*, *Demansia psammophis*, *Tropidechis carinatus*, and *Ramphotyphlops proximus*) and therefore their absence from lantana patches may have only been due to random variation in their occurrences. The other two species (*Eulamprus tenuis* and *Varanus varius*), however, were observed in higher numbers at all treatments except lantana.

*E. tenuis* is a large rainforest skink, however is often seen within disturbed habitats, including gardens (Wilson 2005). It was therefore expected that this species would be observed within lantana infested sites. However, at Curramore this species does not prefer the lantana sites. *Eulamprus* species are usually associated with specific microhabitats in forests, including diverse refuges such as tree hollows and strangler figs (Kanowski *et al.* 2006) and *E. tenuis* may require similar microhabitats at Curramore.

These types of microhabitats do not occur within the lantana and this demonstrates that lantana may block out habitat specialist species that do not prefer the habitat characteristics of lantana patches.

*V. varius* was also not found in lantana and was likely excluded from these areas due to its large size. The density of lantana is likely to impede movement of these species, an effect also found by DECC (2008) for *Varanus* species. Lantana therefore may also be causing the exclusion of a number of larger species of reptiles, particularly monitors. This finding paired with the likelihood that lantana may block out habitat specialists is a significant issue associated with the widespread occurrence of lantana and its extending range into a number of forest ecosystems.

The presence or absence of certain species of reptiles in lantana may have been related to thermal factors. The thermal habitat heterogeneity over small areas (tens of metres) can influence the use of areas by reptiles (Fischer and Lindenmayer 2005). This has been shown for groundcover in subtropical Australia (Fischer and Lindenmayer 2005), and may also be the same for lantana patches. This is because they are highly dense and block out sunlight (Totland *et al.* 2005), and may therefore create different thermal characteristics that could be more or less suited to certain species.

It is important to note that the rare species in this study (*Erotoscincus graciloides* and *Saproscincus rosei*), as listed in the Queensland Nature Conservation Act (1992), were utilising the lantana infested sites; *S. rosei* was even noted to be occurring in lantana at higher abundances than in other sites. DECC (2008) has also found positive effects of lantana on *S. rosei*. This species may prefer the dense understorey structure or open

upper canopy of these habitats, for example. This may be because the understorey can allow for a greater variety and availability of microhabitats for use by the species (Hadden and Westbrooke 1996) and may also be a useful structure for predator avoidance behaviour, where skinks with readily available cover are better able to evade predators such as birds (Smith 1997). *Saproscincus* species are also known to prefer similar habitats as other common *Lampropholis* skinks, favouring sites within open canopy gaps (Kanowski *et al.* 2006). This may be due to similar reasons discussed above for *Lampropholis*. The use of lantana by *S. rosei* is important to note for the conservation of rare species, particularly when undertaking management of lantana and clearing potentially important habitats for the species.

Past studies by DECC (2008), however, found that *E. graciloides* is under high-priority threat from lantana. This species was only found in the current study by a few observations occurring within different treatment types. The low occurrences may have lead to trends not being detected with the preference of habitats for this skink. The species was still found to utilise the lantana, however, suggesting that these weed infested areas do not exclude the species at Curramore.

The sunlight availability at lantana sites may have been affected by the methodology of setting the pitfall arrays. Within each lantana patch, a thin, cleared track needed to be created to set the traps and as a result, created an open canopy. This may have influenced the results, as the access effectively created a strip, which may have caused an increase in the occurrence and activity of reptiles along the strip. As these tracks were also the location of transects used for conducting the time-constrained searches, results may have

been biased towards higher numbers of reptiles within the lantana sites. The creation of the tracks, however, could not be avoided due to the difficulty in accessing areas within lantana patches. Further, these effects were possibly reduced after a short period of time when lantana regrowth occurred over the track. Due to the relatively large areas covered by the lantana patches, the species attracted to the transects were likely to be species already using the lantana. The tracks also allowed for visibility into the lantana while conducting time-constrained searches and therefore it is believed that the tracks allowed for an effective quantification of reptile communities within the lantana sites.

Although supporting a number of reptile species, including common species (*Lampropholis*) in high abundances and rare species, the results suggest that lantana causes the exclusion of certain reptiles, particularly large individuals and habitat specialists. This needs to be taken into account when undertaking management of lantana in order to limit negative effects on particular species.

#### *4.1.2.3 Trends within the burnt sites*

A high species richness of reptiles was encountered in the burnt treatments; higher richness than that of undisturbed forest. After the use of prescribed fire, a rapid regrowth of native vegetation occurred (compared to manually cleared sites) shown in descriptive analyses and comparison of understorey vegetation measurements. This is because fire can assist in germinating the seed-bank and resprouting species (Bell *et al.* 2008; Keeley 2008), and in particular, in eucalypt forests, fire releases a high amount of plant nutrients from vegetation and litter that increases the soil nutrients and encourages the regeneration of vegetation (Briese 1996). This regrowth of flora increased the structural

diversity of understorey vegetation and therefore heterogeneity of the site. The heterogeneity of these habitats is likely to be accelerated by the combination of herbicide, which inhibits the regrowth of lantana (Day *et al.* 2003b; NHT 2003), with the use of fire, that encourages the recovery of native species (Briese 1996; Bell *et al.* 2008; Keeley 2008).

The comparison of habitat structure among the treatments showed that burnt treatments were more variable in the structural attributes measured compared to all treatments, highlighting the importance of the use of prescribed fire for the management of natural ecosystems and maintaining habitat heterogeneity. Heterogenous environments commonly support a greater variety of reptile species (Brown and Nelson 1993; Brown 2001; Jellinek *et al.* 2004; Fischer *et al.* 2005; Leynaud and Bucher 2005) and this pattern may have lead to the high species richness at burnt sites.

Some studies generally report on the initial reduction of microhabitats available for reptile use after a fire, often causing negative effects (McLeod and Gates 1998; Ford *et al.* 1999; Singh *et al.* 2002; Andersen *et al.* 2005). Other studies, however, have recorded higher diversity and abundances of reptiles after fire (Fenner and Bull 2007). This study was not consistent with a negative effect from the use of a low-intensity planned fire, as more than six months had elapsed between the planned fire and the initial surveys, and fire is quite often a short-term impact (Fenner and Bull 2007). The reptile assemblages may have been related to the successional stage of vegetation after the fire (Letnic *et al.* 2004).

A study on the effects of planned burning for weed management by Valentine and Schwarzkopf (2008), however, found that the impacts on reptile assemblages increased after a longer period of time post-fire, with a lower abundance of some reptile species three years after the fire. Studies on the impacts of fire often indicate a reptile succession as different species prevail when their required habitat presides (Letnic *et al.* 2004; Valentine and Schwarzkopf 2008). The current stage of succession at Curramore may be supporting the high diversity of reptiles seen in this study in the burnt sites, however this might change in the future and would require further monitoring. Due to this, it will be important to use small-scale fires that will allow reptiles to move on to nearby unburned habitat in order to limit any negative effects caused by the fires (Valentine and Schwarzkopf 2008).

#### *4.1.2.4 Trends within the undisturbed forest sites*

A different composition of reptiles was found in the undisturbed forest sites compared to all other treatments. This included low abundances of reptiles, mainly attributable to a lower abundance of common species (*Lampropholis* species). At the undisturbed sites there was, however, a high abundance of large rainforest skinks, such as *Eulamprus* species. This assemblage of reptile species, differing from all the other treatments, may have been caused by a number of habitat variables differing within these sites.

The undisturbed forest was a closed forest habitat with a higher canopy cover and was denser and more structurally diverse compared to the other treatments. This is similar to patterns found in a study in subtropical south-east Queensland by Kanowski *et al.* (2006) that compared rainforest reference sites to a number of other disturbed habitats including

regrowth, where the rainforest sites had a more closed canopy with a high density of a diverse range of vegetation in different size classes. In Kanowski *et al.* (2006), *Eulamprus* species were almost completely restricted to the rainforest sites due to their requirements for tree hollows and strangler figs. This is consistent with the findings in the current study, where *E. tenuis* was absent from all lantana patches, possibly due to its requirement for particular microhabitats that did not occur within lantana.

Furthermore, Kanowski *et al.* (2006) typically recorded *Lampropholis* and *Saproscincus* skinks in lower abundances at rainforest sites due to their preference for canopy openings, consistent with the trends found with these species at Curramore where they were in low abundances at undisturbed sites compared to the other treatments. Although canopy cover was not found to be associated with reptile composition, this is likely to be the main factor affecting the reptile assemblages here. A study by Greenberg (2001) found lower numbers of reptiles in closed canopy old growth forests, compared to sites with large canopy gaps, consistent with the results of the current study. The results from Greenberg (2001) suggest that light availability was the main determining factor affecting the reptile assemblages and this may be a similar trend at Curramore.

In the undisturbed forests, the temperature was also often notably cooler than at other sites due to the lack of sunlight penetration (pers. obs.), which also may have caused low reptile activity, and therefore lower abundance measures. Taking site temperature measurements prior to each sites survey may have been able to better account for this variation, and is important for future studies.

Although the undisturbed forest did not have the highest species richness or abundances, the use of this habitat by certain habitat specialists highlights its importance. The aim of forest management would be to allow for a mosaic of different habitat types to increase the diversity and heterogeneity of Curramore Sanctuary, which will promote a greater number of reptile species. The retention of undisturbed forest is an important requirement to maintain this diversity.

## ***4.2 Determinants of reptile communities***

### **4.2.1 Patterns in habitat attributes**

Habitat attributes were significantly different among all treatments types, suggesting that the habitat differs among the four treatments, which was confirmed by the high amount of particular variables also found to be different among the treatments. These trends were expected due to lantana's long presence at Curramore, which is likely to have caused significant changes to the site. Lantana suppresses native vegetation, transforming the areas in which it occurs (Swarbrick *et al.* 1995; Gentle and Duggin 1997; Hiremath and Sundaram 2005; Sharma *et al.* 2007).

The treatment sites where lantana removal occurred were also likely to have different habitat characteristics due to the nature of the management that took place. The manually cleared sites had lower percentage cover values for most vegetative ground cover variables, except sedge and grass. With the use of manually clearing and herbicide application for weed removal, the ground is often left bare (Swarbrick *et al.* 1995), which was also observed during this study. The burnt sites, however, had higher vegetative ground cover. Although lantana was also removed from these sites, the use of prescribed

fire encouraged the faster regrowth of native vegetation (Briese 1996) than manually cleared sites.

Canopy cover was the most important habitat variable explaining the variation between the 24 study sites, and was also significantly different between treatment types. The undisturbed sites had high canopy cover above and below shrubs, whereas lantana and burnt sites had a high canopy cover below shrubs. As lantana creates large thickets and encroaches into areas from canopy gaps (Duggin and Gentle 1998; Sharma *et al.* 2005; Totland *et al.* 2005), the lantana infested areas were expected to have low canopy cover above shrubs, however a high canopy cover below shrubs due to the density of lantana. As the burnt and manually cleared sites were previously infested with lantana, they were also expected to have a lower canopy cover above shrubs compared with undisturbed sites.

#### **4.2.2 Reptile associations with habitat attributes and weather**

The habitat attributes that occurred in both of the models explaining reptile composition were palm frond litter and silt content in soil. The former correlation may have been linked to the importance of complex ground substrates for utilisation by reptiles (Letnic *et al.* 2004; Garden *et al.* 2007a). Palm frond litter may provide a high number of refuges that reptiles can use and therefore certain reptile species may have occurred more frequently at sites with high palm frond cover. This trend has not been previously studied for palm frond litter in particular; however the importance of the cover and depth of other forms of litter has been shown in a number of studies with reptile species dependent on

these variables (Brown and Nelson 1993; Brown 2001; Jellinek *et al.* 2004; Kanowski *et al.* 2006; Valentine and Schwarzkopf 2008).

The correlation of reptiles with silt content may have represented a preference by reptiles to different soil types, which will most likely be linked to soil compaction (Garden *et al.* 2007a). A number of reptiles dig burrows for refuge and the ability to do this is affected by the compaction of the soil, where these reptiles avoid areas with harder soils (Garden *et al.* 2007a). A study by Garden *et al.* (2007a) also found that reptiles were negatively influenced by increased soil compaction indirectly through the impacts of compact soil on vegetation structure and the subsequent influence on reptiles. This second effect may also be related to the relationship of silt content with vegetation structure at Curramore Sanctuary and this may have also affected the reptile assemblages.

The prevalence of silt can be a representative of higher canopy covers, as rain and water erosion in areas with low canopy cover can remove silt particles, leading to natural closed forest having higher quantities of silt (Islam and Weil 2000). High values of percentage of silt mainly only occurred in the undisturbed forest sites and this may have meant that there were different microhabitats available, light availability or canopy cover within these ecosystems. These factors may better explain differences in reptile composition, particularly canopy cover; however the only significant characteristic for the trend found with different reptile composition in undisturbed sites was a correlation with silt. Silt was closely related to canopy cover above and below shrubs at Curramore, consistent with Islam and Weil (2000), potentially confirming that silt was a surrogate for the canopy cover (as a measure of site productivity) at the undisturbed sites.

The trends detected in reptile composition may have been due to responses to instantaneous environmental variables. Reptile behaviour is affected by temperature and sunlight, where the activity of reptiles is usually correlated with these factors (Stevenson 1985; Read and Moseby 2001). The movement of reptiles is often simply individuals shuttling between sunny and shaded areas in order to maintain homeostasis (Stevenson 1985; Hertz *et al.* 1993). Rapid responses to temperature and sunlight may have been exhibited during this study, which have significant influences on daily capture rates of reptiles (Read and Moseby 2001), and the general site characteristic assessments made may not have shown these trends.

Reptile activity is usually correlated with rainfall, temperature and cloud cover (Read and Moseby 2001). Noteworthy trends with weather were not found during this study, but lower abundances of most reptile species were observed on cooler and cloudy days (pers. obs.). Future studies could incorporate measurements of light penetration, temperature and other factors undertaken prior to each survey, which would lead to more specific interactions occurring with weather variables.

#### ***4.3 Invertebrate communities***

Invertebrate composition and biomass was not affected by the treatment types, however was affected by time. This may have been due to a seasonal effect causing variation over time (Watt *et al.* 1997). A number of factors can influence the abundance and composition of invertebrates, however, including temporal variation due to a combination of factors, spatial variation, parasitoids, predators and herbivore-induced responses in plants (Watt *et al.* 1997).

Although few studies have examined the effects of lantana on ground invertebrates, a study by Toft (2001) on a similar native-suppressing weed in New Zealand, *Tradescantia fluminensis*, found negative impacts caused on beetles and fungus gnats. This effect corresponded with a decrease in vascular plant richness and heterogeneity, an impact that is also caused by the presence of lantana.

It was also expected that the increase in diversity and complexity of shrub and seedling species within the burnt sites would have had an effect on the composition of invertebrates. High shrub diversity is likely to cause increases in the diversity of invertebrates, providing a greater range of food sources for reptiles, as found by Hadden and Westbrooke (1996). With this in mind, however, this trend was not found in this study. This could have been due to the survey being undertaken during an early stage in succession, without significant changes in the invertebrate composition occurring yet. Another possible explanation is that differences in invertebrates were not found across treatments and therefore the communities may be nested across the larger landscape.

Broad classifications of invertebrates were used in this study and therefore it is possible that species specific differences may have occurred among treatments that were not evident at this level of detail. However, for this study, the results suggest that invertebrate communities are not impacted by the different treatment types. This therefore also suggests that invertebrates are not a limiting food resource for reptiles among the sites. This means that the invertebrates were not likely to be causing any of the differences recorded in reptile assemblages among the treatment types.

Still, significant correlations were found with reptiles and invertebrates. The most important invertebrates for reptiles appeared to be spiders and pillbugs. Past studies have found significant correlations with invertebrates, where Hadden and Westbrooke (1996) found reptiles to be significantly correlated with springtails, ants and spiders, and Fischer *et al.* (2004) found relationships with *Carlia tetradactyla* (four-fingered skink) that occurred in areas with high numbers of spiders. Spiders are therefore likely to be an important part of reptiles' diets, consistent with the results of this study. As shown in Hadden and Westbrooke (1996) and Fischer *et al.* (2004), invertebrates important for reptiles are likely to differ between locations, and therefore other invertebrate groups were also found to be significantly correlated with reptiles at Curramore (earwigs, bugs, centipedes, crickets and other).

Significant correlations were found with invertebrate composition and a number of habitat attributes, explaining a high proportion of the variation in the data (60.3%). The significance of light availability below shrubs is consistent with a study by Humphrey *et al.* (1999), which found that insect diversity was correlated with high canopy cover values, which is linked to light availability. The other significant attributes (rock, coarse woody debris, palm frond litter and sedge cover) highlight the use of a heterogeneous ground cover by invertebrates. CWD was expected to be related to invertebrate diversity as it supports a wide variety of fauna, particularly invertebrates (Greenberg 2001; Mac Nally *et al.* 2001; Grove and Meggs 2003). This may be also true for the other site characteristics related to invertebrates.

Rock cover, palm litter and sedge cover may have been correlated with invertebrates

through increasing the structural diversity of the ground layer, which is important for invertebrate diversity (Wettstein and Schmid 1999; Toft 2001). Structural heterogeneity and vegetation complexity is important for increasing the diversity of invertebrate communities (Wettstein and Schmid 1999; Toft 2001; Tews 2004; Schmidt *et al.* 2005), particularly of plants used for food, nesting or shelter (Tews 2004). The findings of this study are therefore supported by past research, with site characteristics that influence on the structure of the habitats being important.

## 5 Conclusions and management implications

It is clear that the composition of reptiles is affected by the presence of lantana. The reptile assemblages between the control lantana and undisturbed sites, as well as at the two lantana treatment sites, were distinct. It has been shown that reptiles, including rare species, are utilising the habitats provided by lantana and this is important for land managers to consider. Lantana, however, may exclude habitat specialists (shown here for *Eulamprus tenuis*) due to changed habitat characteristics, such as an open upper canopy and the lack of particular microhabitats, and can block out larger species such as monitors. Therefore ongoing attempts to control and eradicate lantana in Australia may be important for protecting these kinds of species.

Lantana management strategies significantly influence the habitat structure and therefore likely the habitat quality of an area. Additionally, the outcomes of the two integrated management approaches in question had quite distinct outcomes. This study showed that the combination of herbicide and burning allowed for the eradication of lantana while encouraging the recovery of understorey vegetation, leading to a more heterogenous habitat structure and composition. Likely in response to this, reptiles were more diverse in the burnt treatment sites. This is a significant result as prescribed burning is an integral part of the management of forest reserves in Australia (Briese 1996; Valentine and Schwarzkopf 2008).

By comparison, the integration of herbicide and manual clearing to treat lantana did not have accelerated regrowth of understorey vegetation and therefore ground cover was much lower. Consequently, common species (mainly *Lampropholis*) prevailed at high

abundances in these reptile assemblages. Other habitat specialists may also be excluded from these cleared areas, due to the removal of important microhabitats like understorey vegetation cover, as species richness was low in these sites. This effect will presumably occur until the native vegetation has recovered, however one year after the clearing was undertaken, little regrowth was evident.

Although changes in habitat structure can seemingly explain the effects caused by lantana and its treatment on the reptile assemblages, the habitat attributes found to be important for reptiles at Curramore (palm frond litter and silt content in soil) were not directly related to treatment or control type, however these two variables were higher in the undisturbed forest. Invertebrate composition was also not affected by the site type and is therefore not a limiting factor for reptile assemblages at Curramore. Nonetheless, reptiles were associated with a number of invertebrates likely to be important for food.

### ***5.1 Future recommendations***

The effects caused by treatment strategy and the utilisation of lantana patches as habitat by particular species highlights the importance of utilising a mosaic strategy to lantana management. This is particularly important in the context of reptile conservation as reptiles often have small home ranges and have been shown to exhibit an inability to move on to more suitable habitat after vegetation clearing has taken place (Cogger *et al.* 2007). This issue is exacerbated when the vegetation that is cleared is not part of a larger, continuous patch (Cogger *et al.* 2007). The use of herbicide and prescribed fire was shown to be an ideal method to control the spread of lantana due to the acceleration in regrowth of vegetation, and the future threat from lantana in these areas may be

minimised. The consideration of faunal communities in land management is increasingly important and this study demonstrates the importance of monitoring the outcomes of weed management. Future control measures aimed at reducing the extent of lantana while restoring native vegetation communities must consider the potential implications for faunal communities, particularly reptile communities that are heavily reliant on the microhabitat structure.

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## Appendix A

### 1. Abundance of reptilian species observed in time-constrained searches at each study site pooled over the five survey periods.

Site	Total species	Total abundance	Elapidae			Pythonidae	Scincidae								Varanidae	Un ID
			<i>Demansia psammophis</i>	<i>Hemiaspis signata</i>	<i>Tropidechis carinatus</i>		<i>Morelia spilota</i>	<i>Eulamprus murrayi</i>	<i>E. tenuis</i>	<i>Lampropholis spp.</i>	<i>L. adonis</i>	<i>L. delicata</i>	<i>L. couperi</i>	<i>Saproscincus rosei</i>	<i>Varanus varius</i>	
L1	3	34	0	0	0	3	0	0	0	22	0	4	0	2	0	3
L2	3	20	0	0	0	0	0	2	0	7	0	0	0	8	0	3
L3	2	42	0	0	0	0	0	0	0	31	0	3	0	6	0	2
L4	2	11	0	0	0	0	0	0	0	5	1	0	0	4	0	1
L5	2	32	0	0	0	0	0	0	0	19	0	0	0	10	0	3
L6	3	17	0	0	0	1	0	0	0	11	0	0	0	5	0	0
B1	3	36	0	1	0	0	0	0	0	31	0	3	0	0	1	0
B2	4	32	1	0	0	0	0	0	0	23	0	2	0	1	1	4
B3	3	20	0	0	0	0	0	0	0	16	0	0	0	1	1	2
B4	3	28	0	0	0	0	0	0	0	19	1	1	0	4	0	3
B5	3	7	0	0	1	0	0	0	0	5	0	0	0	0	1	0
B6	1	14	0	0	0	0	0	0	0	10	4	0	0	0	0	0
C1	2	51	0	0	0	0	0	0	0	49	0	0	1	0	1	0
C2	3	25	0	0	0	0	0	1	0	23	0	0	0	1	0	0
C3	1	30	0	0	0	0	0	0	0	28	0	0	0	0	0	2
C4	2	45	0	0	0	0	0	0	0	44	0	0	0	1	0	0
C5	2	26	0	0	0	0	0	0	0	25	0	0	0	1	0	0
C6	1	45	0	0	0	0	0	0	0	44	0	1	0	0	0	0
U1	2	25	0	0	0	0	0	0	0	19	3	0	0	1	0	2
U2	3	12	0	0	0	1	0	0	0	4	0	0	0	6	0	1
U3	3	4	0	0	0	1	1	0	0	2	0	0	0	0	0	0
U4	1	3	0	0	0	0	0	0	0	3	0	0	0	0	0	0
U5	5	8	0	0	0	1	1	1	1	3	0	0	0	2	0	0
U6	4	7	0	0	0	0	0	2	0	2	0	0	0	1	2	0

**2. Abundance of reptilian species caught in pitfall traps at each study site pooled over the five survey periods.**

Site	Total species	Total abundance	Elapidae		Scincidae								Typhlopidae
			<i>Cryptophis nigrescens</i>	<i>Hemiaspis signata</i>	<i>Cyclodomorphus gerrardii</i>	<i>Erotoscincus graciloides</i>	<i>Eulamprus murrayi</i>	<i>E. tenuis</i>	<i>Lampropholis adonis</i>	<i>L. couperi</i>	<i>L. delicata</i>	<i>Saproscincus rosei</i>	<i>Ramphotyphlops proximus</i>
L1	3	12	0	0	0	0	0	0	3	6	3	0	0
L2	3	5	0	0	0	0	3	0	0	1	1	0	0
L3	3	18	0	0	0	0	0	0	2	3	13	0	0
L4	5	5	0	0	0	1	1	0	1	1	0	1	0
L5	6	9	0	1	1	1	0	0	0	2	2	2	0
L6	3	5	0	0	0	1	2	0	0	2	0	0	0
B1	4	7	0	0	1	0	0	0	0	1	3	2	0
B2	5	20	1	0	0	1	0	0	0	9	7	2	0
B3	2	9	0	0	0	0	0	0	0	4	5	0	0
B4	5	18	0	0	0	1	0	0	1	9	2	5	0
B5	2	4	0	0	0	0	0	0	2	2	0	0	0
B6	4	6	0	0	0	0	0	1	1	3	0	1	0
C1	2	2	0	0	0	0	0	0	0	1	1	0	0
C2	3	8	0	2	1	0	0	0	0	5	0	0	0
C3	1	1	0	0	0	1	0	0	0	0	0	0	0
C4	1	6	0	0	0	0	0	0	0	6	0	0	0
C5	4	7	0	0	0	0	0	0	1	3	1	2	0
C6	3	5	0	0	0	0	0	0	1	0	2	2	0
U1	1	4	0	0	0	0	0	0	4	0	0	0	0
U2	4	6	0	0	1	0	0	0	2	2	0	1	0
U3	0	0	0	0	0	0	0	0	0	0	0	0	0
U4	1	2	0	0	0	0	0	2	0	0	0	0	0
U5	1	1	0	0	0	0	1	0	0	0	0	0	0
U6	2	5	0	0	0	0	4	0	0	0	0	0	1

## Appendix B

Potentially occurring reptilian species at Curramore Sanctuary based on species occurring in the Blackall/Conondale Ranges (AWC n.d.) and surrounding areas (including Bellthorpe and Mount Mee) (Eyre *et al.* 1998) with additional species from EPA (2008b) of species recorded within a 20 Km radius of Curramore, including their primary habitat (from Wilson 2005), with species observed during this study noted by method (O = opportunistic, P = pitfall trap and T = time-constrained search).

Family	Common Name	Species	Primary habitat	Method of observation
Agamidae	Eastern water dragon	<i>Physignathus lesueurii</i>	Margins of waterways	-
	Bearded dragon	<i>Pogona barbata</i>	Woodlands, dry sclerophyll	-
	Southern angle-headed dragon	<i>Hypsilurus spinipes</i>	Subtropical rainforests	-
Colubridae	Brown tree snake	<i>Boiga irregularis</i>	Woodlands to forests	-
	Common tree snake	<i>Dendrelaphis punctulata</i>	Woodlands, rainforests	O
	Keelback	<i>Tropidonophis mairii</i>	Well-watered habitats	-
Elapidae	White-crowned snake	<i>Cacophis harrietae</i>	Moist, sheltered sites	-
	Dwarf crowned snake	<i>Cacophis krefftii</i>	Rainforests, moist areas	-
	Golden-crowned snake	<i>Cacophis squamulosus</i>	Well-watered habitats	-
	Small-eyed snake	<i>Cryptophis nigrescens</i>	Forests, rainforests	P
	Yellow-faced whip snake	<i>Demansia psammophis</i>	Variety	T
	Black-bellied swamp snake	<i>Hemiaspis signata</i>	Rainforest, wet-sclerophyll	P, T
	Stephens' banded snake	<i>Hoplocephalus stephensii</i>	Rainforest, wet sclerophyll	-
	Red-bellied black snake	<i>Pseudechis porphyriacus</i>	Riverbanks, rainforest	O
	Rough-scaled snake	<i>Tropidechis carinatus</i>	Wet-sclerophyll, rainforest	T
	Bandy bandy	<i>Vermicella annulata</i>	Rainforest to desert	-
Gekkonidae	Spotted velvet Gecko	<i>Oedura tryoni</i>	Brigalow and coastal woodlands	-
Pythonidae	Spotted python	<i>Antaresia maculosa</i>	Variety	-
	Carpet python	<i>Morelia spilota</i>	Variety	T
Scincidae	Verreaux's skink	<i>Anomalopus verreauxii</i>	Dry sclerophyll	-
	Burrowing skink	<i>Calypotis</i>	Rainforests	-

		<i>lepidorostrum</i>		
	Burrowing skink	<i>Calyptotis scutirostrum</i>	Rainforests	-
	Open litter rainbow skink	<i>Carlia pectoralis</i>	Dry sclerophyll	-
	Three-toed snake- tooth skink	<i>Coeranoscincus reticulatus</i>	Subtropical rainforests	-
	Wall skink	<i>Cryptoblepharus virgatus</i>	Variety	-
	Striped skink	<i>Ctenotus arcanus</i>	Rock outcrops, woodland	-
	Striped skink	<i>Ctenotus robustus</i>	Woodlands, rock outcrops	-
	Pink-tongued skink	<i>Cyclodomorphus gerrardii</i>	Moist-timbered areas	P
	Land mullet	<i>Egernia major</i>	Subtropical rainforests	-
	Elf skink	<i>Erotoscincus graciloides</i>	Wet sclerophyll, rainforest	P
	Martin's skink	<i>Eulamprus martini</i>	Rocky, forested	-
	Murray's skink	<i>Eulamprus murrayi</i>	Rainforests	P, T
	Eastern water skink	<i>Eulamprus quoyii</i>	Margins of waterways	-
	Bar-sided skink	<i>Eulamprus tenuis</i>	Forested, rocky	P, T
		<i>Lampropholis adonis</i>	Rainforest edges, clearings	P, T
	Couper's skink	<i>Lampropholis couperi</i>	Rainforests	P, T
	Eastern grass skink	<i>Lampropholis delicata</i>	Moist areas	P, T
	A limbless skink	<i>Ophioscincus ophioscincus</i>	Wet sclerophyll	-
	Challenger skink	<i>Saproscincus challengeri</i>	Rainforests	-
		<i>Saproscincus rosei</i>	Rainforests	P, T
	Blue-tongued skink	<i>Tiliqua scincoides</i>	Broad range	-
Typhlopidae	Blind snake	<i>Ramphotyphlops nigrescens</i>	Forest, woodland	-
Varanidae	Lace monitor	<i>Varanus varius</i>	Well-timbered areas	T

## Appendix C

**Total biomass of invertebrates and abundance of invertebrate groups caught in pitfall traps at all study sites pooled over the five survey periods.**

Site	Biomass (g)	Ant	Spider	Cricket	Slater	Earwig	Beetle	Tick	Scorpion	Decapod	Bug	Caterpillar	Other	Cockroach	Antlion	Fly	Centi-pede	Mantis
L1	5.174	41	62	25	47	7	58	51	4	253	6	2	8	2	5	4	13	1
L2	5.827	36	45	37	16	5	21	64	0	125	10	1	2	2	1	0	49	0
L3	3.584	95	56	10	9	1	6	0	0	29	2	0	4	0	0	0	11	1
L4	3.943	152	63	9	20	6	28	3	0	39	7	1	3	1	0	0	10	1
L5	5.341	327	64	21	34	5	41	7	2	91	11	1	3	0	0	1	9	0
L6	5.684	421	65	19	31	1	68	19	1	124	13	1	2	8	0	1	12	2
B1	2.495	112	84	39	8	1	22	4	2	25	8	3	0	2	3	0	7	1
B2	3.430	723	81	19	17	4	40	1	20	24	7	2	6	2	0	1	12	1
B3	4.223	561	69	20	12	2	30	2	1	50	16	0	4	1	0	3	6	0
B4	3.929	542	96	19	26	4	51	2	8	51	16	4	2	5	0	4	13	1
B5	5.693	105	48	27	15	2	37	5	5	43	1	1	4	3	1	1	15	0
B6	3.599	350	51	9	14	2	26	0	3	40	15	0	0	1	0	1	4	0
C1	6.551	759	77	13	10	2	32	11	0	33	5	2	8	4	0	3	16	2
C2	4.871	653	64	27	11	7	35	10	1	53	15	4	2	3	0	2	16	0
C3	4.377	372	56	25	19	3	31	13	0	24	7	4	0	2	0	0	3	0
C4	3.216	575	115	9	9	5	24	10	0	15	7	1	4	1	0	3	8	1
C5	4.233	827	135	17	22	12	35	23	0	23	10	3	6	0	0	2	15	1
C6	3.392	154	107	17	21	13	18	3	0	38	14	1	8	2	1	4	23	0
U1	3.940	153	78	16	33	1	31	2	1	24	15	2	0	2	0	7	20	0
U2	5.697	799	56	3	20	4	37	11	4	36	24	1	0	2	0	3	18	1
U3	3.619	339	44	3	11	0	43	2	7	37	17	0	2	2	0	2	7	0
U4	4.449	66	45	21	29	3	34	8	8	163	25	3	14	5	2	2	17	1
U5	2.483	28	13	1	0	0	2	0	6	3	1	0	0	0	0	0	16	0
U6	2.189	8	14	2	0	0	0	0	5	0	0	0	1	0	0	2	22	1

## Appendix D

### 1. Habitat attributes at all study sites (part a).

Site	Aspect	Altitude	Rock % cover	CWD % cover	% Canopy above shrubs	% Canopy below shrubs	Light below shrubs (lux)	Light above shrubs (lux)	Leaf litter % cover
L1	125.0	609.0	0.9	1.3	55.0	78.4	10437.1	22001.1	60.8
L2	175.0	558.0	3.1	12.5	47.8	85.0	605.9	11193.3	63.5
L3	132.5	525.0	11.6	1.6	54.6	86.0	386.8	12088.9	76.9
L4	135.0	542.0	3.3	6.6	77.2	89.1	918.4	13565.6	89.0
L5	75.5	604.0	0.1	3.4	62.8	89.0	464.9	5179.8	77.8
L6	102.0	616.0	0.7	1.2	77.5	89.7	453.2	2670.6	76.2
B1	17.5	583.0	1.0	6.2	64.4	76.4	8206.8	8862.2	76.3
B2	80.0	554.0	0.7	6.2	62.1	69.2	6667.7	5263.3	70.5
B3	60.0	564.0	0.0	6.2	80.5	84.0	1999.8	2714.1	78.7
B4	76.0	586.0	0.6	5.1	53.5	79.9	4464.4	5125.6	56.3
B5	190.0	570.0	0.6	4.3	90.5	90.8	232.2	208.3	81.6
B6	116.0	562.0	10.1	2.4	92.3	91.2	1487.0	5619.3	68.3
C1	210.0	507.0	0.0	2.1	85.9	84.0	3601.1	2874.4	76.0
C2	185.0	468.0	1.8	3.8	84.9	86.1	391.2	467.1	53.2
C3	340.0	468.0	0.0	4.4	40.6	60.5	1087.3	2090.0	7.9
C4	259.5	497.0	1.9	1.6	83.1	84.0	4270.8	1512.2	80.3
C5	320.0	448.0	0.2	2.0	82.5	83.3	13825.6	12025.6	55.6
C6	315.0	484.0	2.4	3.2	89.0	86.0	591.9	618.2	82.5
U1	160.0	527.0	0.1	2.4	87.8	87.4	478.7	621.3	76.6
U2	125.5	584.0	0.0	3.7	86.2	87.5	2862.1	1477.9	72.2
U3	140.0	588.0	0.8	6.0	86.7	90.9	413.1	467.9	82.2
U4	102.5	585.0	6.1	5.7	91.3	92.1	105.4	163.6	44.4
U5	212.5	544.0	0.9	4.3	90.6	91.4	95.9	155.3	84.0
U6	217.5	534.0	1.7	7.6	95.2	96.5	62.4	84.4	76.3

## 2. Habitat attributes at all study sites (part b).

Site	Bark litter % cover	Twig % cover	Palm frond litter % cover	Debris % cover	Trunk % cover	Root % cover	Stump % cover	Seedling % cover	Vine % cover
L1	5.5	4.0	0.0	1.3	0.0	0.0	0.0	0.0	3.1
L2	4.7	12.1	3.3	0.0	0.2	0.2	0.3	0.4	0.9
L3	9.2	6.7	2.0	0.0	0.5	0.0	0.0	0.0	1.9
L4	12.8	7.1	0.3	0.2	0.0	0.0	0.0	1.5	2.6
L5	1.7	2.1	0.0	0.0	0.0	0.0	0.0	0.7	2.5
L6	4.9	3.3	0.0	0.0	0.0	0.0	0.0	0.1	1.5
B1	6.0	12.7	0.2	0.8	1.0	0.0	0.0	0.2	18.2
B2	7.8	7.4	0.0	3.0	0.0	0.0	0.8	3.0	8.6
B3	6.4	10.4	0.3	1.0	2.6	0.5	0.0	5.6	1.8
B4	4.3	7.0	0.0	0.2	0.0	0.0	0.0	16.3	19.1
B5	6.9	10.6	0.0	1.3	1.3	0.2	0.0	0.0	1.2
B6	5.3	5.6	0.0	0.0	8.2	1.9	0.0	6.0	0.3
C1	5.3	5.9	0.0	1.1	2.5	0.0	0.0	0.0	0.0
C2	1.7	7.8	0.1	1.3	0.5	0.1	0.0	2.8	0.5
C3	3.6	12.3	0.1	0.0	0.0	0.0	0.0	9.0	2.4
C4	13.0	8.5	0.0	0.0	0.8	0.1	0.0	5.0	1.4
C5	1.4	9.4	0.0	0.3	0.8	0.0	0.0	4.1	1.8
C6	1.7	6.5	0.0	0.0	2.9	0.0	0.0	0.8	1.9
U1	15.7	14.7	1.1	0.5	2.7	0.0	0.0	0.0	0.3
U2	5.1	2.9	0.0	0.0	5.8	0.0	0.5	2.1	0.3
U3	2.8	2.4	1.3	0.1	0.0	0.0	0.0	2.7	4.0
U4	0.2	5.4	1.0	0.0	0.5	0.2	0.0	3.9	6.2
U5	7.5	7.5	2.2	1.1	0.5	0.0	0.0	0.0	0.0
U6	6.6	11.3	10.1	0.7	1.5	0.2	0.0	0.0	0.4

### 3. Habitat attributes at all study sites (part c).

Site	Fern % cover	Herb % cover	Sedge % cover	Grass % cover	Shrub % cover	Lantana % cover	Soil Colour	% Clay in soil	% Silt in soil
L1	0.3	0.8	0.0	0.0	0.2	40.2	7	20.8	0.0
L2	0.7	0.9	0.0	0.0	0.0	38.8	7	37.5	0.0
L3	0.0	0.0	0.0	0.0	0.0	42.0	2	15.0	0.0
L4	0.7	1.5	0.2	0.0	0.0	22.4	7	37.5	0.0
L5	3.2	0.2	0.0	0.0	0.0	68.4	7	38.3	0.0
L6	0.7	0.6	0.0	0.0	0.2	64.5	7	30.0	8.3
B1	3.4	14.4	2.4	6.0	0.0	0.0	5	31.7	0.0
B2	1.0	12.8	1.1	0.4	1.0	0.0	7	27.5	0.0
B3	10.1	8.0	0.0	4.5	1.1	0.0	7	35.8	0.0
B4	0.4	2.6	0.9	0.2	0.0	0.0	7	29.2	0.0
B5	8.4	8.1	0.0	0.0	0.0	0.8	7	31.7	16.7
B6	4.7	0.4	0.3	0.0	0.0	0.0	6	33.3	0.0
C1	0.0	5.0	1.2	11.6	0.0	0.0	5	24.2	8.3
C2	4.5	2.0	1.1	3.6	0.0	0.0	4	26.7	0.0
C3	3.0	5.9	1.3	14.5	7.6	0.0	5	23.8	0.0
C4	2.3	2.5	2.1	7.6	0.0	0.0	3	21.7	0.0
C5	1.0	4.0	3.1	8.6	0.0	0.0	1	18.3	0.0
C6	6.0	0.9	2.1	3.4	0.1	0.2	1	18.3	0.0
U1	5.1	5.6	0.3	0.3	0.0	0.0	7	40.0	8.3
U2	22.6	0.5	0.4	0.6	0.0	0.1	7	41.7	8.3
U3	4.3	1.5	0.3	0.0	1.3	0.0	7	33.3	8.3
U4	1.3	0.0	0.0	0.0	0.5	0.0	7	37.5	12.5
U5	9.0	7.0	0.0	0.2	0.0	0.0	2	35.8	8.3
U6	1.8	10.3	0.0	0.0	0.0	0.0	7	35.8	8.3

## Appendix E

Two-dimensional MDS plot of reptile composition from pitfall traps including outlier site with no reptile observations showing patterns in *Lampropholis couperi*. Each point represents a separate site.

