

**Comparative Impact of Invasive Alien Trees and Vineyards
on Arthropod Diversity in the Cape Floristic Region,
Western Cape**

by

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Declaration

By submitting this thesis/dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, and that I have not previously in its entirety or in part submitted it for obtaining any qualification.



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GENERAL ABSTRACT

Invasive alien trees (IATs) and agriculture are rapidly altering the capacity of ecosystems to provide a variety of essential services, with decreasing habitat quality having an adverse effect on arthropod biodiversity. Although both agricultural intensification and IATs have an impact on ecosystems, it is unclear which of these two is currently the most severe. Here I compare the influences of vineyards and IATs on arthropod diversity in the Cape Floristic Region (CFR) from two perspectives. Firstly, insect diversity from four different vegetation types (native fynbos, IATs, cleared invasive alien trees (CIATs) and vineyards), and their ecotones were assessed. Secondly, soil compaction in the different vegetation types was assessed and compared to determine how soil compaction relates to arthropod species richness and abundance.

Surface-active arthropods were used for these evaluations, as they were considered to be the most abundant, yet sensitive groups for these comparative assessments. Pitfall trapping, which has been widely used for such comparative studies, was the method of choice. Additionally, as a result of incidental wildfires at some of the study sites (IATs and fynbos vegetation), it was decided opportunistically to assess recovery of arthropod diversity following the impact of fire.

The results categorically indicate that IATs had by far the most significant impact, in comparison with vineyards, in reducing arthropod diversity and changing assemblage composition. Both IATs and vineyards, not surprisingly, supported lower arthropod diversity compared to fynbos, while CIATs supported arthropod species richness comparable to that of fynbos. However, different invertebrate species responded differently to IAT invasion, with some species even adapted to conditions in IAT patches. Environmental factors such as leaf litter and soil compaction were strongly correlated with arthropod species richness. However, other factors, such as soil moisture content, were not correlated with species richness, despite varying significantly between different vegetation types. IATs were found to reduce soil compaction, while vineyards increased soil compaction.

The ecotone between adjacent land-use types was important for sustaining high arthropod species richness, particularly for the CIATs/fynbos ecotone, followed by

the interface between CIATs/vineyard and vineyard/fynbos. Some species even preferred ecotones.

Surface-active invertebrate species richness declined substantially immediately after fire in both IATs and fynbos. Nevertheless, recovery was rapid, with some species recolonizing or re-establishing within three months. Indeed, both species richness and abundance were high following fire. Fire had no significant impact on arthropod assemblage within IATs, despite having a serious impact in the fynbos where a different arthropod assemblage was observed after fire. The recovery of the historic assemblages may be dependent on both the pre-fire composition of the habitat and its adjacent vegetation.

Effective co-operation among conservation biologists, farmers, and landowners to actively plan the future of CFR landscape biodiversity is required.

ALGEMENE OPSOMMING

Indringer uitheemse bome (IUB) en landbou bedrywighede is vinnig besig om die kapasiteit van ekosisteme vir die voorsiening van 'n verskeidenheid van noodsaaklike dienste, met 'n afname in habitatkwaliteit en 'n nadelige uitwerking op artropood biodiversiteit, te verander. Alhoewel beide, landbou intensifikasie en IUB's, 'n impak op ekosisteme uitoefen, is dit onduidelik watter van hierdie twee huidig die mees ernstige een is. In hierdie studie vergelyk ek die invloed van wingerde en IUB's op artropood diversiteit in die Kaapse Floristiese Streek (KFS) vanuit twee perspektiewe. Eerstens, insekdiversiteit van vier verskillende vegetasietipes (natuurlike fynbos, IUB's, verwyderde indringer uitheemse bome (VIUB's) en wingerde) en hulle ekotone is bepaal. Tweedens, is grondverdigting in en van die verskillende vegetasietipes ondersoek en vergelyk om te bepaal hoe grondverdigting verband hou met artropood spesies rykheid en volopheid.

Oppervlak-aktiewe artropode is vir hierdie bepalings gebruik aangesien hulle beskou word as die mees volopste, maar ook mees sensitiewe groepe vir hierdie vergelykende bepalings. Pitvalvangste, wat alom gebruik word vir sulke vergelykende studies, was die verkose werkswyse. Bykomend, as gevolg van onbeplande veldbrande in sommige studie gebiede (IUB's en fynbos plantegroei), is oppertunisties van die geleentheid gebruik gemaak om die herstel van artropood diversiteit na die impak van brand te bepaal.

Die resultate toon duidelik aan dat IUB's, in vergelyking met wingerde, by verre die mees beduidende impak gehad het deur 'n verlaging van artropood diversiteit en veranderende groeperingssamestellings. Beide, IUB's en wingerde, het, nie onverwags nie, in vergelyking met fynbos, laer artropood diversiteit ondersteun, terwyl VIUB's artropood spesies verskeidenheid, vergelykbaar met dié van fynbos, ondersteun het. Verskillende invertebrate spesies het egter verskillend gereageer op IUB indringing, en sommige spesies het selfs by toestand in IUB kolle aangepas. Omgewingsfaktore soos blaarafval en grondverdigting was sterk gekorreleer met artropood spesies rykheid. Ander faktore, soos grondvoginhoud, was egter nie in ooreenstemming met spesies rykheid nie, ten spyte van 'n beduidende wisseling tussen verskillende vegetasietipes. IUB's het grondverdigting verminder, terwyl by wingerde grondverdigting toegeneem het.

Die ekotone tussen aanliggende grondgebruike was belangrik om hoë artropood spesies rykheid, veral by die VIUB's/fynbos ekotoon, gevolg deur die kontaksones tussen VIUB's/wingerd en wingerd/fynbos, te onderhou. Sommige spesies het selfs ekotone verkies.

Oppervlak-aktiewe invertebraat spesies rykheid het onmiddelik na brand in beide, IUB's en fynbos, wesenlik afgeneem. Tog was herstel vinnig, met sekere spesies wat binne drie maande terugkeer of hulle hervestig het. Inderdaad was beide, spesies rykheid en volopheid, hoog na 'n brand. Brand het geen noemenswaardige invloed op artropood groeperings binne IUB's gehad nie, ten spyte van 'n ernstige impak in fynbos waar 'n verskillende artropood groepering na 'n brand waargeneem is. Die herstel van die historiese groeperings van beide, die voor-brand samestelling van die habitat en sy aangrensende plantegroei, mag afhanklik wees.

Effektiewe en aktiewe samewerking tussen bewaringsbioloë, boere, en grondeienaars om die toekoms van KFS landskap biodiversiteit te beplan, word benodig.

The above translation of abstract in Afrikaans was done by Prof. Henk Geertsema, a retired entomologist at Stellenbosch University.

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Chapter 1 – General Introduction

Biodiversity crisis

As human influence on the environment increases, biodiversity conservation becomes more important and urgent (Pimm et al., 1995; Daily, 2000). Biodiversity includes all levels of biological complexity, ranging from subspecies diversity (ecotypes, life cycles, genes, physiology, behaviour), species richness, supraspecies diversity (foodweb interactions, non-trophic relationships, above and below ground relationships) (Wolters, 2001; Kim and Byrne, 2006) to community diversity across landscapes (Sala et al., 2000). Therefore, a decrease in biodiversity may result in reduction of natural life-support processes, including diversity-ecosystem quality relationships (Christensen et al., 1996; Millenium Ecosystem Assessment, 2005). Indeed, global biodiversity is changing at an unprecedented rate because of several anthropogenic impacts (Macdonald, 1989; Ehrlich and Ehrlich, 1992; Sisk et al., 1994; Pimm et al., 1995; Krebs et al., 1999; Ekbom, 2000; Burel et al., 2000; Warren et al., 2001; Robinson and Sutherland, 2002; Turner et al., 2004; Kotiaho et al., 2005; Milder et al., 2007).

When invertebrate populations become isolated in landscape fragments, they are likely to face a higher probability of extinction (Kotze and O'Hara, 2003). The growing list of documented recent extinction rates of insects is an evidence of a global biodiversity crisis (Edwards and Abivardi 1998; Conrad et al., 2006; Worm and Duffy, 2003) and diverse species, not now threatened, will succumb (Pimm et al., 1995; Thuiller, 2007). Agricultural activities and invasive alien plants are important drivers of this biodiversity loss (Pimm et al., 1995; Matson et al., 1997; Thompson and Jones, 1999).

Impact of alien organisms

Biological invasions are second only to habitat loss, usually through conversion of natural vegetation, and are threat to global biodiversity (United Nations Development Programme et al., 2000; Sala et al., 2000; McLaughlin et al., 2002; Thuiller, 2007; Leather et al., 2008). The South African Cape Floristic Region (CFR) is under threat from alien plant invasions and transformation to alternative land uses (Holmes and Richardson, 1999). It is undeniable that with such continuing habitat loss, pressure on the CFR biodiversity will intensify.

Impact of agriculture

Agricultural production, while maintaining biodiversity and ecosystem function, is one of the greatest challenges facing the CFR, if not the earth in general. In terms of landscape disturbance, agricultural lands are among the most modified habitats in the world (Matson et al., 1997), primarily driven by economic (Mitchell, 1991) and political considerations (Ekbom, 2000), with as much as 80% of some region's natural vegetation having been transformed into agricultural land, threatening biodiversity (Tscharntke et al., 2005; Harvey et al., 2008). Agriculture affects terrestrial biodiversity (Tscharntke et al., 2005) by changing the amount of natural habitat present and its spatial configuration (Theobald et al., 1997). Below-ground organisms are highly and directly affected by land use change for agricultural purposes (Duelli et al., 1999).

Other studies indicate that certain heterogeneous agricultural landscapes contribute to biodiversity conservation (Pryke and Samways, 2003; Tscharntke et al., 2005; Harvey et al., 2008) while contributing positively to food production (Daily, 2000). Although trees may also reduce agricultural production through competition for resources, it is important to integrate them in densities and spatial arrangements that will still provide biodiversity benefits (Harvey et al., 2008). In fact, biodiversity friendly farming practices are those that mimic the structural and floristic diversity of native vegetation and those less dependent on agrochemicals (Tscharntke et al., 2005). Since agricultural land use affects large parts of terrestrial area, its contribution to biodiversity is critical for successful conservation in the future (Tscharntke et al., 2005).

Ecologically unsustainable use of ecosystem services raises the potential for serious and irreversible ecological change. In landscapes, such as in and around the CFR, which are influenced by intensive agriculture, it is important to understand the interaction of land use with arthropod assemblage composition for maintaining biodiversity. Up to now, agricultural management and efforts to increase connectivity have been largely independent of each other, which is likely to have had a strong influence on arthropod diversity (Fleishman et al., 1999). Due to intensification of human activities (urban development and modern agriculture) since early 1900s (Figure 1.1 to Figure 1.3), the CFR fynbos vegetation area has declined. Therefore, there is a serious concern for biodiversity conservation in the CFR.

Soil compaction

When soil conditions are favourable, a wide range of arthropod species are normally present (Boughey, 1968; Gordon and Forman, 1983; Holland and Luff, 2004). Invasion with pines, for example may cause changes in soil structure (Watts, 1951; Armstrong and van Hensbergen, 1996; Morris, 1984; Payet et al., 2001; Wolfe et al., 2008), altering soil function. Thus, measurement of level of disturbance to the soil profile is critical, yet often neglected.

Biodiversity conservation

Biodiversity conservation is often in direct conflict with human practices, as human-mediated transformation of ecosystems generally transforms natural habitats and their biodiversity without regard for the immediate and distant future consequences (Kim and Byrne, 2006). Indeed, there is still a long way to go for effective biodiversity conservation (Sisk et al., 1994; Wilcove et al., 1998; Samways, 2005; Innes and Timko, 2007; Leather et al., 2008; Fraser et al., 2008a, 2008b), owing to the limited knowledge about the reaction of biota to rapid habitat changes (May, 1988; Gaston, 1991; Pimm et al., 1995; Kremen and Ostfeld, 2005; Thuiller, 2007).

Worldwide entomological training is in decline, and in a country such as UK for example, it is virtually nonexistent (Leather, 2007; Corbet, 2007). Furthermore, estimates of future extinctions are hampered by the limited understanding of which areas are rich in endemics (Pimm et al., 1995). More often, terrestrial biodiversity is concentrated in biodiversity hotspots that are associated with high losses of vegetation and increased human population density (Sechrest et al., 2002).

Many insect species are experiencing steep declines in both population range and densities (Williams, 1982), despite their importance for conservation being emphasised (Pyle et al., 1981; Wilson, 1987; Franklin, 1993; Kremen et al., 1993; New, 1999; McGeoch, 2002; Cardoso et al., 2008). Where effort has been made, it tends to be focused on rare, attractive and threatened insect species rather than on common and inconspicuous beneficials (Duelli et al., 1999). Less than 40% of the globally existing insects have been named today (Hammond, 1992). Although the estimate of the insect species in a global scale varies highly (Stork, 1988; Gaston, 1991; Alroy, 2002), there are possibly over 4 million (Novotny et al., 2002). Also, while financial support and publicity for the conservation of species such as elephants and lions is readily available

(Wilson, 1987), the situation is somewhat different for insects that make up over 60% of the described global biota (Leather et al., 2008).

Although less charismatic than big animals, the importance of insects cannot be underestimated due to their key ecological processes they facilitate, including pollination, as agents of seed dispersal, herbivory and in decomposition processes, and themselves being food source for a wide range of animals (Curry, 1994). A good understanding of insect requirements is core to a successful insect conservation programme (New, 2007), considering their widely appreciated importance to ecosystem function (Edwards and Abivardi, 1998).

An understanding of spatial distribution of biodiversity is of practical importance, providing knowledge concerning the probability of species extinction due to loss of habitat (Rozenzweig, 1999). However, the efforts and costs associated with collecting, sorting and identifying insects are often too high when compared to those associated with higher plants and birds (Duelli et al., 1999). The available methods and empirical data concerning insect and other arthropod diversity in agricultural landscapes do not yet allow comprehensive biodiversity evaluation (Duelli et al., 1999).

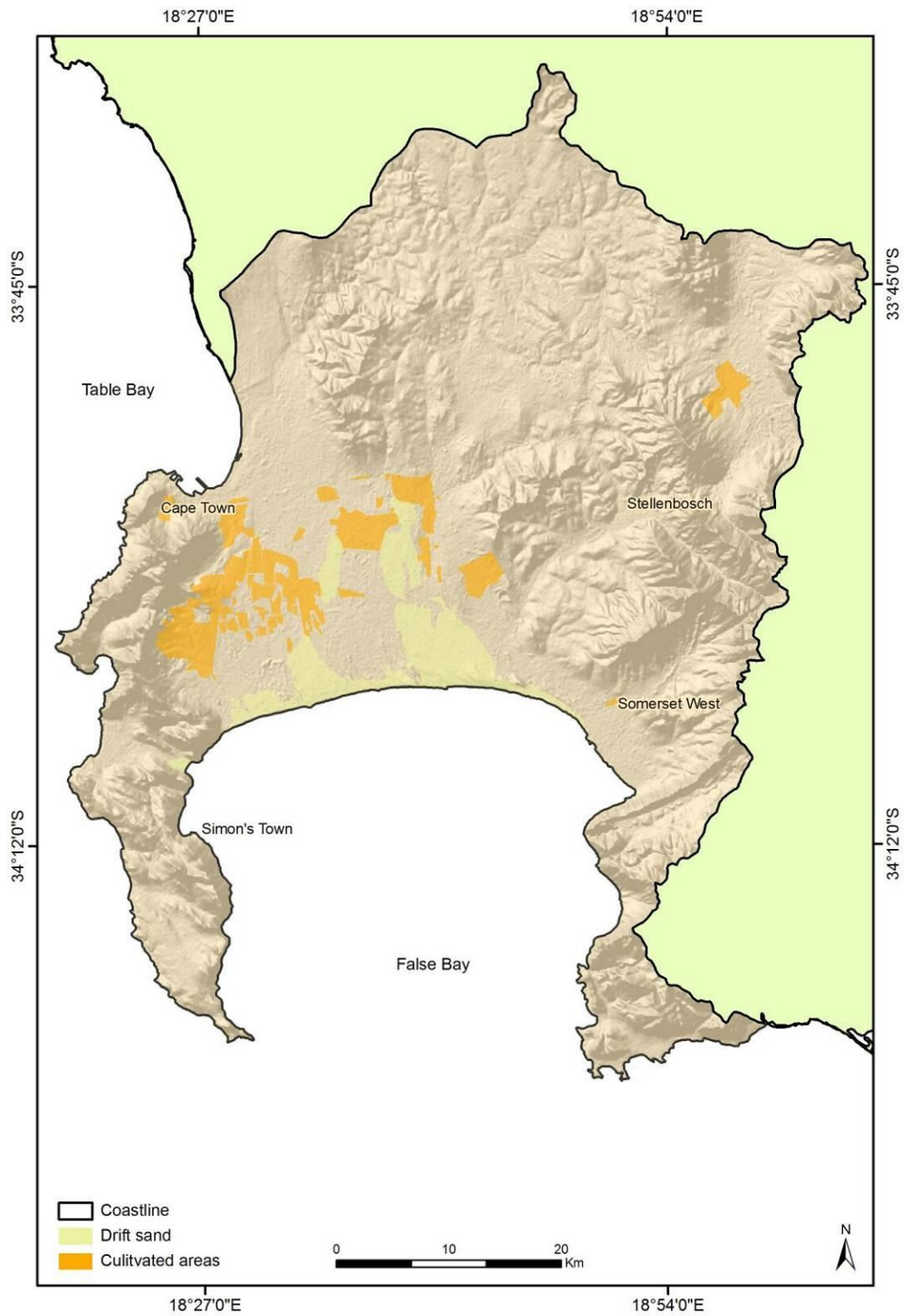


Figure 1.1 Agricultural areas within the study area in the year 1900. (Data: RSA Planning Department).

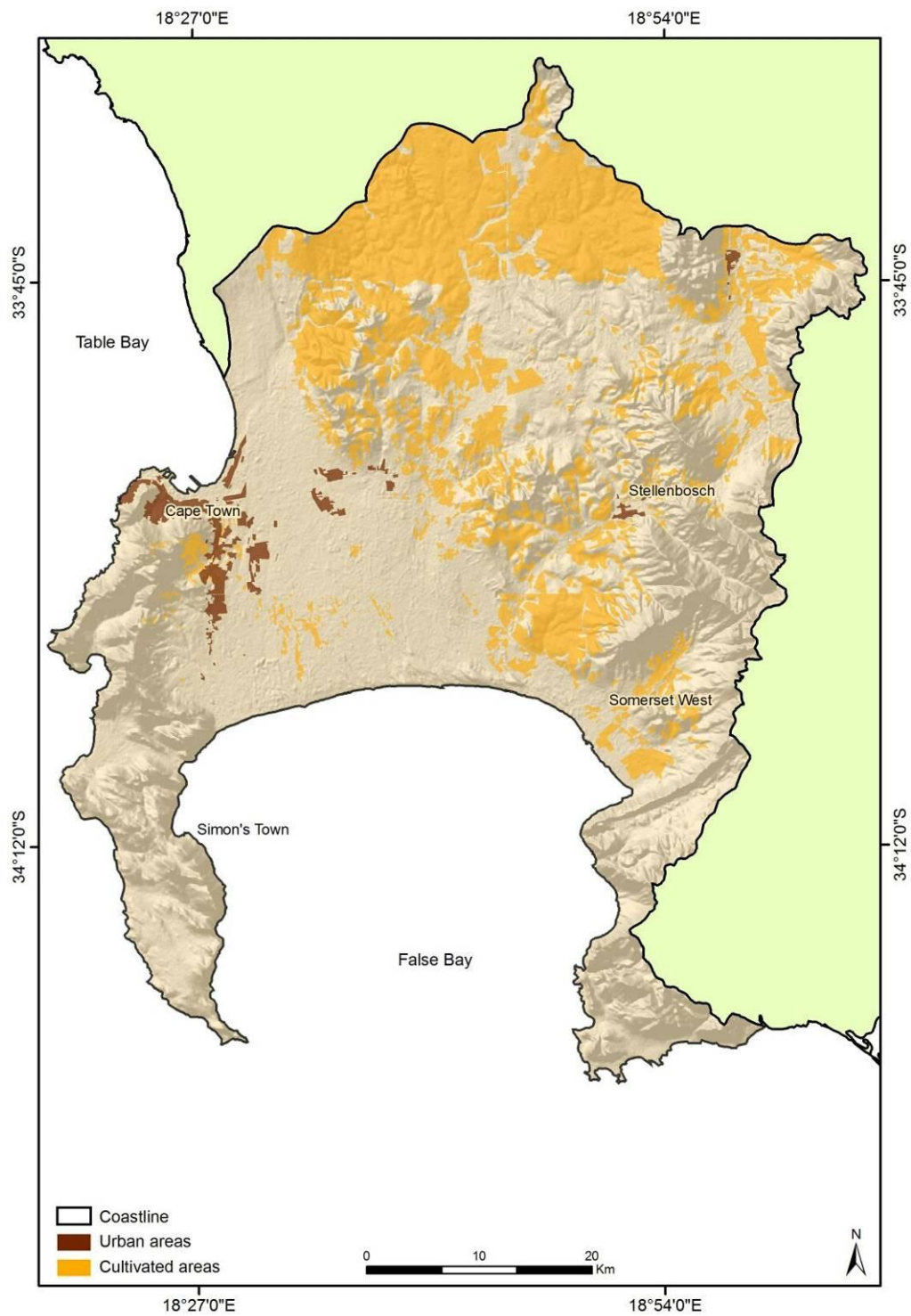


Figure 1.2 Expansion of urban and agricultural areas within the study area by 1942. (Data: RSA Planning Department)

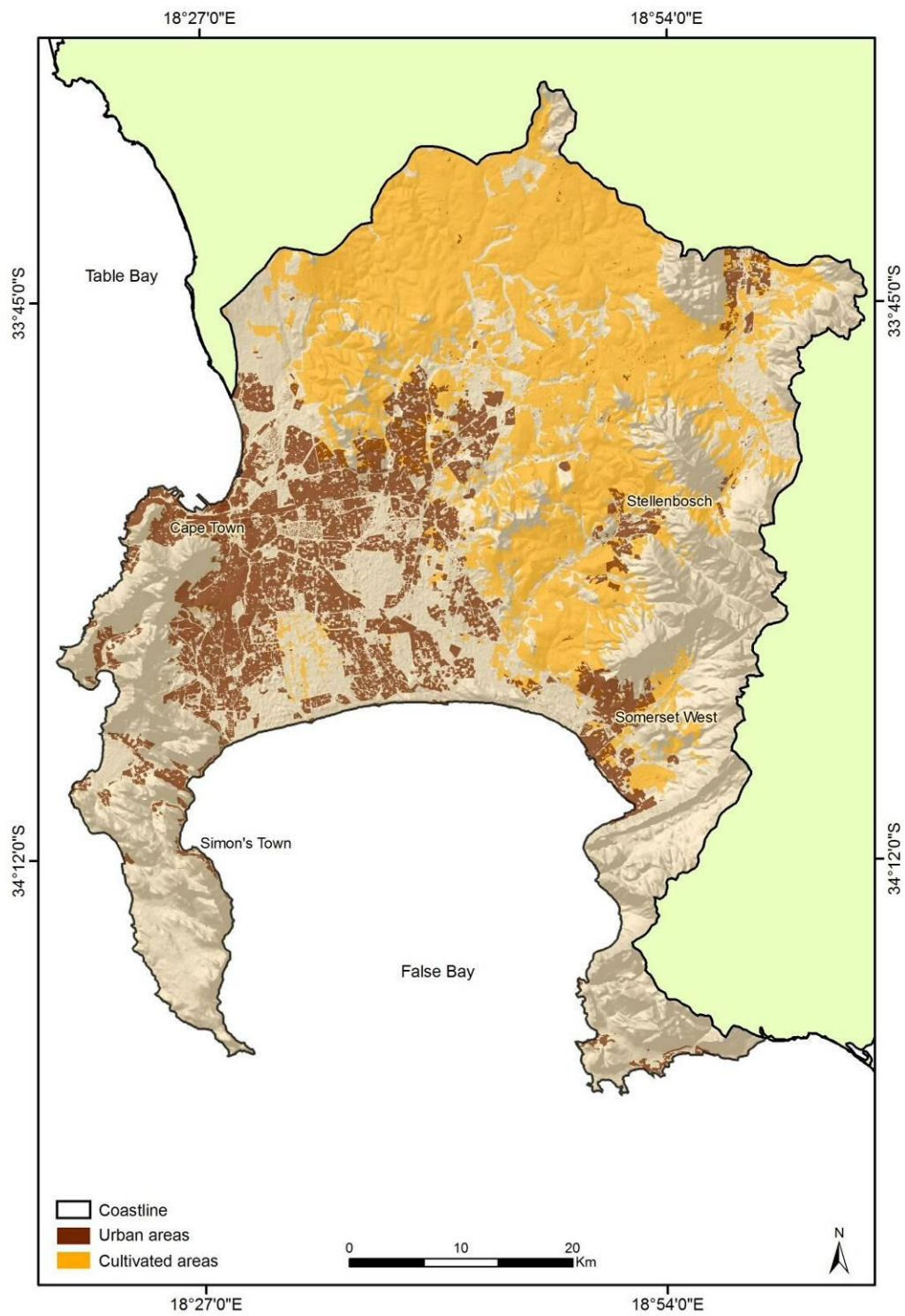


Figure 1.3 Expansion of urban and agricultural areas within the study area by 2006. (Data: RSA Planning Department).

National Environmental Management: Biodiversity Act No.10 of 2004

The South African community in general has a very limited knowledge and capacity to participate actively in the implementation of compliance with environmental and planning legislation, such as the National Environmental Management Act (NEMA) and its associated Acts such as the Conservation of Agricultural Resources Act (CARA) (RSA, 2002) etc.

The development of the Biodiversity Act in 2004 within the framework of the NEMA of 1998 is a positive sign that the South African government is aware of the importance of biodiversity conservation and the need to link conservation with sustainable development (RSA, 2004). The act also covers invasive alien species, which are a major threat to biodiversity, and puts obligations on private landowners and the government to clear invasive alien vegetation from their properties. Although, South Africa has a reasonably well-implemented River Health Programme for monitoring aquatic systems, one of the remaining challenges is to develop such a monitoring system for the terrestrial arthropods.

History of South African vineyard and plantation forestry

Vineyards

Vineyard establishment was one of the earliest developments of European colonisation in the CFR, and the first South African wine was produced at Groot Constantia in 1659 (Christopher, 1982). The main areas of vineyards were around Stellenbosch and on the eastern side of the Cape Peninsula. Vine plantings have also been extended up hillsides or areas previously ignored, and continued expansion of cultivated areas may be expected. However, urban pressures are resulting in some of the old established vineyards being built upon (Figure 1.4).



Figure 1.4 Residential area developments in the De Wijnlanden vineyards outside Stellenbosch.

Plantation forestry

Forestry consists mostly of imported species (Christopher, 1982) and the native forests have been substantially depleted and reduced to smaller patches. Alien-species plantations were a result of the indigenous forests not meeting the demand for timber, and so commercial plantation was begun in the late 1880s when the wattle bark was successfully exploited for the extraction of tannin for the leather industry. Commercial forest establishment rapidly expanded in the 1890s (Christopher, 1982). In South Africa, the impact of plantation forestry on biodiversity has been a serious concern (Johns, 1993; Cellier, 1994; Armstrong and van Hensbergen, 1996; Pryke and Samways, 2003).

Invasive alien plant introduction in South Africa

Alien plant invasions are becoming more widespread and serious throughout the world (Richardson et al., 2004). The establishment of alien trees in South Africa started immediately after European colonisation in 1652 (Geldenhuys et al., 1986). Invasive alien trees (IATs) are commonly coming from similar climatic environments overseas without their natural enemies and grow undisturbed (Mitchell, 1991). The first alien tree, *Quercus robur*, in South Africa was planted by Jan van Riebeeck in 1656 mainly for timber, fibre, firewood, shade, shelter and for their aesthetic value (Donald, 1978; Geldenhuys et al., 1986). Commercial plantations of the Cape Peninsula, mostly with species of *Pinus* and *Eucalyptus* started at Tokai in 1884 and then spread to other parts of the country (Campbell and Moll, 1977; Spilhaus, 1950).

Invasive alien plant control in South Africa

Constitutionally, South Africa has had a primary alien vegetation legislation promulgated in terms of the Conservation of Agricultural Resources Act (CARA) No. 43 of 1983, which categorizes invasive plants and stipulates what needs to be done with respect to their management and control (Zimmermann et al., 2004; RSA, 2002). The main objectives of this Act are to provide for the conservation of the natural agricultural resources of the country by the maintenance of the production potential of land, by combating and prevention of erosion and weakening or destruction of water sources, and by the protection of indigenous vegetation and combating of weeds and invader plants. Species such as of pine and gum trees are considered invader plants under category 2 species of CARA no. 43 of 1983, but are more commonly grown for commercial purposes or any viable and beneficial function, such as woodlots, fire belts, building material, animal fodder and soil stabilization.

Although prevention is the most cost-effective way to avoid the enormous expenses associated with plant invasions, invasive plants continue to be imported as trade commodities for horticultural use. Large-scale control operations were first carried out in the CFR during 1973, and the approach followed has been to cut down the infestations, burn the area, and then carry out regular follow-up operations (Combrink, 1985). Efforts to control alien invasions on private land are relatively limited, probably because the methods are expensive or there is a lack of knowledge about the invasions. Alien vegetation eradication is extremely labour intensive and requires initiative and skills. Nevertheless, there has been a national agreement on "kill them all" as the response to control current and future invasions by alien trees in South Africa.

The damaging effects of invasive alien organisms have triggered the development of invasive alien species eradication programme "Working for Water" in South Africa, which was launched by the national Department of Water Affairs and Forestry in 1995, aiming to control the spread of alien invasive plants, with the primary goal of increasing water supplies (Macdonald, 2004). The Western Cape is highly invaded with IATs compared to other South African Provinces (Figure 1.5). This programme evaluates biological and historical characteristics of species and prioritizes those that should be the focus of exclusion, and/or control. Alternative methods such as the use of fungus

Colletotrichum gloeosporioides have been considered to control the spread of shrub *Hakea sericea* in South Africa (Morris, 1989).

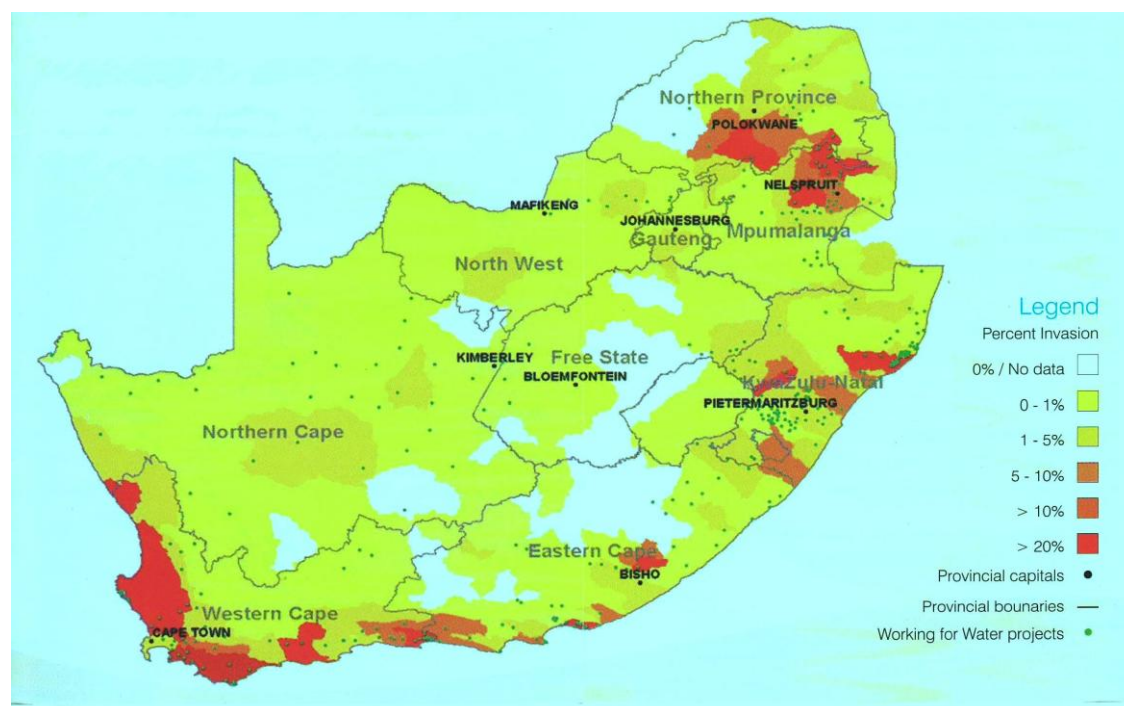


Figure 1.5 Map of Working for Water projects and invasions, showing estimated percentages of invasive alien trees covered per quaternary water catchment. Map: Working for Water programme.

Importance of arthropods in ecosystem processes

It is very challenging to determine the importance and value of insects to humans (Clark and Downes, 1996). Nevertheless, Edwards and Abivardi (1998) detailed the value of biodiversity from economic perspectives (i.e. costs and benefits) and recognize biodiversity as an essential and yet diminishing resource for human survival. Arthropods are essential in ecosystem processes because they perform many ecological tasks (Wilson, 1987) that human and all other animal species (and many plants) depend on (Daily, 2000). Arthropods pollinate crops, decompose wastes, maintain soil fertility, and are food for many mammals, birds, reptiles, amphibians and fish. Many of these processes would be reduced and even lost without arthropods. However, some insects are destructive to various crop plants that are of great importance to humans (Borror et al., 1989). Nevertheless, humans benefit from insects in many ways (Borror et al., 1989;

Clark and Downes, 1996; Daily, 2000). In addition, arthropods are good ecological indicators in most habitat types (Duelli and Obrist, 1998; Lawes et al., 2005).

Arthropods as ecological indicators

There is a distinct lack of detailed distribution and abundance data for most species of conservation concern in most parts of the world (Andelman and Fagan, 2000). Because arthropods are easy to sample in terrestrial habitats, they have potential to provide reliable data more useful to short-time ecosystem monitoring, impact assessment and conservation practices than inventories of vertebrates and plants (Murphy and Wilcox, 1986; Kremen et al., 1993; Brown, 1997). Indeed, the majority of animals in terrestrial habitats are arthropods (Duelli et al., 1999; Hansen, 2000), and are a vital component of terrestrial ecosystems (Lawes et al., 2005; Conrad et al., 2006). Moreover, many taxonomic groups are good candidates for a quantitative biodiversity assessment in all habitat types (Duelli and Obrist, 1998; Lawton et al., 1998; Duelli et al., 1999; Andelman and Fagan, 2000; Lawes et al., 2005). The majority of surface-active arthropods cannot avoid the impact of local vegetation disturbance (Lawes et al., 2005), owing to their close association with organic leaf litter layer (Giller, 1996; Andersen et al., 2004).

Aims of this study

Agriculture and IATs are among the major threats to biodiversity in the vicinity of Stellenbosch. It is crucial that multi-species approaches to conserving biodiversity be considered if the remaining habitat fragments are to be managed appropriately (Clevenger and Waltho, 2000).

Although some research has been done on the impact of invasive alien plants and agricultural intensification on biodiversity, little comparative work has been done on these two main threats to biodiversity, and none in the CFR, a major global biodiversity hotspot (Mittermeier et al., 2005). Biodiversity in agro-ecosystems has declined dramatically, and such a loss of diversity is hugely detrimental (Edwards and Abivardi, 1998). While it is known that conversion of the natural landscape to agriculture and the effect of invasive alien trees (IATs) are reducing insect diversity, it is not clear which of these factors is currently the most important, or even how comparable they are in terms of their effects on arthropod biodiversity. Put simply, are agricultural patches and IATs affecting insect species richness and abundance in a similar way or not?

The insights gained from this study should not only enable understanding whether human-modified landscapes can support indigenous species, but will strive to provide effective agricultural management strategies that are helpful for future conservation of biodiversity in agricultural areas. This study will seek to fill the gaps in the knowledge of vineyard versus invasive alien tree impact on arthropod diversity in the CFR. Specifically it aims to: (i) compare the richness of the fynbos arthropod fauna with that of vineyards, invasive alien plants, and vegetation cleared of invasive alien plants, and (ii) investigate correlation between vegetation type and insect diversity. The soil compaction in different vegetation types will also be compared to determine the impact of invasive alien trees and vineyards on soil profiles (specifically in relation to compaction), which in turn, may have an effect on surface-active arthropod diversity (McNaughton, 1988; Holland and Luff, 2004) and vegetation structure (Mitchell, 1991). This is an area of focus that has never been done before in the CFR.

Recent studies (Samways and Grant, 2006; Pryke and Samways, 2008) have shown that the results of arthropod sampling in the CFR are highly dependent on season. In response, this study was sensitive to this fact and considers seasonal variation in the surface-active arthropods sampled here.

The impact of alien plants vis-à-vis agricultural conversion were examined by contrasting arthropod species richness, abundance, soil compaction, soil moisture and leaf litter at several sites experiencing similar environmental conditions but situated in areas with different and adjacent vegetation types. Towards these aims, the following specific questions were addressed:

- Effect of vegetation type and clearing of alien vegetation on surface-active arthropods.
 - What is the comparative effect of vegetation type (fynbos vs IATs vs vineyard vs CIATs) on surface-active arthropod species richness and abundance?
 - In comparison with fynbos reference sites, is there any significant arthropod loss because of IAT infestation?

- How comparable is the surface-active arthropod species richness in CIATs with those in fynbos reference sites and those with IAT infestation?
 - How do arthropod assemblage patterns change in the different vegetation types, and which species are responsible for these changes, and which ones could be considered indicators of changes in vegetation types?
- Juxtaposition of IAT patches, vineyards and natural patches and the impact on arthropods.
 - Which two distinct but adjacent land-use types (including natural fynbos as a reference site) are important for sustaining high arthropod species richness?
 - Is species richness and abundance higher in ecotones than in adjacent land-use types?
- Soil profile in areas of invasive alien trees and agricultural practices, and the effect on arthropods.
 - Do IATs and vineyard activities have an impact on soil compaction?
 - Is there any significant correlation between arthropod species richness, soil compaction, leaf litter, site location and soil moisture content?
- Fires have a major impact upon arthropods, especially in the Fynbos (Pryke, 2008). However, few studies have actually quantified the impact, especially where IATs are involved. While part of the original plan, and as a result of incidental fires in some of the study sites (IATs and fynbos vegetation), opportunistically, the effect of fire was assessed on the recovery ability of ground-living invertebrate diversity within fragments of IATs and fynbos vegetation. Furthermore, the significance of vineyard and fynbos fragments adjacent to the burned area in surface-active invertebrates was assessed.

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Chapter 2 – Sites, Material and Methods

SITES

Seven vineyards and three nature reserves in the Cape Winelands of Western Cape Province (WCP) (Figure 2.1) were selected. This spatial replication overcame pseudoreplication. Site selection was based on land use and land cover. Land cover was established from direct area observation and aerial photographs, whereas land use was determined from interviews with land owners. Beside impact of insecticides, variation in arthropod diversity in agricultural landscapes often depends on biodiversity of the surrounding matrix as well as differing land use or management regimes (Duelli et al., 1999), and thus biodiversity evaluation in this study is at the landscape level.

All sites were selected from an elevation ranging from about 304–390 m above sea level. At each of these ten localities, transects were established, so that the focal land cover/ land use types were adjacent to each other. In total, there were 36 transects, 256 m-long. Half of the transect, i.e. 128 m, was on either side of the boundary of land use/ land cover types, with the exception of two transects (128 m-long) that were established across native vegetation and small invasive alien trees (IATs) fragments (Table 2.1).

Land cover/ land use type (hereafter referred to as vegetation type) was categorized as 1) natural or semi-natural (fynbos), 2) invasive alien trees (IATs), and 3) cleared of invasive alien trees (CIATs), and 4) vineyards. There were six pairs of vegetation types (Table 2.1 and Figure 2.2).

Vegetation type (fynbos) at the non-agricultural sites (e.g. Hottentots Holland and Jonkershoek nature reserves) was untransformed by human activity, with common plant species being *Protea* and *Salix* species. IATs sites had mainly *Acacia*, *Hakea*, *Pinus*, *Eucalyptus* and *Populus* trees (Macdonald and Richardson, 1986), with an understorey of grasses and forbs.

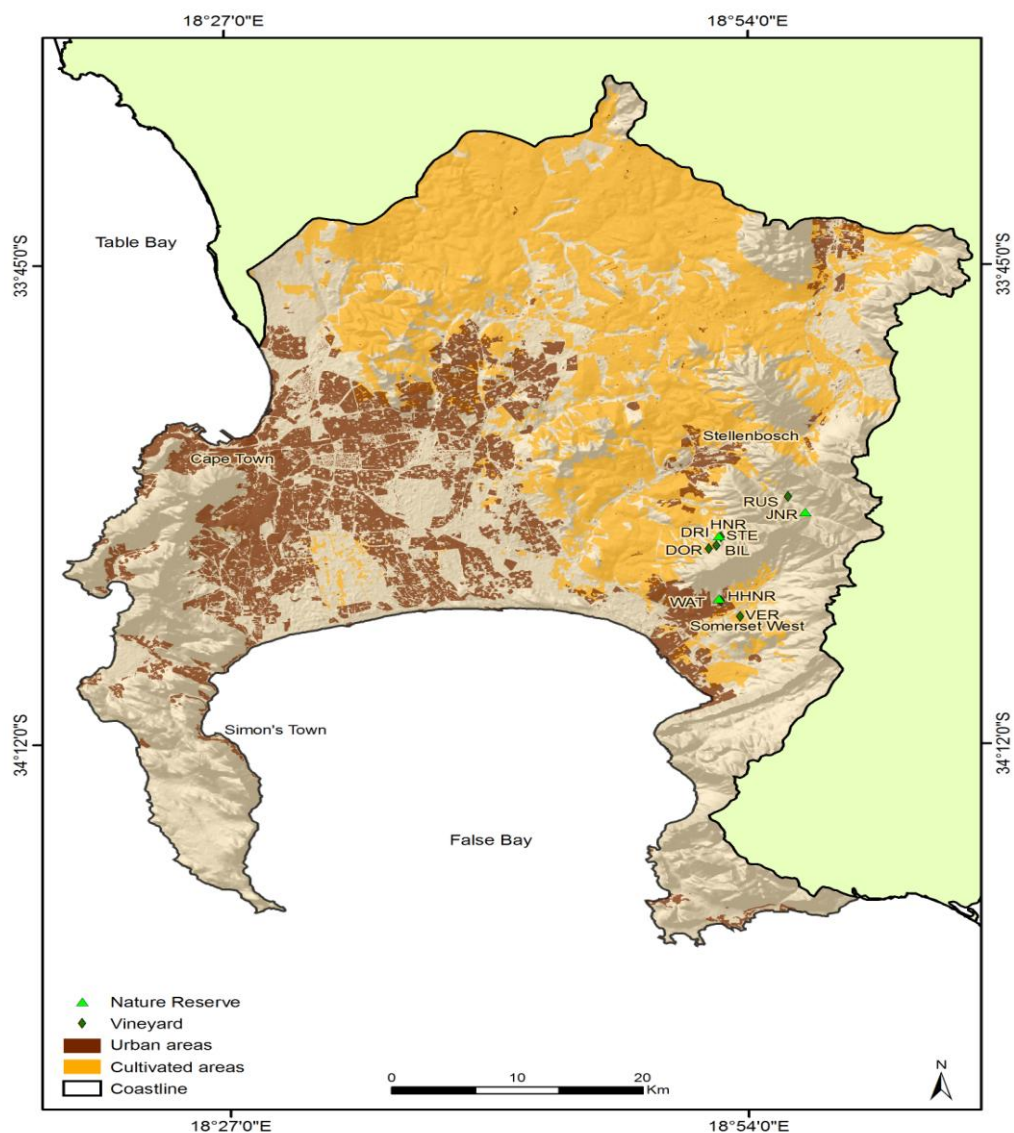


Figure 2.1 Map of the study sites around Stellenbosch and Somerset West, Western Cape Province, South Africa. RUS=Rustenberg; JNR=Jonkershoek Nature Reserve; HNR=Helderberg Nature Reserve; DRI=Driekoppen; STE=Stellenzicht; DOR=Dornier; BIL=Bilton; HHNR=Hottentots Holland Nature Reserve; WAT=Waterford; VER=Vergelegen.

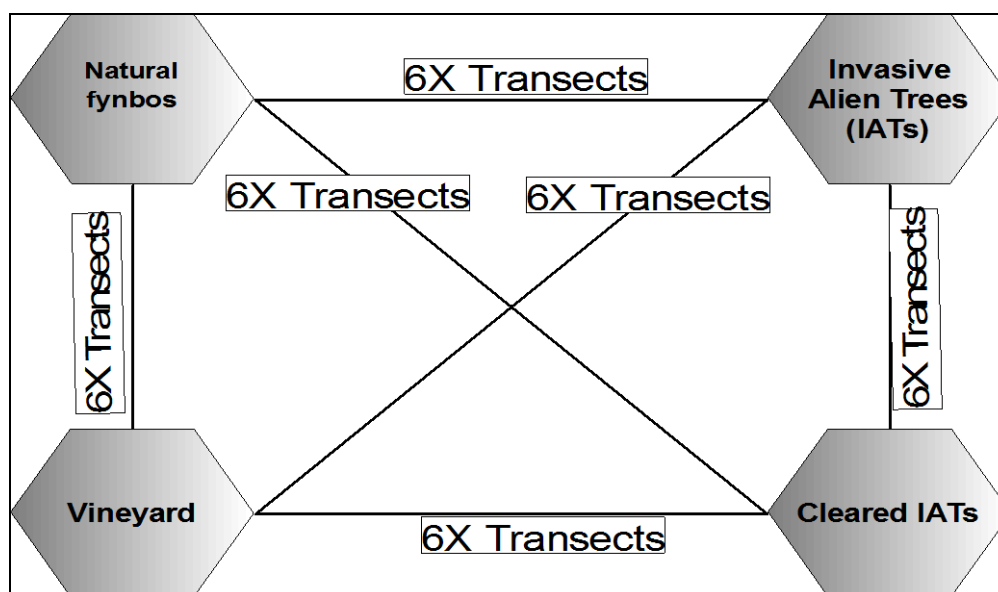


Figure 2.2 Schematic diagram showing arrangement of transects across selected vegetation pairs within the study area.

Table 2.1 Details of the study sites and transects. IATs = Invasive alien trees.

Category	Site name	Locality	Transect Code	Pairs of vegetation		Transect Length (m)	No. pitfall traps	
Vineyards	Vergelegen	S: 34.09206 E: 18.89851	Ver1	Cleared IATs	Natural Fynbos	256	28	
			Ver2	Cleared IATs	Natural Fynbos	256	28	
			Ver3	Cleared IATs	Natural Fynbos	256	28	
			Ver4	Cleared IATs	Natural Fynbos	256	28	
			Ver5	IATs	Natural Fynbos	256	28	
			Ver6	IATs	Cleared IATs	256	28	
			Ver7	IATs	Cleared IATs	256	28	
	Bilton	S:34.01431 E:18.87259	B1	Vineyard	IATs	256	28	
			B2	Vineyard	IATs	256	28	
			B3	Vineyard	IATs	256	28	
			B4	Vineyard	Natural Fynbos	256	28	
			B5	Vineyard	Natural Fynbos	256	28	
	Stellenzicht-Driekoppen	S:34.98575 E:18.95216	SD1	Vineyard	Cleared IATs	256	28	
			SD2	Vineyard	Cleared IATs	256	28	
	Waterford	S: 34.06625 E: 18.87626	W1	Vineyard	IATs	256	28	
			W2	IATs	Cleared IATs	256	28	
	Rustenberg	S: 33.96862 E: 18.9354	R1	Vineyard	Cleared IATs	256	28	
			R3	Vineyard	Cleared IATs	256	28	
			R2	Vineyard	Natural Fynbos	256	28	
			R4	Vineyard	Natural Fynbos	256	28	
			R5	Vineyard	Natural Fynbos	256	28	
R6			Vineyard	Natural Fynbos	256	28		
R7			Vineyard	IATs	256	28		
Dornier	S: 34.01731 E: 18.86607	D1	Vineyard	IATs	256	28		
Waterford-Driekoppen	S: 34.0063 E: 18.87639	WD 1	Vineyard	Cleared IATs	256	28		
		WD 2	Vineyard	Cleared IATs	256	28		
Natural reserves	Jonkershoek	S: 33.98317 E: 18.94967	J1	IATs	Natural Fynbos	256	28	
			J2	IATs	Natural Fynbos	256	28	
			J3	IATs	Natural Fynbos	128	24	
			J4	IATs	Natural Fynbos	128	24	
	Helderberg	S: 34.00535 E: 18.8748	H 1	IATs	Cleared IATs	256	28	
			H 2	IATs	Natural Fynbos	256	28	
	Hottentots Holland	S: 34.06436 E: 18.87469	HOT1	IATs	Cleared IATs	256	28	
			HOT2	IATs	Cleared IATs	256	28	
			HOT3	Cleared IATs	Natural Fynbos	256	28	
			HOT4	Cleared IATs	Natural Fynbos	256	28	
	Total number of pitfall traps							1000

Details of Sites

1) Vergelegen Estate

The natural vegetation on the this estate was predominantly mountain fynbos and renosterveld, including *Aspalathus forbesii*, *A. aspalathoides*, *Lebeckia sepiaria*, *Lotononis prostrate*, *Cyphia phyteuma*, *Chasmanthe aethiopica*, *Watsonia borbonica* and *Protea repens*. Also at this site were IATs (Table 2.2), some of which have been cleared (Table 2.3a). There were vineyards also at this site (Table 2.3b). Generally, soils under vines had very high clay content.

Table 2.2 Alien plant species at the Vergelegen study site.

Scientific name	Common name
<i>Acacia mearnsii</i>	Black wattle
<i>Eucalyptus lehmannii</i>	Spider gums
<i>E. diversicolor</i>	Karri gums
<i>Hakea drupacea</i>	Sweet hakea
<i>H. sericea</i>	Silky hakea
<i>Pinus pinaster</i>	Cluster pine
<i>P. radiata</i>	Radiata pine

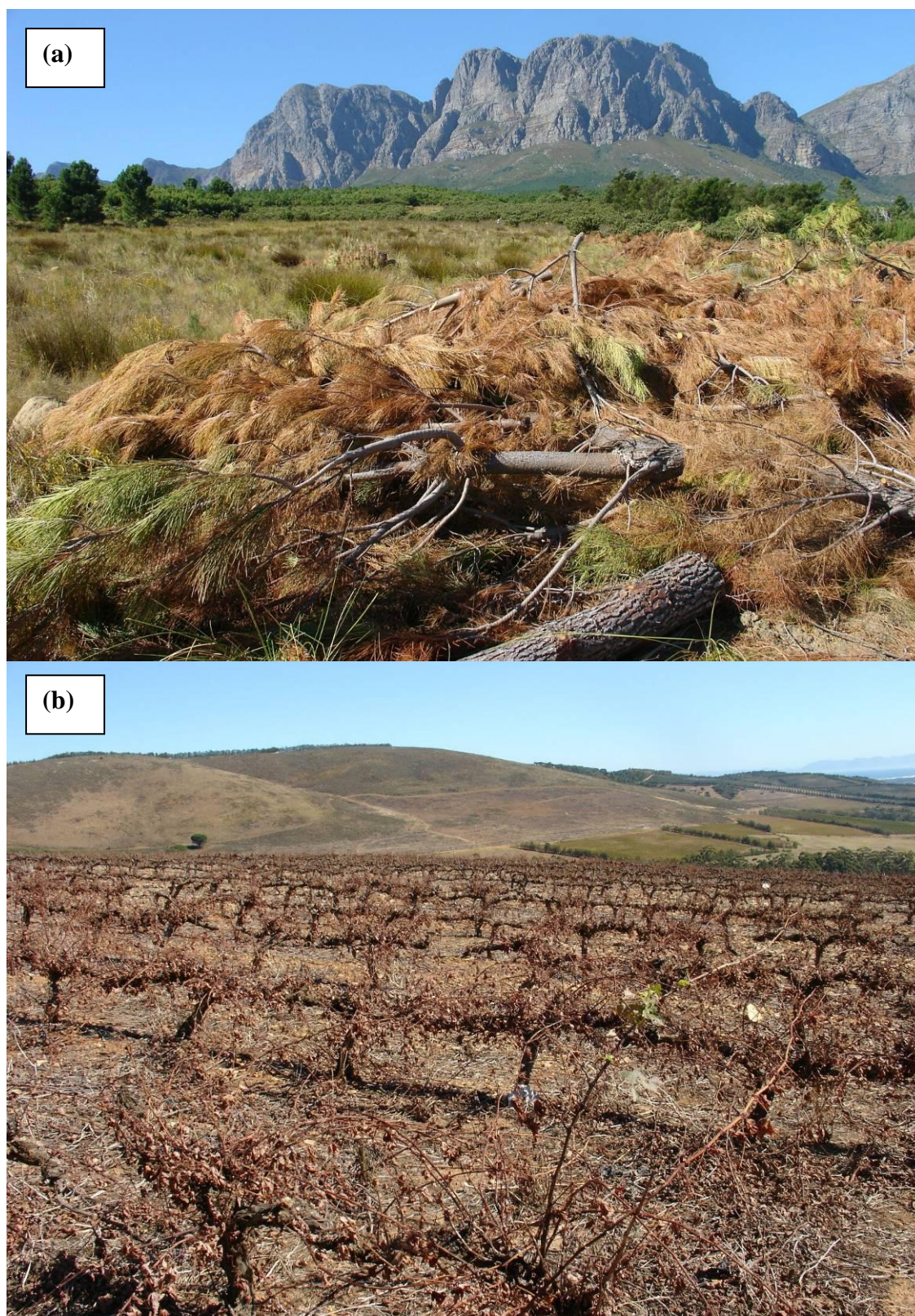


Figure 2.3 Vergelegen Estate showing (a), site cleared of pine trees adjacent to a mountainous fynbos vegetation remnant and (b), vineyards (early winter) below land cleared invasive alien trees.

2) Bilton Estate

Natural fynbos vegetation was heavily invaded, mainly by acacias and pines adjacent to vineyards (Table 2.3). However, small remnants of fynbos vegetation were also present adjacent to the vineyards. Dominant fynbos species included *Protea repens*, *Gymnodiscus capillaries*, *Dimorphotheca pluvialis* and *Hymenolepis crithmoides*. Soil type at this site had Cape granite mixed with sandstone of the Table Mountain Group. These vineyards had a relatively high content of organic materials along with a wheat cover crop (Figure 2.4).

Table 2.3 Alien plant species at the Bilton study site.

Scientific name	Common name
<i>Acacia longifolia</i>	Long-leaved wattle
<i>A. saligna</i>	Port Jackson
<i>Pinus pinaster</i>	Cluster pine
<i>Eucalyptus diversicolor</i>	Karri gums



Figure 2.4 Bilton Estate with wheat as a cover crop adjacent to a *Eucalyptus diversicolor* (Karri gums) remnant.

3) Stellenzicht Estate

This estate consisted mainly of vineyards on the slopes facing west and north-west and on soils originating from decomposed granite and Table Mountain sandstone. Dominant fynbos species included *Protea repens*, *Helichrysum retortum*, *Lachnospermum fasciculatum* and *Metasia densa*. This site was selected mainly for its neighbouring fynbos vegetation cleared of dense invasive alien trees (i.e. *Pinus* and *Acacia* species) (Figure 2.5). Some scattered *Acacia saligna* trees were still present at the site.



Figure 2.5 Stellenzicht Estate with wheat as a cover crop adjacent to a site (at back of picture) cleared of invasive alien trees (mostly pine and acacia species).

4) Driekoppen Estate

This estate was selected mainly for having vineyards adjacent to natural vegetation cleared of dense IATs (i.e. *Pinus* and *Hakea* species) (Figure 2.6). The CIATs vegetation had a low, open structure, with a canopy cover of about 80%. Dominant fynbos species included *Aspalathus forbesii* and *Protea repens*. Relatively few scattered *Pinus* species, were still present in the area. The geological formation consisted of sandstone of the Table Mountain Group (Dr. K. Riemann, UMVOTO Pty Ltd, pers. com.).



Figure 2.6 Driekoppen Estate showing a fynbos vegetation remnant cleared of invasive alien trees.

5) Waterford Estate

This estate is located between the Simonsberg and Helderberg mountain ranges (Figure 2.7 and Figure 2.8). Vegetation was predominantly IATs, mainly poplar and pines, adjacent to the vineyards (Table 2.4). Fynbos species included *Wurmbea spicata* and *Bolusafrab bituminosa*. The geological formation consisted of sandstone of the Table Mountain Group, with more fine organic materials in rocky vineyard areas. A *Eucalyptus diversicolor* patch on the foothill is situated on a plain of coarse alluvium (Figure 2.8).

Table 2.4 Alien plant species at the Waterford study site.

Scientific name	Common name
<i>Populus canescens</i>	Grey poplar
<i>Pinus radiata</i>	Radiata pine
<i>Eucalyptus diversicolor</i>	Karri gum



Figure 2.7 Aerial photograph showing Waterford vineyards. Photo: Antenie Carstens.



Figure 2.8 Waterford Estate with a fynbos vegetation remnant cleared of invasive alien trees adjacent to *Eucalyptus diversicolor* (Karri gum) remnant.

6) Rustenberg Estate

The vegetation at this estate was predominantly mountain fynbos (Figure 2.9) adjacent to vineyards. Dominant fynbos species included *Aspalathus forbesii*, *A. cephalotes*, *A. tridentate*, *Liparia vestita*, *Xiphotheca fruticosa*, *Protea compacta* and *P. repens*. Alien trees, mainly black wattle, hakea and pines have invaded some sections of Rustenberg adjacent to the vineyards (Table 2.5). Since 2000, this estate started to clear invasive alien trees, with waterways being given a priority. Currently, about 500 ha has been set aside for conservation purposes. Upper slopes consist of sandstone of the Table Mountain Group, with some granite and shale intrusions on the foothill and valley, where the vineyards are cultivated (Dr. K. Riemann, UMVOTO Pty Ltd, pers. com.).

Table 2.5 Alien plant species at the Rustenberg study site.

Scientific name	Common name
<i>Acacia mearnsii</i>	Black wattle
<i>A. saligna</i>	Port Jackson
<i>Hakea sericea</i>	Silky hakea
<i>Pinus radiata</i>	Radiata pine

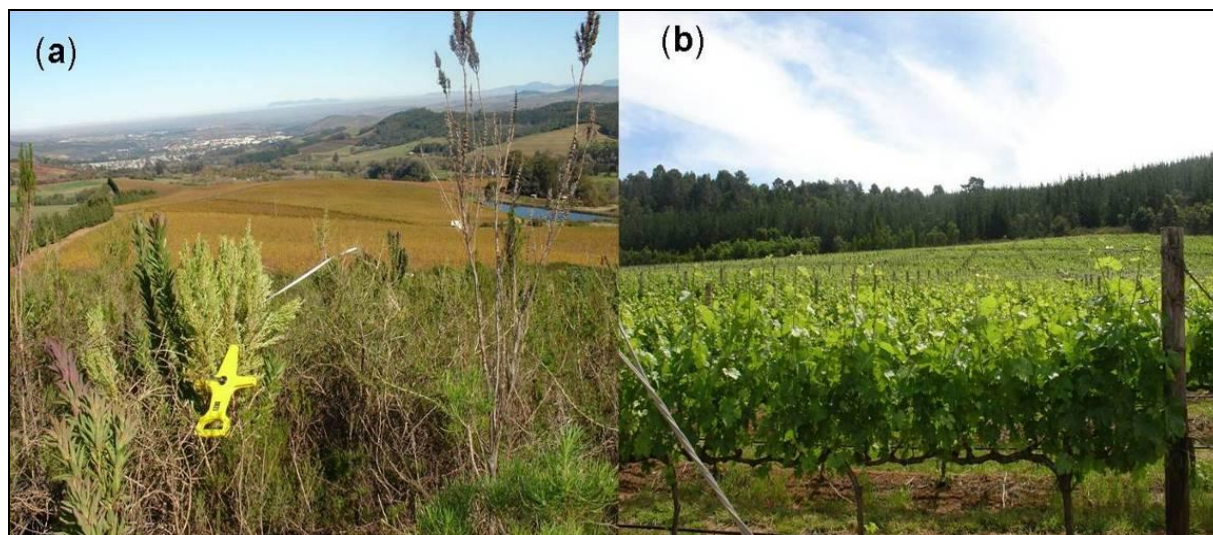


Figure 2.9 Rustenberg Estate showing (a), fynbos vegetation remnant adjacent to the vineyard area and (b), invasive alien trees (mostly pines) adjacent to the vineyard area.

7) Dornier Estate

This estate is situated on the foothills of Stellenbosch Mountain (Figure 2.10). Vegetation was predominantly mountain fynbos, although there were some IATs, mainly poplar and pines adjacent to the study vineyard (Table 2.6). Soils at this estate consisted of coarse alluvium and Cape granite frequently buried by rocky debris from the upper sandstone of the Table Mountain Group.

Table 2.6 Alien plant species at Dornier study site.

Scientific name	Common name
<i>Populus canescens</i>	Grey poplar
<i>Pinus radiata</i>	Radiata pine



Figure 2.10 Dornier Estate showing vineyards adjacent to invasive alien trees, *Populus canescens* (Grey poplar).

8) Jonkershoek Nature Reserve

This nature reserve is 9 km from the town of Stellenbosch. The 9800 ha reserve comprises the Jonkershoek mountains and portions of the upper Jonkershoek valley (Figure 2.11). The Jonkershoek mountains, form part of the larger Boland Mountain range. The Eerste river has its source high in these mountains and flows through the Jonkershoek valley. Distinctive species were *Aspalathus forbesii*, *Liparia vestita*, *Xiphotheca fruticosa*, *Cyphia phyteuma*, *Watsonia borbonica*, *Leucadendron salignum*, *Protea repens* and *P. neriifolia*, as well as various ericas and restios. Oak trees (*Quercus robur*), although not indigenous, have been allowed to remain because of their special historical value. Large Radiata pine (*Pinus radiata*) plantations are a distinctive feature of the lower Jonkershoek valley. Upper slopes consisted of sandstone of the Table Mountain Group with some shale intrusions. Lower slopes consisted of Cape granite frequently buried by rocky debris from the upper sandstone (Dr. K. Riemann, UMVOTO Pty Ltd, pers. com.). The reserve was selected for presence of mountain fynbos adjacent to invasive alien trees (i.e. *Pinus* and *Hakea* spp.).



Figure 2.11 Jonkershoek Nature Reserve study site showing pine trees adjacent to fynbos vegetation.

9) Helderberg Nature Reserve

This nature reserve is located in the south-eastern slopes of Helderberg Mountains. The 286 ha reserve is few kilometres outside Somerset West town (Figure 2.12). It has many protea species, and a wide variety of pelargoniums and bulbous plants. Mountain fynbos and renosterveld were the two main vegetation types. Distinctive vegetation species were *Aspalathus forbesii*, *A. aspalathoides*, *Lotononis prostrate*, *Xiphotheca fruticosa*, *Haemanthus coccineus* and *Cyphia phyteuma*. The reserve consisted of sandy to loamy soils with more fine materials under renosterveld and contains coarse shale under pines. This study site was selected for its invasive alien trees adjacent to vegetation cleared of invasive alien trees (i.e. *Pinus* spp.).



Figure 2.12 Helderberg Nature Reserve study site showing pine trees adjacent to vegetation cleared of pine and hakea species.

10) Hottentots Holland Nature Reserve

This nature reserve is in the Hottentots Holland Mountains, 90 km south east of Cape Town (Figure 2.13). The 42 000 ha reserve stretches from Elgin in the south to beyond Villiersdorp in the north, and from the Stellenbosch mountains in the west, eastwards to the Groenland mountains. The reserve consists of sandy to loamy soils with more fine materials under fynbos compared to under pines. The reserve was selected mainly for the presence of natural mountain fynbos adjacent to IATs (i.e. *Pinus radiata*). Distinctive vegetation species were *Amphithalea ericifolia*, *Berzelia lanuginose*, *B. abrotanoides*, *Staberoha distachya* and *Leucadendron salignum*. Specifically, the study site was at Nuweberg, high in Viljoen's Pass between Grabouw and Villiersdorp.



Figure 2.13 Hottentots Holland Nature Reserve study site showing mountainous fynbos vegetation adjacent to pine trees.

SAMPLING

Transects have been widely used for studying insect assemblages and associated environmental variables (Whittaker, 1972; Jonhston et al., 1992; Procheş and Cowling, 2006). Transects have the advantage of providing maximum data with minimal effort when correctly located and sampled. A wide range of surface and litter arthropods was sampled. Use of multi-taxon approach is considered reliable for reaching more general conclusions regarding impacts of disturbance on biodiversity (Duelli, 1997; Jeanneret et al., 2003).

There are various methods for collecting arthropods in managed and unmanaged landscapes. To obtain reproducible findings for scientific assessment of species diversity and abundance at a particular site, the sampling method must be strictly standardized. Pitfall traps are one type of standardization for surface-active arthropods (Luff, 1975; Margules and Usher, 1981; Eyre and Rushton, 1989; Duelli et al., 1999; Golden and Crist, 2000; Kotze and Niemelä, 2002; Druce et al., 2007; Lach, 2008), and have been widely used as an arthropod collection method in many ecosystems and for selective biodiversity assessments (Blumberg and Crossley, 1988; Chiverton and Sotherton, 1991; Duelli et al., 1999; Lach, 2008). Moreover, pitfall trap catches in most habitat types allow for standard statistical analysis (Duelli et al., 1999). Pitfall traps can be left for long periods in the field unattended, and are both time and cost effective, while minimizing biased estimates of active arthropod abundance (Fraser et al., 2008). Such a rigorous, feasible, rapid, and effective sampling protocol was chosen to collect comparable data for arthropods. Nevertheless, it is recognized here that pitfall trapping will not sample all species equally, and some not at all (Slotow and Hammer, 2000). However, the point here was comparative analysis across different landscape types using a standard, comparative method, of which pitfall trapping is one of the best available (Samways et al., 2009).

There is no absolute answer as to how many and how long should traps be left at a site for the most effective sampling (Wells and Decker, 2006). Fraser et al. (2008) concluded that within a habitat, pitfall traps sample more of the arthropod assemblage when traps are widely spaced, and also, when sampling both marginal and core habitats. Agosti and Alonso (2000) recommended about a 48 hour-period for use of pitfall traps

for standardized arthropod surveys. Nevertheless, recent findings for sampling leaf litter arthropods (Borgelt and New, 2006), showed that a one week period had yielded samples that may be adequate for appraising representativeness of local arthropod assemblages. Such arthropod data are most commonly needed for site monitoring or comparisons. However, more extensive sampling is necessary to incorporate all rarer species (Borgelt and New, 2006).

Dominant plant species were assessed using line-intercept method. This is a valuable approach, especially when sampling shrubs, which are often difficult to measure quantitatively, and where plants are often sparsely distributed (Kent and Coker, 1992).

Sampling design

From August 2006 to January 2008, three field visits (i.e. August-October 2006, May-July 2007 and November-January 2007/8 (November and December 07- January 08)) at each study sites were undertaken to sample arthropods and measure habitat characteristics (i.e. soil compaction, soil moisture and leaf litter). The most effective sampling method is not always the most efficient one (Druce et al., 2004) for different arthropod species. Nevertheless, the use of pitfall traps was considered to be more appropriate to sample arthropods for the purpose of this study. Sampling design (Figure 2.14) allowed a measure of within-site variance of pitfall trap collection, while controlling the distance from the ecotone or transitional zone.

Transects were used to sample arthropods, vegetation and soil. Transects were laid out as shown in Figure 2.14. A trap-set of two individual pitfall traps, 1 m apart, was placed at log 2 intervals: 2, 4, 8, 16, 32, 64 and 128 m on either side of the ecotone between two adjoining vegetation types, resulting in 256 m-long transects across adjoining vegetation types. There were, therefore, 1000 pitfall traps (two per set, fourteen sets per transect, six transects per vegetation type pair and six vegetation pairs from four vegetation types) (Table 2.1). This logarithmic arrangement was to give more detailed data in the vicinity of ecotones.

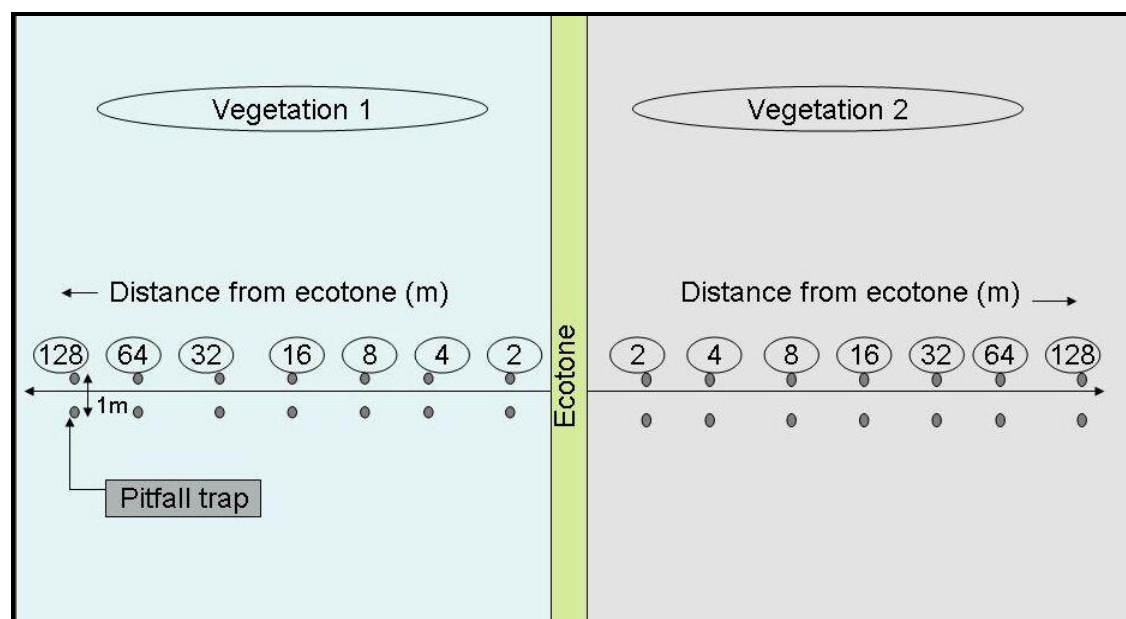


Figure 2.14 Schematic representation of the experimental design showing one pair of vegetation types, and pitfall trap distance placement within transects. Seven pitfall trap locations were placed on either side of the ecotone. (N.B: Drawing not to scale).

Sampling vegetation and leaf litter

The line-intercept method was used to record dominant plant species at all selected sites during the study period (Figure 2.15). A 128 m-length of tape measure was laid out, and all species intercepting or touching the line was recorded (Figure 2.15). The method was extended to such an extent that all species within 0.5 m on either side of the line were included. At least five dominant plant species along each transect were identified to species level (see Details of Sites).

The 2 m² area quadrats for leaf-litter depth survey were created at each sampling station around the trap set. A sampling station was defined as a specific pitfall trap location along the transects (at a specific distance from the ecotone zone) from where arthropods were collected. Mean leaf-litter depth was calculated from three random estimated measurements in each quadrat by inserting a steel rod, 4 mm in diameter, into the leaf-litter until the harder soil layer was reached (Lawes et al., 2005).



Figure 2.15 Vegetation assessment using line intercept method in Driekoppen Estate. A measuring tape was used for determining 1m intervals along transect where plant species were recorded.

Sampling arthropods

Pitfall traps were used to sample surface and litter arthropods and were made of 500 ml plastic honey jars, each containing a replaceable paper cups which were 8 cm wide (diameter) and 12 cm deep (Figure 2.16). Each trap was one-third filled with 70% Ethenediol (antifreeze fluid). Samples were washed and placed in 70% ethanol in the laboratory. Permanent pitfall traps were established at all sampling stations, which remained closed during non-sampling periods and opened for five days at a time.

Sampling in vineyards was done under vine rows to minimize impact of disturbance by farm activity. All pitfall traps were set up at least three months prior initial sampling to eliminate ‘digging effect’. Traps were operational for five consecutive days without rain, to prevent filling up of traps during the sampling periods from August 2006 to January 2008. Arthropods from each trap set were combined. This resulted in one sample per sampling station (i.e. 1000 pitfall traps resulted in 500 samples per sampling period). As a result, a total of 1 500 samples were collected over three sampling periods. The following norms were used for sampling period determination: Sampling period 1 = August, September, October; sampling period 2 = November, December, January; and sampling period 3 = May, June, July.

Arthropods from each sample were identified to species level where possible, or assigned to morphospecies. Voucher specimens of each arthropod species collected were deposited in the Entomology Museum, Stellenbosch University. Spider specimens were deposited in the National Collection of Arachnida, National Museum, Pretoria. Identification was by using Picker et al. (2004); Leroy and Leroy (2003); McGavin (2000); Preston-Mafham and Preston-Mafham (2005).



Figure 2.16 (A) pitfall trap consisting of (1) open white 500 ml holding jar sunk directly into the soil, (2) collecting paper cup which is one-third filled with a preservative of 70% Ethanediol in water. The collecting cup was fitted into the holding jar, and could be easily lifted from the holding jar for emptying of the trap, (3) holding jar lid for closing the trap when not in use. (B) Closed pitfall trap buried flush with the soil surface to avoid unnecessary collection and provide protection from rain immediately after collection.

Soil compaction

Soil compaction was measured at each pitfall trap set (Figure 2.14) (i.e. 14 sampling stations along each 256 m-long transect and 12 sampling stations for 128 m-long transects) during the study period using a radioactive moisture density gauge, Troxler 3411-B instrument (Figure 2.17), which measures both soil compaction and soil moisture content at a selected depth. It was assumed that ground-dwelling insects will be affected down to a depth of 15 cm, the maximum depth studied here.

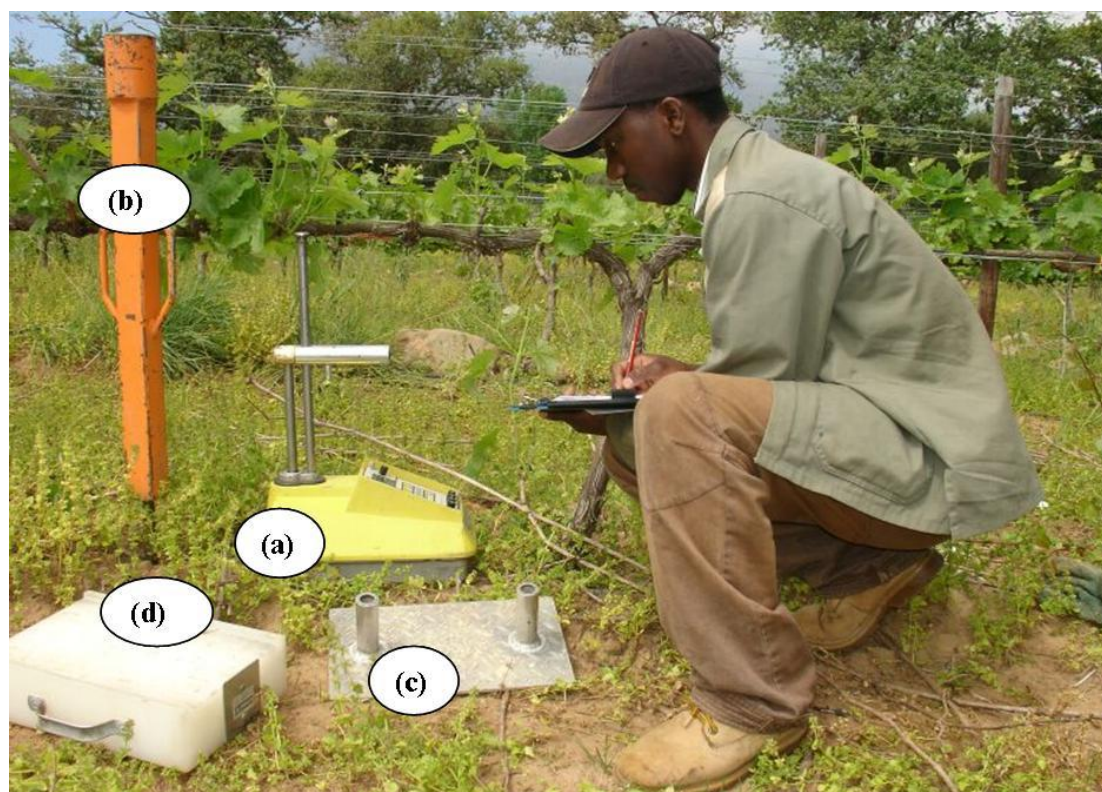


Figure 2.17 Soil compaction measurement in the Dornier vineyard site. (a) Troxler 3411-B instrument with sensing steel rod, (b) hammer used to create a hole on the ground for the sensing rod, (c) hammer stand used to ensure a straight hole in the ground, (d) basement used to hold Troxler instrument during calibration.

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Chapter 3 – Comparative Footprint of Different Vegetation types on Arthropod Diversity

INTRODUCTION

Ecological impacts of invasive alien trees

Invasions by alien tree species is a global environmental problem (Mack et al., 2000; Van Wilgen et al., 2001; Hulme, 2006; Richardson and Pyšek, 2006; Wolfe et al., 2008), threatening natural habitats and arthropod biodiversity (Skaife, 1955; Winterbottom 1968, 1972; Cowling et al. 1976; Scott, 1978; Donnelly, 1983; Bond and Slingsby, 1984; Donnelly and Giliomee, 1985; Macdonald and Richardson, 1986; Richardson and van Wilgen, 1986; Manders 1989; Wood and Samways 1991; Samways and Moore 1991; Richardson et al. 1992; Armstrong and van Hensbergen 1995, 1996; Calder, 1999; Dye and Jarman, 2004; Forsyth et al., 2004; Marais et al., 2004; Richardson and van Wilgen, 2004; Samways and Taylor, 2004; Zimmermann et al., 2004; Cushman and Meentemeyer, 2008). However, individual natural habitats vary considerably in their susceptibility to invasion (Chytrý et al., 2008). The impacts of alien tree species in natural systems may be dependent not only on invader attributes but also on characteristics of the invaded community (Mason and French, 2008). Alien trees and shrubs in particular, are widespread in the Cape Floristic Region (CFR) and are very successful because of their good colonizing abilities, especially in disturbed areas (Holmes and Richardson, 1999). Certain invasive alien trees (IATs) are successful because they possess novel characteristics that are unique to plant communities in their introduced ranges (Callaway et al., 2008). Rapid evolutionary changes are common during invasions, which include rapid adaptation of invaders to new environments (Whitney and Gabler, 2008).

In many parts of the world, alien trees are the heart of commercial forestry, and many were planted for a wide range of uses before spreading from planting sites. IATs such as *Pinus*, *Acacia* and *Eucalyptus* species are of major commercial importance in South Africa, but they are also a threat to water supplies and biodiversity, as is direct human transformation of natural habitats (Wittenberg and Cock, 2001; Le Maitre et al., 2004). Alien organisms (both fauna and flora) have greatly affected fynbos vegetation, and

will continue to do so until they are successfully controlled (Macdonald and Richardson, 1986).

Plantation forestry impact on biodiversity

Plantation forestry in South Africa encompasses the growing of alien trees, notably *Pinus*, *Acacia* and *Eucalyptus* species (Lückhoff, 1973; Geertsema and van den Berg, 1973; Geldenhuys et al., 1986; Richardson and Bond, 1991; Armstrong and van Hensbergen, 1996). Plantation forests increase the amount of edge habitat in the CFR, and they break up formerly continuous old-growth fynbos vegetation into smaller fragments. This fragmentation has affected the species, ecosystems, and processes associated with vegetation interiors by reducing interior habitat quality and isolating the remaining interior habitat. Species of *Eucalyptus* and *Pinus* have threatened natural vegetation through their spread, establishment, and rapid growth in disturbed areas, as well as by suppressing regeneration of indigenous species (Phillips, 1928; Pott, 1996). Moreover, compared to indigenous vegetation (Esler and Cowling, 1990), establishment of an overstorey of IATs reduces species richness of native vegetation markedly (Richardson and van Wilgen, 1986; Richardson et al., 1989; Leege and Murphy, 2001).

Impact of agriculture

Conversion of natural ecosystems into agriculture changes ecosystem composition and its function (Donald and Evans, 2006), and its biodiversity (Turin and den Boer, 1988; Newton, 2004; Gaigher, 2008). Agriculture is one of the most significant human-induced disturbances that threatens terrestrial biodiversity (Wilson, 1988; Sala et al., 2000; Tilman et al., 2001), by affecting the availability of suitable terrestrial habitats (Erhardt, 1985; Thomas and Mallorie, 1985; Murphy and Wilcox, 1986; Mattoni, 1989; Morris et al., 1989; Launer and Murphy, 1994; Brown, 1997; Feber et al., 1996; Warren et al., 1997; Jeanneret et al., 2003; Kleijn and van Langevelde, 2006). For example, vegetation structure is important to some arthropods because it can affect their ability to thermoregulate and reproduce (Holl, 1996). However, vegetation loss for agricultural purposes does not threaten all arthropods equally (Fleishman et al., 1999), with some relatively small-scale agriculture maintaining open, early successional habitats favoured by some arthropod species (Shreeve and Mason, 1980; Sibatani, 1980).

Many insecticides used in agricultural areas are known to be detrimental to a range of flora (Pimentel and Greiner, 1997) as well as some beneficial arthropod species (Villanueva-Jiménez and Hoy, 1998; Wakgari and Giliomee, 2003), and are generally counterproductive from a long-term perspective, although most efficient means of pest control available for a short-term (Ekbom, 2000). Therefore, regular application of insecticide products in the vineyards is likely to have a negative impact on various insects, agricultural production and in the long term, reduces agricultural sustainability (Barbosa and Schultz, 1987; Pimentel and Greiner, 1997). For example, exclusion of beneficial agricultural predators of pests has led to proliferation of several pests and diseases (Pimentel and Greiner, 1997).

Cape Floristic Region biodiversity

The CFR is one of the world's centres of plant diversity and endemism (Linder, 2005; Procheş and Cowling, 2006), with 67% of species endemic to the region (Linder, 2005). Studies on insect–plant diversity relationships are available for only a limited range of geographical regions, spatial scales, and insect groups, with most studies addressing a single insect group (Strong et al., 1984). Fynbos vegetation of the CFR has been suggested to have a proportionately poor insect fauna (Giliomee, 2003; Johnson, 1992), suggesting a deviation from the generally good large-scale relationship between plant and insect diversity (Price et al., 1998; Wright and Samways, 1998; Hawkins and Porter, 2003). However, for certain insect groups in the CFR (Cottrell, 1985), high species diversity and endemism has been found (Davies, 1988a, b; Picker and Samways 1996; Procheş and Cowling, 2006; Procheş et al., 2009). Of concern is that not only are alien trees impacting on this diversity but also agriculture, especially vineyards, the dominant agricultural activity in the region (Rouget et al., 2003), which is a global biodiversity hotspot (Mittermeier et al., 2005).

Aims of this chapter

While alien trees and agricultural conversion are both known to have a major impact upon arthropods, the differential impacts of these two types of vegetation change on the arthropod assemblage are poorly known. Furthermore, little is known on the recovery of arthropod diversity once alien trees have been removed. Following on from this, the following primary question is addressed here: Is vegetation type (fynbos vs invasive alien trees (IATs), fynbos vs areas cleared of invasive alien trees (CIATs), IATs vs CIATs, IATs vs vineyards, CIATs vs vineyards and fynbos vs vineyards) an important

determinant of surface-active arthropod species richness, abundance and diversity? Embedded in this question is also that of whether the quantitative and qualitative impact of IATs is the same or not as that of agriculture, particularly vineyards. If so, then I ask in which way does the arthropod assemblage change in terms of characteristic or indicator species? Furthermore, are certain arthropod species lost to both IATs and agricultural development? Do these particular species recover when IATs are removed? It is hypothesized that the impacts of these two types of landscape transformation would be the same. I also ask whether the clearing of IATs enabled some quantitative and qualitative recovery of arthropod diversity. The value of answering this question lies in the fact that currently, there is a large-scale removal of invasive alien trees in the CFR, yet, little is known about how this restoration benefits terrestrial biodiversity, especially invertebrates.

STUDY SITES

The Cape Floristic Region (CFR) is within the Western Cape Province of South Africa. Sampling was carried out in nature reserves, stands of IATs and wine estates adjacent to each other within the CFR.

➤ *See Chapter 2 for details of study sites*

METHODS

Sampling was from August 2006 to January 2008 (i.e. August-October 2006, May-July 2007 and December 2007-January 2008)) at each study site. Surface-active arthropod taxa were sampled using pitfall traps for standardized sampling and comparative analysis (Borgelt and New, 2006). Surface-active arthropod diversity was sampled in each of four vegetation types with varying degree of invasion or transformation, highly invaded, previously cleared of invasive alien trees, pristine/natural, and vineyards. The collected surface-active arthropods were sorted and placed into families (Appendix 3a). Where possible, they were further identified to species. Nevertheless, all specimens were recorded at least to family. Species identifications were undertaken or verified by taxonomic specialists: Araneae and Solifugae (Dr. Ansie Dippenaar-Schoeman of ARC-Plant Protection Research Institute), other arthropods (Prof. Henk Geertsema and Mr. Patrick Reavel of Stellenbosch University).

➤ *See Chapter 2 for details of Material and Methods*

DATA ANALYSES

Species accumulation curves were used to determine whether adequate sampling had been done to make comparisons between the vegetation types based on adequate data. The curves were produced using EstimateS version 8.0.0 with samples randomized 50 times (Colwell, 2006), for all vegetation types separately, as well as for all vegetation combined. A variety of non-parametric species estimators were used to provide the best overall arthropod species estimates for all vegetation types (Hortal et al., 2006). The incidence based Coverage Estimator (ICE) is considered a robust and accurate estimator of species richness (Chazdon et al., 1998), whereas Chao2 and Jackknife estimators provide the least biased estimates should insufficient sampling be an issue (Colwell and Coddington, 1994). Therefore, these estimators were calculated using EstimateS (Colwell, 2006) for all vegetation types separately and for a combination of them.

Temporal analysis was also undertaken on the arthropod species data for all the vegetation types in sampling period 1-3. Residuals for the arthropod species richness per sampling period were tested for normality using Shapiro and Wilk's W statistic (SPSS Inc., 2006).

Arthropod species were normally distributed and their variances were homogeneous. One-way analysis of variance (ANOVA) was performed on species and log transformed abundance data comparing the impacts on arthropods of the different vegetation types and different sampling periods, with multiple comparisons of the means using the Bonferroni methods (Legendre and Legendre, 1998). ANOVA was used when testing whether there were differences between the means of several populations, based on samples taken from each population. In one-way ANOVA, the null hypothesis tested was that all the populations have the same mean.

ANOVAs were also performed on the selected environmental variables in the different vegetation types using SPSS v17 software (SPSS Inc., 2006). If the results of one-way ANOVA had a P-value > 0.05 , it was concluded that there were no significant differences between any of the sample means. If the result of one-way ANOVA was a P-value ≤ 0.05 , it was concluded that at least two of the samples have significantly

different means. Where the ANOVAs were not significant, the analysis was terminated. However, where the result was significant, it was investigated further using multi comparison tests.

Classification trees, using CHAID growth limits (SPSS Inc., 2006) were used to determine the relationship between the vegetation types in terms of species richness and abundance relative to environmental variables. Significance level for splitting nodes and merging categories was 0.05, and the significance values were adjusted using the Bonferroni method. Correlations between species richness and abundance with environmental variables for all vegetation types, separately and collectively, were calculated using Spearman's Rank Order Coefficient, since the data were not normally distributed.

The statistical package of choice for multivariate analysis on arthropod and environmental datasets was Primer Ver. 5 (Clarke and Gorley, 2001). Primer software package had been used by ecologists worldwide to investigate the similarities and patterns of biological communities and environmental variables (e.g. Gutt and Starmans, 1998; O'Hara, 2001). In this study, multivariate analysis was successfully used to detect trends in samples and to explore the differences in arthropod assemblages between different vegetation types. Extensive use was made of Bray-Curtis similarity coefficients to derive similarity matrices of arthropod data, then clustering dendrograms and non-metric, multi-dimensional scaling (MDS) ordination plots to detect trends in similarity.

MDS is one of the best ordination techniques available (Everitt, 1978). The main advantage of MDS is its greater ability to represent more complex relations accurately in low-dimensional space (Clarke and Warwick, 2001). If the stress is low (<0.1), an MDS ordination is probably a more useful representation than a cluster analysis, with no prospect of misinterpretation (Clarke and Warwick, 2001). Joint absences in the MDS were ignored to emphasize similarity in common or rare species, comparing only percentage composition (Clarke and Warwick 2001). The dendrograms were comprised of group-average linking on Bray-Curtis species similarities from standardised abundance data.

Principal component analysis (PCA) is a multivariate method. There is no null hypothesis for basic PCA. PCA identifies which characteristics vary most between samples. PCA analysis was used to determine whether insects from the different vegetation types fall into distinct groups, which might result in distinct subspecies (Clarke and Warwick, 2001). The PCA technique considers a different starting point by making different assumptions about the definition of (dis) similarity of samples being compared. Species that were less common were excluded from the PCA analysis, so that the species retained were more comparable with the number of samples (Clarke and Warwick 2001). Distances between vegetation types on the ordination attempt to match the corresponding dissimilarities in arthropod assemblage composition: similar vegetation types would have very similar arthropod assemblages, while vegetation types which are dissimilar would have few species in common, or the same species at very different levels of abundance. However, PCA often produces inconclusive results, so the data were further utilised for cluster analysis, which is common practice for understanding how vegetation types vary and how they are grouped in terms of arthropod assemblages.

Cluster analysis is straightforward to apply, and produces results that broadly agree with PCA (Clarke and Warwick, 2001). In most cases, it is much easier to observe grouping in cluster analysis than in PCA. Nevertheless, it may be easier to understand what the groupings indicate if they have been produced by PCA. Therefore, using a combination of cluster analysis and PCA is an effective approach.

Analysis of similarity (ANOSIM) allows testing of the hypothesis that there is no assemblage differences between groups of samples specified; vegetation type in this case (Clarke and Warwick, 2001). ANOSIM tests were employed to test for significance differences between arthropod assemblages in fynbos, IATs, CIATs, and vineyard vegetation types. R is approximately zero if the null hypothesis is true, indicating that similarities between and within vegetation types will be the same on average (Clarke and Warwick 2001).

Various diversity indices (d = Margalef's index, a measure of the number of species present for a given number of individuals; J = Pielou's evenness index ($J' = H'/H'max$), where $H'max$ is the maximum possible value of Shanon diversity that would be achieved if all species were equally abundant; H' = Shanon diversity index; and $1-$

Lambda' = Simpson index) were calculated for arthropods in the different vegetation types using PRIMER v5.

As a result of large number of species sampled, a more automatic, analytical procedure for identifying influential species (discriminating species) was carried out. Average arthropod dissimilarity between inter-vegetation type samples (e.g. every sample in fynbos vegetation type paired with every sample in IATs vegetation type) and then this average was broken down into separate contributions from each species to average dissimilarity. If average dissimilarity of a particular species was large, and the ratio Diss/SD (similarity/standard deviation) was also large, then a particular species not only contributed much to the dissimilarity between two vegetation types, but it also did so consistently in inter-comparisons of all samples in the two vegetation types, thus being considered a good discriminating species.

RESULTS

Species richness and abundance

In total, 198 species from 106 families and 24 orders were recorded, from 25 225 individuals collected. Species estimators for all sites were: ICE = 203.83, Chao2 = 207.41 (SD \pm 7.25) and Jackknife2 = 215.96. Fynbos sites had 179 observed species from 104 families and 23 orders, with species estimator scores of: ICE = 186.42, Chao2 = 185.27 (SD \pm 4.33) and Jackknife2 = 193.04. Invasive alien tree sites (IATs) had 153 observed species from 90 families and 21 orders with species estimator scores of: ICE = 164.87, Chao2 = 164.46 (SD \pm 6.54) and Jackknife2 = 177.93. Cleared invasive alien trees (CIATs) sites had 169 observed species from 94 families and 23 orders with species estimator scores of: ICE = 176.99, Chao2 = 177.43 (SD \pm 5.51) and Jackknife2 = 187.95. Vineyard sites had 159 observed species from 87 families and 23 orders with species estimator scores of: ICE = 173.14, Chao2 = 177.6 (SD \pm 9.83) and Jackknife2 = 193.76.

Species accumulation curves for individual sites, fynbos sites, IATs, CIATs and vineyard, although flattening, did not quite reach an asymptote, despite the large number of individuals sampled (Figure 3.1). The lowest species accumulation curves were the IATs and vineyard site curves, with fynbos and CIATs curves showing a similar, higher pattern. Nevertheless, an overall species accumulation curve did reach

asymptotes after counting over 400 samples (Figure 3.2) and 20 000 individuals (Figure 3.3).

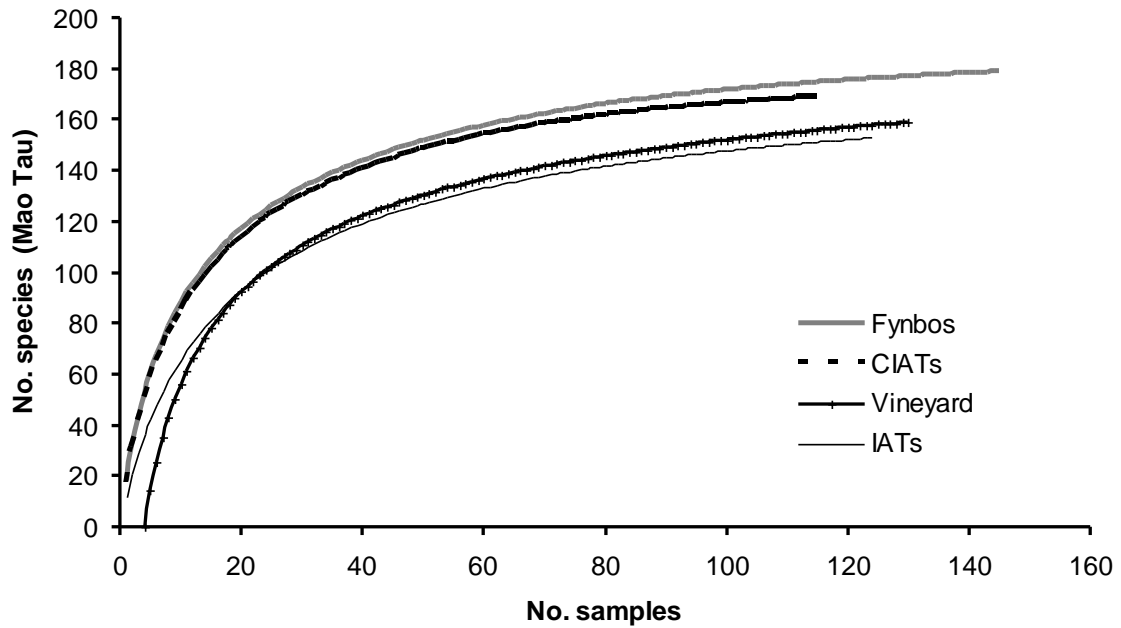


Figure 3.1 Sample rarefaction curves for the natural fynbos vegetation, vegetation cleared of invasive alien trees (CIATs), vineyard and invasive alien trees (IATs) sites.

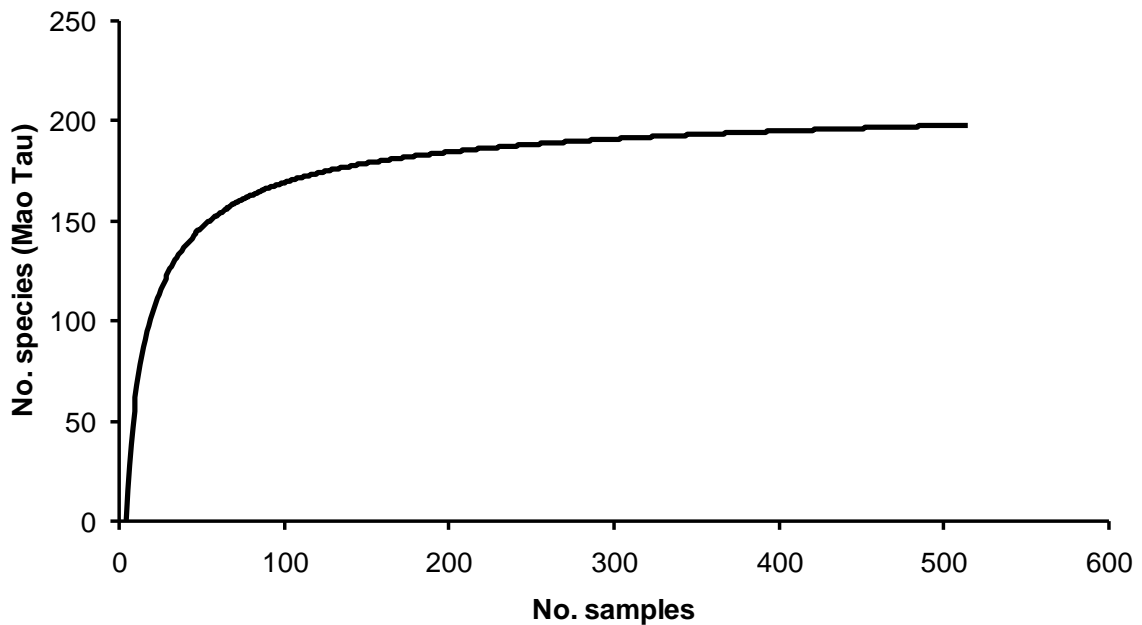


Figure 3.2 Sample rarefaction curves for all vegetation types combined.

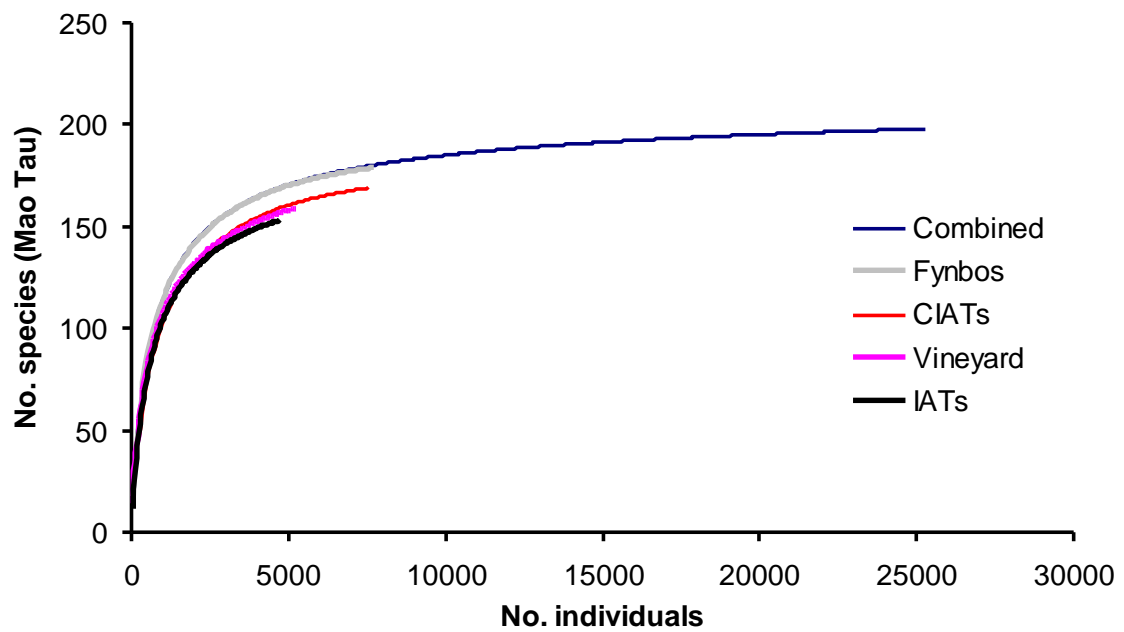


Figure 3.3 Sample rarefaction curves for all (combined) vegetation types, fynbos sites, sites cleared of invasive alien trees (CIATs), vineyard sites, and invasive alien trees (IATs) sites.

Arthropod species richness and abundance in the different vegetation types

Arthropod species richness and abundance were analyzed according to landscape context, i.e. relative to the type of neighbouring vegetation type. Nested ANOVAs revealed that there were significant differences in number of arthropod species and their abundance between different vegetation types (Table 3.1).

Table 3.1 Analysis of variance (ANOVA) for arthropod species richness and abundance between the different vegetation types.

Variable	ANOVA	Sum of Squares	df	Mean Square	F	Sig.
Species richness	Among adjacent vegetation types	4578.679	5	915.736	13.486	0.000
Species abundance	Among adjacent vegetation types	481878.095	5	96375.619	23.338	0.000

Effect of vegetation type on arthropod species richness and abundance

Fynbos and CIATs supported relatively higher species richness and abundance respectively (Table 3.2). IATs and vineyard sites had the least number of species and abundance respectively (Table 3.2). ANOVA among fynbos, IATs, CIATs, and vineyard sites showed that there were significant differences among ($df = 3, f = 41.645, p = 0.00$) and within ($df = 509, f = 41.645, p = 0.00$) vegetation in terms of arthropod species richness.

Table 3.2 Species richness and abundance means, standard deviation (Std.Dev), variance and standard error (Std.Err.) for fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites.

Variable	Vegetation	Means	N	Std.Dev.	Variance	Std.Err.
Species richness	Cleared IATs	18.130	115	5.736	32.904	0.535
	Fynbos	18.062	145	6.284	39.489	0.522
	IATs	11.573	124	4.601	21.174	0.413
	Vineyard	14.056	126	5.547	30.773	0.494
	<i>All vegetation types</i>	<i>15.510</i>	<i>510</i>	<i>6.239</i>	<i>38.922</i>	<i>0.276</i>
Species abundance	Cleared IATs	65.765	115	39.252	1540.778	3.660
	Fynbos	53.421	145	33.297	1108.732	2.765
	IATs	38.105	124	26.496	702.030	2.379
	Vineyard	41.437	126	25.885	670.024	2.306
	<i>All vegetation types</i>	<i>49.520</i>	<i>510</i>	<i>33.239</i>	<i>1104.824</i>	<i>1.472</i>

Nested ANOVAs among fynbos, IATs, CIATs, and vineyard site species richness revealed significant differences between fynbos and IATs, fynbos and vineyard, CIATs and IATs, CIATs and vineyard. However, there were no statistically significant differences between CIATs and fynbos vegetation in terms of arthropod species richness (Table 3.3). Although vineyards had relatively higher mean species abundance than IATs (Table 3.2), there was no significant difference between them in terms of species abundance (Table 3.3). Details on species are given in Appendix 3a.

Table 3.3 Nested analysis of variance (ANOVA) among fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard site species richness.

*indicates that the mean difference is significant at the 0.05 level.

Variable	Vegetation (I)	Vegetation (J)	(I-J) Mean species richness		Sig.	95% Confidence Interval	
			difference	Std. Error		Lower Bound	Upper Bound
Species richness	Fynbos	IATs	6.489*	0.685	0.000	4.67	8.30
		Vineyard	4.007*	0.682	0.000	2.20	5.81
	CIATs	Fynbos	0.068	0.700	1.000	-1.78	1.92
		IATs	6.558*	0.725	0.000	4.64	8.48
	Vineyard	IATs	4.075*	0.723	0.000	2.16	5.99
Species abundance	Fynbos	IATs	15.316*	3.866	0.001	5.08	25.55
		Vineyard	11.984*	3.849	0.012	1.79	22.18
	CIATs	Fynbos	12.345*	3.946	0.011	1.89	22.80
		IATs	27.660*	4.092	0.000	16.82	38.50
	Vineyard	IATs	24.329*	4.076	0.000	13.53	35.12
Vineyard	IATs	3.332	3.998	1.000	-7.26	13.92	

Arthropods restricted to a particular vegetation type(s)

Some of the sampled arthropods (77) were not recorded from at least one out of four possible vegetation types (Table 3.4). Few of these arthropod species were restricted to only one vegetation type: fynbos (6); IATs (4); CIATs (3) and the vineyard (5). Most of these restricted arthropods were recorded from fynbos (77%), followed by CIATs (48%), vineyard (38%) and IATs (32%) respectively.

Table 3.4 Sampled arthropods restricted to a particular vegetation type(s): fynbos, invasive alien trees (IATs), cleared of invasive alien trees (CIATs) and vineyard. ‘+’ refers to species present, while ‘-’ refers to species absent.

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard
Araneae					
Gnaphosidae	<i>Camillina</i> sp.2	-	+	-	+
Lycosidae	<i>Geolycosa</i> sp.1	+	-	+	-
Nemesiidae	<i>Pionothele straminae</i>	+	-	+	+
Salticidae	undetermined genus	+	-	+	+

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard
Theridiidae	<i>Theridion</i> sp.	+	-	-	+
Thomisidae	<i>Ozyptila</i> sp.	-	+	-	-
Trochanteriidae	<i>Platyoides</i> sp.	+	-	-	-
Coleoptera					
Cantharidae	<i>Cantharis</i> sp.1	+	-	+	+
Carabidae	<i>Anthia decemquttata</i>	+	+	-	-
Carabidae	<i>Graphipterus trilineatus</i>	+	-	-	+
Carabidae	<i>Passalidius fortipes</i>	+	-	-	-
Chrysomelidae	<i>Leptinotarsa</i> sp.1	+	-	+	+
Cicindelinae	<i>Platydelia quadriguttata</i>	+	-	-	-
Coccinellidae	<i>Harmonia axyridis</i>	-	+	+	+
Dermestidae	<i>Dermestes maculatus</i>	+	+	+	-
Elateridae	<i>Cardiotarsus acuminatus</i>	+	+	+	-
Lampyridae	<i>Lampyrus disticollis</i>	-	+	+	-
Lampyridae	<i>Luciola</i> sp.	+	-	+	-
Lycidae	Sp.1	+	-	-	-
Meloidae	<i>Decapotoma lunata</i>	+	+	-	+
Melyridae	<i>Apalochrus</i> sp.1	+	+	+	-
Mordellidae	Sp.1	+	-	+	+
Scarabaeidae	<i>Aphodius</i> sp.	+	-	+	+
Silvanidae	<i>Oryzaephilus</i> sp.	+	-	+	+
Tenebrionidae	<i>Psammodes striatus</i>	-	-	-	+
Tenebrionidae	Sp.3	+	-	+	+
Tenebrionidae	<i>Zophosis</i> sp.1	+	-	+	-
Diptera					
Bombyliidae	<i>Exoprosopa</i> sp.	+	+	-	-
Calliphoridae	<i>Chrysomya chloropyga</i>	+	-	+	+
Calliphoridae	Sp.1	+	+	-	+
Cecidomyiidae	<i>Cecidomyia</i> sp.1	+	+	+	-
Heleomyzidae	<i>Helomyza picta</i>	+	+	+	-
Therividae	<i>Therevid</i> sp.	+	+	-	+
Hemiptera					
Cicadellidae	<i>Ciphalilus</i> sp.	+	-	+	+
Cicindelidae	<i>Cicindela quadriguttata</i>	+	+	+	-
Cydnidae	Sp.1	+	+	-	+
Pentatomidae	<i>Agonoscelis</i> sp.1	+	-	+	+
Pentatomidae	<i>Agonoscelis</i> sp.2	+	-	-	-
Pentatomidae	<i>Bagrada hilaris</i>	+	-	-	+
Pentatomidae	<i>Coenomorpha</i> sp.	+	-	+	-
Reduviidae	<i>Acanthaspis sanguinosa</i>	+	-	-	-
Reduviidae	<i>Ectrichodia crux</i>	+	-	+	+
Reduviidae	<i>Holoptilus</i> sp.1	-	-	+	-
Reduviidae	?Oncocephalinae sp.	+	+	-	+
Piesmatidae	Sp.1	-	-	-	+
Hymenoptera					
Anthophoridae	<i>Xylocopa</i> sp.	+	+	-	-
Apidae	<i>Meliponula</i> sp.1	+	-	+	+
Braconidae	<i>Charops</i> sp.1	-	+	+	-
Chrysididae	<i>Spintharina</i> sp.	-	+	-	-
Formicidae	<i>Camponotus maculatus</i>	+	-	+	+
Formicidae	<i>Camponotus</i> sp.3	+	+	+	-
Formicidae	<i>Dorylus helvolus</i>	-	+	-	-
Formicidae	Sp.2	+	-	-	-
Halictidae	<i>Nomia amabilis</i>	+	+	+	-
Halictidae	Sp.2	+	-	-	+
Halictidae	Sp.3	-	+	+	+
Masaridae	<i>Ceramius</i> sp.1	-	-	+	-

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard
Melittidae	Sp.1	+	+	+	-
Scoliidae	<i>Campsomeriella</i> sp.1	+	+	+	-
Sphecidae	Sp.3	+	-	+	+
Sphecidae	Sp.4	-	-	+	-
Sphecidae	<i>Sphex tydei</i> var <i>capensis</i>	+	-	+	+
Lepidoptera					
Geometridae	Sp.1	+	-	+	+
Lasiocampidae	<i>Eutricha capensis</i>	-	-	+	+
Noctuidae	Sp.1	-	+	+	+
Psychidae	Sp.1	+	-	+	-
Sphingidae	<i>Coelonia fulvinotata</i>	-	-	-	+
Tineidae	Sp.1	+	+	-	+
Tortricidae	Sp.1	-	+	-	-
Mantodea					
Hymenopodidae	<i>Harpagomantis tricolor</i>	+	-	+	-
Mantidae	<i>Polyspilota aeruginosa</i>	-	-	-	+
Mantidae	Sp.2	+	-	+	+
Mantidae	Sp.3	+	+	+	-
Neuroptera					
Nemopteridae	<i>Laurhervasia setacea</i>	+	-	+	+
Lithobiomorpha					
Peripatopsidae	<i>Peripatopsis</i> sp.	+	+	+	-
Scorpiones					
Scorpionidae	<i>Opisthophthalmus macer</i>	+	+	+	-
Solifugae					
Eremobatidae	<i>Eremobates</i> sp.1	+	-	+	-
TOTAL +		59	32	48	38
TOTAL -		18	45	29	39
% TOTAL +		77	42	62	49

Classification of vegetation type and sites according to arthropod species richness and abundance

The classification tree for vegetation, where significance values were adjusted using Bonferroni method, indicated similarity between natural fynbos sites and sites cleared of invasive alien trees (Figure 3.4). In turn, IATs and vineyards were significantly different from each other, and both were not comparable to either fynbos or CIATs.

As there were significant differences between vegetation types for arthropod species richness and abundance (Table 3.3), the classification of sites was also investigated. Classification of site locations based on species richness resulted in four nodes (Figure 3.5). Several site locations (Nodes 2 and 3) provided significantly similar number of species (Figure 3.5). However, Vergelegen sites (Node 1) provided significantly higher number of species than any other location. The lowest number of mean species richness was from Bilton (Node 4) (Figure 3.5).

Classification of different vegetation types in terms of arthropod abundance resulted in three different nodes (Figure 3.6). IATs and vineyards clustered together and had significantly lower species abundance than fynbos and CIATs vegetation types.

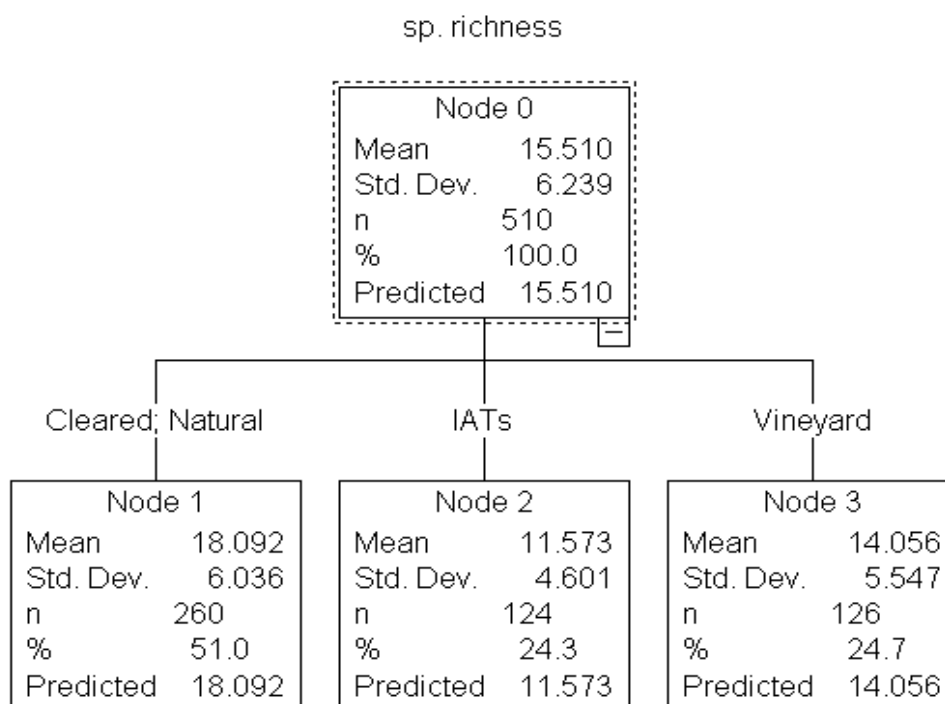


Figure 3.4 Classification tree of the four vegetation types (at all locations combined) for mean arthropod species richness. Cleared = vegetation cleared of invasive alien trees, natural = fynbos, IATs = invasive alien trees.

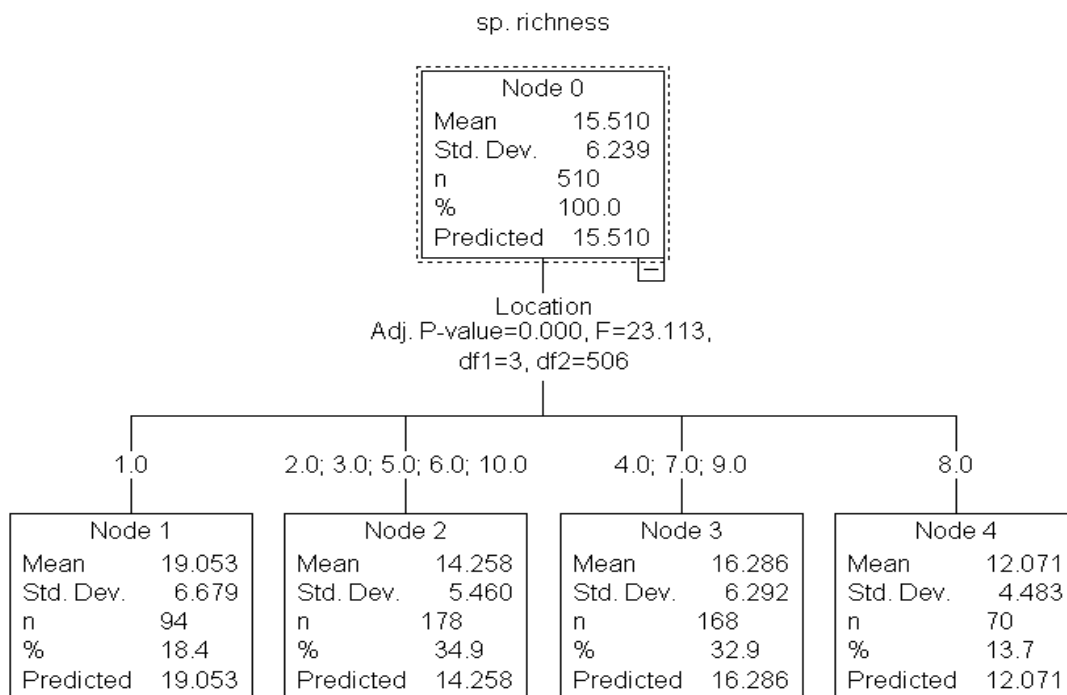


Figure 3.5 Classification tree of site locations in terms of arthropod species richness. 1.0 = Vergelegen, 2.0 = Stellenzichdt-Driekoppen, 3.0 = Waterford, 4.0 = Rustenberg, 5.0 = Waterford-Driekoppen, 6.0 = Helderberg, 7.0 = Hottentots Holland, 8.0 = Bilton, 9.0 = Dornier, 10.0 = Jonkershoek.

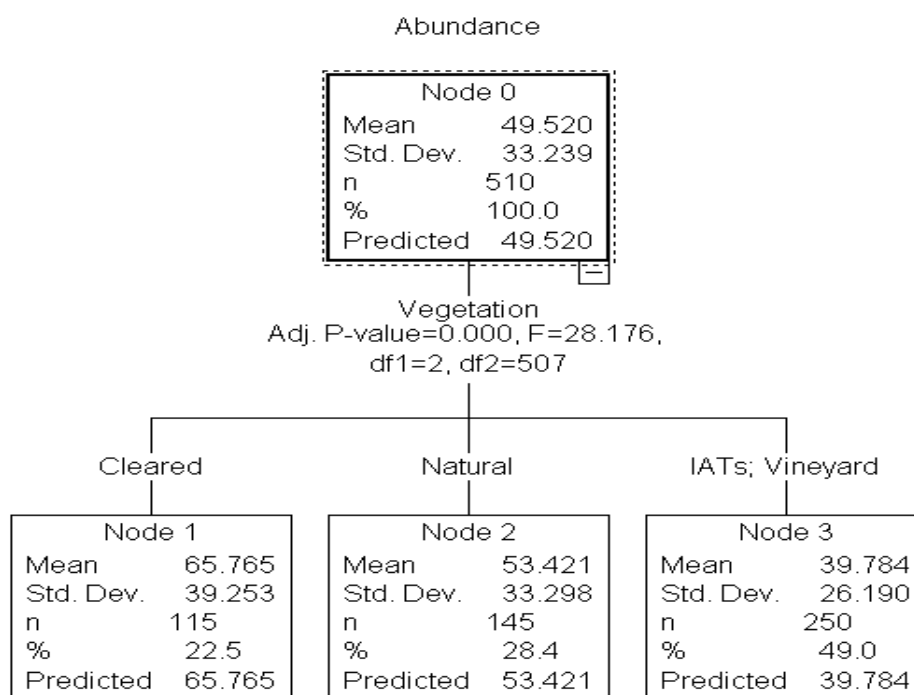


Figure 3.6 Classification tree of vegetation in terms of arthropod species abundance. Cleared = vegetation cleared of invasive alien trees, natural = fynbos, IATs = invasive alien trees.

Arthropod assemblage composition

The Cluster (Figure 3.7) and MDS (Figure 3.8) analyses highlight the potential for differences in arthropod assemblages in the four vegetation types. Figure 3.7 displays the results of a cluster analysis on vegetation types based on arthropod assemblage composition at an arbitrary similarity level of around 76%. Figure 3.8 shows the 2-dimensional MDS plot of the same species composition similarities. The groups determined from the cluster analysis were superimposed and indicate a good measure of agreement. Vegetation type grouped according to arthropod assemblage composition with high percentage similarities.

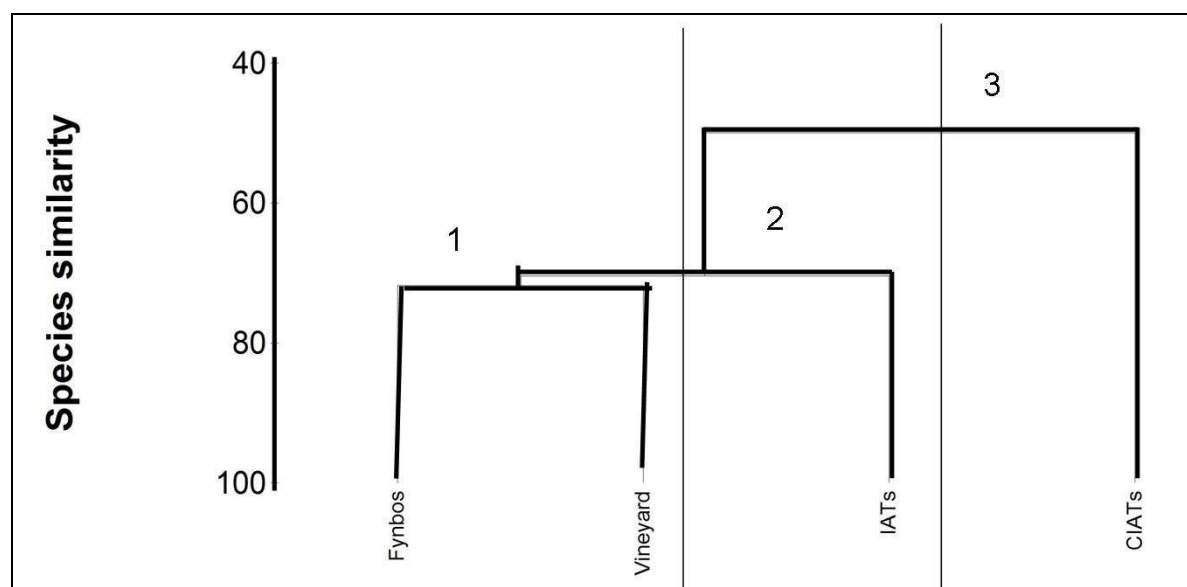


Figure 3.7 Classification tree of vegetation types in terms of arthropod assemblages using group-average linking on Bray-Curtis species similarities. Cleared = vegetation cleared of invasive alien trees, natural = fynbos, IATs = invasive alien trees.

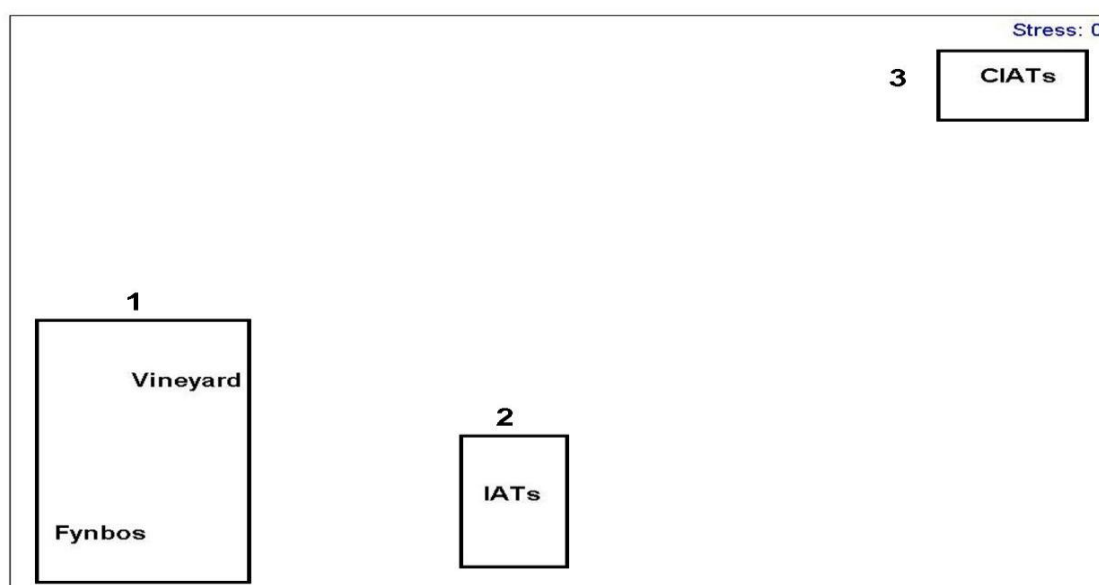


Figure 3.8 Multi-dimensional scaling (MDS) analysis of Bray-Curtis similarity between arthropod samples from different vegetation types (i.e. fynbos, vineyard, invasive alien trees (IATs), vegetation cleared of invasive alien trees (CIATs)).

There was a good separation of vegetation types based upon their arthropod assemblage composition, as shown by MDS analysis (Figure 3.8). These indicated that there were

no clear similarities among different vegetation types based on assemblage composition. However, fynbos and vineyard had more comparable arthropod assemblage composition. CIAT vegetation type was more different from other vegetation types based on arthropod assemblage composition.

Testing the significance of assemblage composition differences

The null hypothesis for these ANOSIM test statistics was that there were no significant differences in arthropod assemblage composition between the sample groups collected from fynbos, vineyard, IATs, and CIATs vegetation types. In each case, it was only possible to test arthropod assemblage composition between two vegetation types at a time.

The global test of the null hypothesis based on arthropods sampled from different vegetation types was rejected. This means that the arthropod assemblage composition between vegetation types was not similar, and that assemblage groups could be distinguished on arthropod assemblage composition. However, when vegetation types were grouped by assemblage composition, there were statistically significant differences between these vegetation types ($R = 0.149$, $P = 0.001$). Therefore, there were vegetation-by-vegetation similarities worth investigating. This was done by completing the pair-wise test for the vegetation type comparisons. The result was that the null hypothesis was rejected when comparing vegetation types in each case (Table 3.5).

The spread of R-values possible from four different vegetation types can be seen in Figure 3.9. The largest of $T = 209$ simulations was less than 0.149 ($t = 0$). An observed value of $R = 0.149$ was seen to be a most unlikely event, with a probability of less than 1 in 1000 if null hypothesis (H_0) is true. Therefore, H_0 was rejected at a significance level of $P < 0.001$. Nevertheless, $R = 0.149$ may still have been the most extreme outcome observed if larger number of simulations was chosen.

Table 3.5 Statistically significant comparisons based upon arthropod assemblage composition in fynbos, vineyard, IATs, and CIATs vegetation types.

Vegetation types	R Statistic	Significance level	Possible Permutations	Actual Permutations	Number \geq Observed
CIATs vs Fynbos	0.035	0.002	Too many	999	1
CIATs vs IATs	0.141	0.001	Too many	999	0
CIATs vs Vineyard	0.166	0.001	Too many	999	0
Fynbos vs IATs	0.196	0.001	Too many	999	0
Fynbos vs Vineyard	0.175	0.001	Too many	999	0
IATs vs Vineyard	0.178	0.001	Too many	999	0
IATs vs Vineyard	0.178	0.001	Too many	999	0

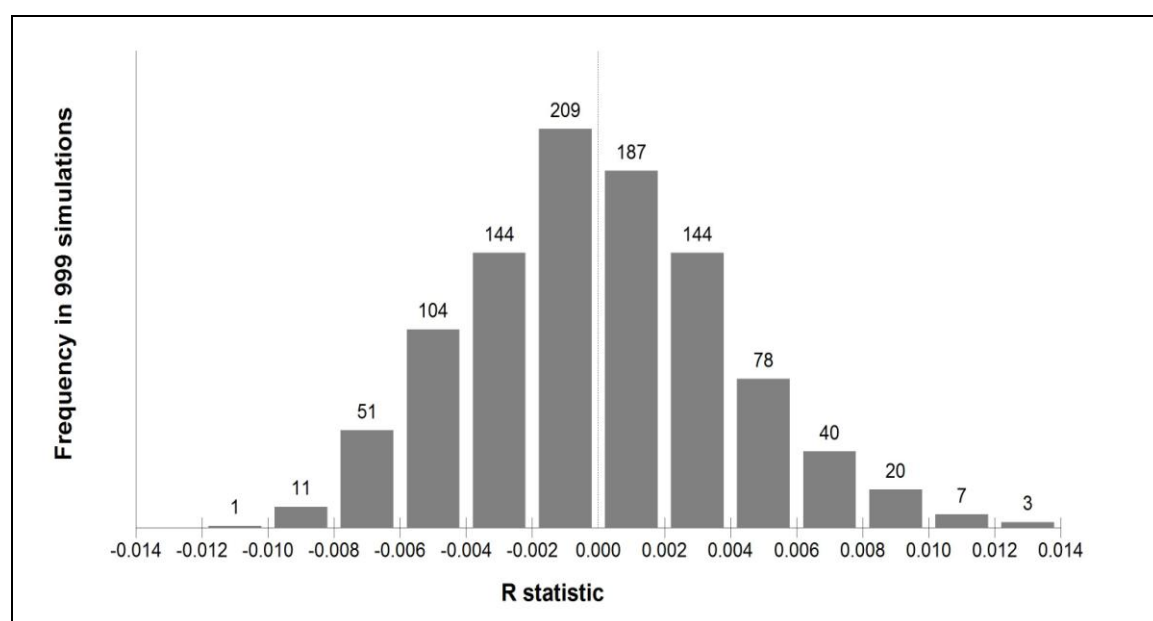


Figure 3.9 Simulated distribution of the test statistic R under the null hypothesis of ‘no vegetation type differences’. This contrasted with an observed value for R of 0.149.

Arthropod assemblage composition in IATs was very different from those in other vegetation types (Figure 3.10). Vineyard and CIATs vegetation types occur close to each other, indicating that they are comparable in terms of arthropod assemblage composition. Furthermore, fynbos vegetation was also separated from other vegetation types, indicating its dissimilarity from other vegetation types based on assemblage composition.

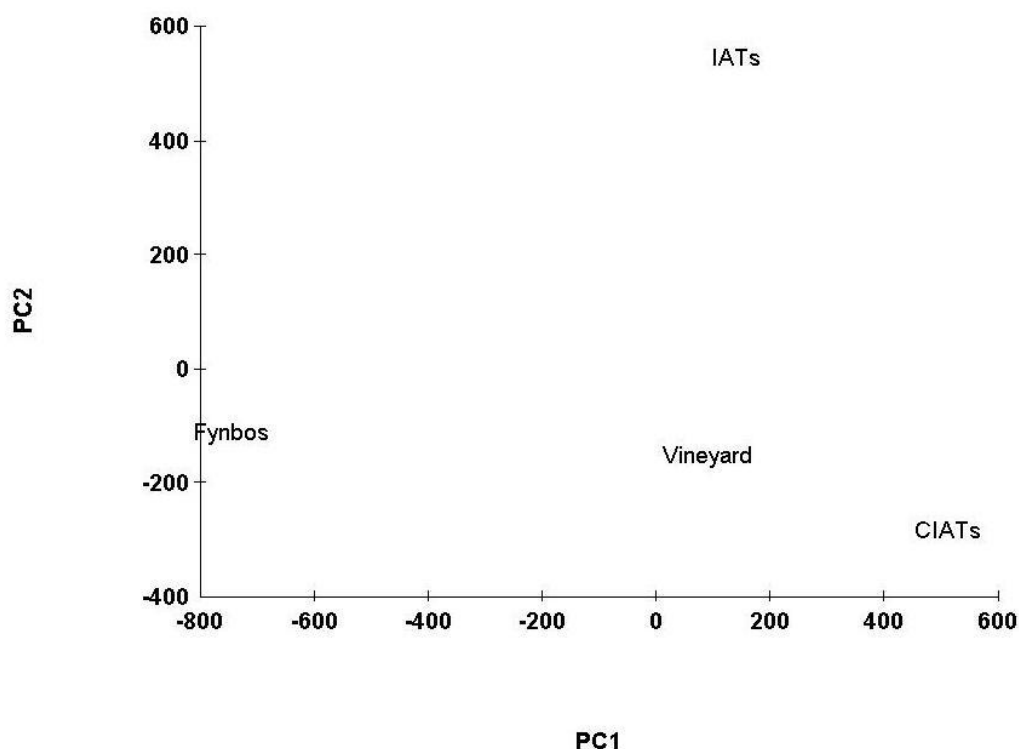


Figure 3.10 Principal components analysis (PCA) ordination of Bray-Curtis similarity between arthropod samples from different vegetation types (i.e. fynbos, vineyard, invasive alien trees (IATs), vegetation cleared of invasive alien trees (CIATs)).

Diversity measures

Highest arthropod diversity was measured in fynbos, and the least in IATs. Both Shannon diversity index (H') and Simpson index ($1-\lambda$) measured higher diversity for the vineyard when compared to the IATs (Table 3.6).

Table 3.6 Univariate diversity indices (d= Margalef's index, J= Pielou's evenness index, H'= Shanon diversity index and 1-Lambda' = Simpson index) for fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites.

Sample	d	J'	H'(loge)	1-Lambda'
Fynbos	19.99	0.7832	4.067	0.9649
IATs	14.15	0.7565	3.468	0.9332
CIATs	18.2	0.7487	3.776	0.947
Vineyard	18.57	0.8004	4.062	0.968

Selected environmental variables influencing arthropod biodiversity

The largest leaf litter depth was recorded from IATs, while vineyards supported a very low leaf litter depth (Table 3.7). As expected, vineyard supported the lowest percentage soil moisture followed by CIATs, whereas fynbos had the highest (Table 3.7).

ANOVA of fynbos, IATs, CIATs, and vineyards showed that there were significant differences among (df = 3, f = 5.81, p = 0.01), and within (df = 509, f = 5.81, p = 0.01) sites in terms of percentage soil moisture. There were also significant differences, among fynbos, IATs, CIATs, and vineyards in terms of leaf litter (df = 3, f = 296.6, p = 0.000). Vineyard sites had significantly different soil moisture content compared to fynbos and IATs, but comparable with CIATs (Table 3.8). All vegetation types were significantly different in terms of leaf litter depth (Table 3.8).

Table 3.7 Environmental variable means, standard deviation (Std.Dev), variance and Standard error (Std.Err.) for fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites.

Environmental variable	Vegetation	Means	N	Std.Dev.	Variance	Std.Err.
% Soil moisture	CIATs	8.778	115	2.918	8.517	0.272
	Fynbos	9.761	145	3.322	11.037	0.276
	IATs	9.491	124	3.344	11.184	0.300
	Vineyard	8.362	126	2.502	6.262	0.223
	<i>All vegetation types</i>	<i>9.128</i>	<i>510</i>	<i>3.096</i>	<i>9.590</i>	<i>0.137</i>
Leaf litter depth (mm)	CIATs	13.652	115	5.535	30.632	0.516
	Fynbos	15.572	145	4.641	21.538	0.385
	IATs	20.798	124	6.437	41.431	0.578
	Vineyard	2.738	126	2.367	5.603	0.211
	<i>All vegetation types</i>	<i>13.239</i>	<i>510</i>	<i>8.194</i>	<i>67.149</i>	<i>0.363</i>
Elevation (m)	CIATs	390.480	115	226.629	51361.030	28.553
	Fynbos	372.960	145	249.372	62186.590	23.054
	IATs	320.290	124	202.614	41052.230	19.964
	Vineyard	304.470	126	64.916	4214.040	7.398
	<i>All vegetation types</i>	<i>346.310</i>	<i>510</i>	<i>206.551</i>	<i>42663.320</i>	<i>10.886</i>

Table 3.8 Nested analysis of variances (ANOVA) among natural fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites in terms of percentage (%) soil moisture.

Dependent Variable	(I) Vegetation	(J) Vegetation	(I-J) Mean Difference	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
% Soil moisture	Fynbos	IATs	0.270	0.373	1.000	-0.719	1.260
		CIATs	0.983	0.381	0.061	-0.027	1.993
		Vineyard	1.399*	0.372	0.001	0.414	2.385
	IATs	CIATs	0.712	0.395	0.432	-0.334	1.760
		Vineyard	1.129*	0.386	0.022	0.106	2.152
		CIATs	0.416	0.394	1.000	-0.627	1.459
Leaf litter depth (mm)	Fynbos	CIATs	1.920*	0.618	0.012	0.280	3.560
		Vineyard	12.834*	0.603	0.000	11.240	14.430
		IATs	5.226*	0.605	0.000	3.620	6.830
	IATs	CIATs	7.146*	0.641	0.000	5.450	8.840
		Vineyard	18.060*	0.626	0.000	16.400	19.720
		CIATs	10.914*	0.638	0.000	9.220	12.600

* indicates that the mean difference is significant at the 0.05 level

Arthropod species and environmental variable correlations

Overall, there were significant correlations between site location and species richness (Spearman's Coefficient = -0.240; $p = 0.000$, $n = 510$), as well as abundance (Spearman's Coefficient = -0.218; $p = 0.000$, $n = 510$). However, there were no significant correlations between percentage soil moisture and species richness (Spearman's Coefficient = 0.05; $p = 0.919$, $n = 510$), as well as abundance (Spearman's Coefficient = 0.061; $p = 0.171$, $n = 510$). Moreover, there were no significant correlations between leaf litter depth and species richness (Spearman's Coefficient = -0.20; $p = 0.651$, $n = 510$), nor abundance (Spearman's Coefficient = -0.033; $p = 0.454$, $n = 510$). Nevertheless, there was significant correlation between species richness and abundance (Spearman's Coefficient = 0.696; $p = 0.000$, $n = 510$).

Arthropod variation during sampling periods

Highest mean species richness was recorded during sampling period 1 (August-October) for both fynbos, IATs, CIATs and vineyards (Table 3.9). However, there were significant differences between all four vegetation types in terms of arthropod species richness during sampling period 1 ($df = 3$, $f = 8.774$, $p = 0.000$), sampling period 2 (November – January) ($df = 3$, $f = 14.601$, $p = 0.000$) and sampling period 3 (May-July) ($df = 3$, $f = 10.374$, $p = 0.000$). In the fynbos, there was a steady decline between sampling period 1 and 2, where there was a steady decline in numbers of sampled species. Species richness for sampling period 2 was relatively high, but further declined in sampling period 3 (Table 3.9). In sampling period 2, there was also a significant decline in species richness between fynbos and IATs, after which there was a rapid increase in CIATs. Nevertheless, there was no significant difference between CIATs and vineyard vegetation types ($p = 1.000$) (Figure 3.11). For sampling period 3, fynbos and CIATs were not significantly different ($p = 1.000$) whereas IATs was more comparable with vineyard vegetation ($p = 1.000$) (Figure 3.11). Overall mean species richness for all the sampling periods was 15.51, with a S.D. of 6.2.

Table 3.9 Arthropod species richness means, standard deviation and Standard error (Std. Err.) for fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard vegetation types during sampling periods 1-3.

Sampling period	Vegetation	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean			
						Lower Bound	Upper Bound	Min	Max
1 (Aug-Oct)	Fynbos	145	7.500	4.764	0.396	6.720	8.290	0	25
	IATs	124	5.230	3.343	0.300	4.640	5.830	0	13
	CIATs	115	8.040	5.879	0.548	6.960	9.130	0	20
	Vineyard	126	6.400	4.431	0.395	5.620	7.180	0	22
	<i>Total</i>	<i>510</i>	<i>6.800</i>	<i>4.776</i>	<i>0.211</i>	<i>6.380</i>	<i>7.220</i>	<i>0</i>	<i>25</i>
2 (Nov-Jan)	Fynbos	145	5.050	3.606	0.299	4.460	5.640	0	15
	IATs	124	2.980	2.225	0.200	2.580	3.370	0	10
	CIATs	115	5.300	3.635	0.339	4.630	5.980	0	17
	Vineyard	126	5.220	3.271	0.291	4.650	5.800	0	17
	<i>Total</i>	<i>510</i>	<i>4.650</i>	<i>3.372</i>	<i>0.149</i>	<i>4.350</i>	<i>4.940</i>	<i>0</i>	<i>17</i>
3 (May-Jul)	Fynbos	145	4.870	3.740	0.311	4.260	5.480	0	17
	IATs	124	3.630	2.158	0.194	3.250	4.010	0	10
	CIATs	115	5.380	3.350	0.312	4.760	6.000	0	13
	Vineyard	126	3.680	2.382	0.212	3.260	4.100	0	14
	<i>Total</i>	<i>510</i>	<i>4.390</i>	<i>3.089</i>	<i>0.137</i>	<i>4.120</i>	<i>4.660</i>	<i>0</i>	<i>17</i>

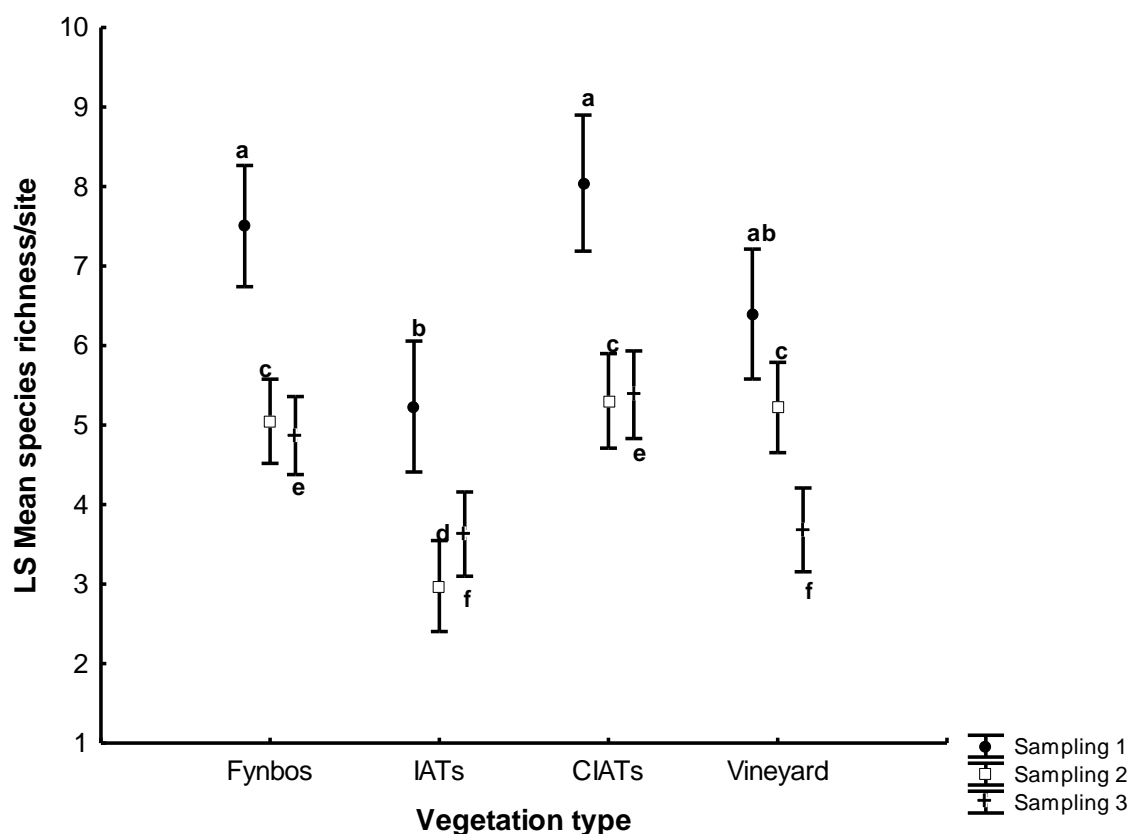


Figure 3.11 Mean arthropod species richness over sampling period 1 (August-October months), 2 (November-January months), and 3 (May-July months) across different vegetation types. Vegetation types: IATs = invasive alien trees, CIATs = vegetation cleared of invasive alien trees.

Species typifying a fynbos vegetation type

Table 3.10 shows the contributions of each species to the Bray-Curtis similarity within a fynbos vegetation type. The Bray-Curtis similarity between all pairs of sites in the fynbos vegetation type was 16.32, made up mainly of the contributions from eight species: *Camponotus* sp.2 (3.81, i.e. 23.33%), *Linepithema humile* (1.34, i.e. 8.18%), *Gryllus bimaculatus* (1.05, i.e. 6.45%), *Microhodotermes viator* (1.02, i.e. 6.23%), *Centrobolus* sp.2 (0.80, i.e. 4.93%), *Zophosis boei* (0.74, i.e. 4.55%), *Diores youngai* (0.74, i.e. 4.52%), *Tetramorium capense* (0.73, i.e. 4.45%), with a cumulative contribution of over 60% of the total similarity of 16.32. However, the list includes only the top species before 90% cumulative percentage was reached. These arthropod species can be identified as typical of fynbos. The lower ratio of their contribution to

the SD was an indication that they were inconsistently present in large numbers in fynbos vegetation (Clarke and Gorley, 2006).

Table 3.10. Average similarities between arthropod samples at fynbos vegetation type. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 16.32.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Camponotus</i> sp.2	5.23	3.81	0.64	23.33	23.33
<i>Linepithema humile</i>	2.48	1.34	0.45	8.18	31.51
<i>Gryllus bimaculatus</i>	1.35	1.05	0.51	6.45	37.96
<i>Microhodotermes viator</i>	5.35	1.02	0.21	6.23	44.19
<i>Centrobolus</i> sp.2	2.80	0.80	0.27	4.93	49.12
<i>Zophosis boei</i>	2.19	0.74	0.37	4.55	53.67
<i>Diores youngai</i>	1.28	0.74	0.47	4.52	58.19
<i>Tetramorium capense</i>	1.66	0.73	0.32	4.45	62.64
<i>Porcello</i> sp.	2.13	0.68	0.24	4.15	66.79
<i>Crematogaster peringueyi</i>	2.07	0.54	0.26	3.29	70.08
<i>Daspletis</i> sp.	1.07	0.45	0.34	2.78	72.86
<i>Microlestia tabida</i>	0.86	0.23	0.22	1.42	74.28
<i>Oxyopes</i> sp.	0.54	0.20	0.27	1.25	75.53
<i>Synema imitator</i>	0.79	0.20	0.16	1.23	76.76
<i>Cophogryllus</i> sp.	0.43	0.18	0.24	1.11	77.87
<i>Achmaedera</i> sp.	0.59	0.18	0.23	1.08	78.95
Hopliini sp.	0.80	0.14	0.15	0.85	79.80
<i>Cleptria rufipes</i>	0.34	0.13	0.24	0.81	80.61
Formicidae sp.3	0.93	0.13	0.11	0.77	81.38
<i>Langona</i> sp.1	0.32	0.13	0.20	0.77	82.15
Sphecidae sp.2	0.39	0.11	0.21	0.67	82.82
<i>Galeodes</i> sp.2	0.31	0.10	0.19	0.64	83.46
<i>Proevippa schreineri</i>	0.28	0.10	0.19	0.61	84.07
<i>Galeodes</i> sp.1	0.32	0.10	0.21	0.59	84.65
<i>Centrobolus</i> sp.1	0.46	0.10	0.15	0.59	85.24
Tenebrionidae sp.1	0.54	0.09	0.15	0.58	85.82
<i>Evarcha</i> sp.1	0.37	0.09	0.22	0.58	86.40
Paederinae sp.	0.32	0.09	0.17	0.52	86.92
<i>Steatoda</i> sp.2	0.28	0.08	0.16	0.50	87.42
<i>Temnopteryx phalerata</i>	0.30	0.08	0.20	0.49	87.92
<i>Aelurillus</i> sp.1	0.26	0.08	0.18	0.49	88.40
<i>Proevippa</i> sp.1	0.31	0.08	0.17	0.47	88.88
Pompilidae sp.1	0.30	0.08	0.18	0.46	89.34
<i>Tenebrio</i> sp.1	0.30	0.07	0.20	0.45	89.78
<i>Diaphorocellus</i> sp.1	0.28	0.07	0.18	0.42	90.21

Species typifying IATs vegetation type

Table 3.11 shows the contributions of each species to the Bray-Curtis similarity within the IAT vegetation type. The Bray-Curtis similarity between all pairs of sites in the IATs vegetation type was 13.05, made up mainly of the contributions from three species: *Centrobolus* sp.2 (3.77, i.e. 28.92%), *Porcello* sp. (2.49, i.e. 19.09%) *G. bimaculatus* (0.73, i.e. 5.57%), with a cumulative contribution of over 50% of the total similarity of 13.05. However, the list includes only the top species before 90% cumulative percentage was reached. These arthropod species can be identified as typical of IATs (Clarke and Gorley, 2006).

Table 3.11. Average similarities between arthropod samples at IATs vegetation type. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 13.05.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Centrobolus</i> sp.2	5.92	3.77	0.47	28.92	28.92
<i>Porcello</i> sp.	4.59	2.49	0.36	19.09	48.02
<i>Gryllus bimaculatus</i>	0.75	0.73	0.39	5.57	53.58
<i>Tetramorium capense</i>	1.27	0.64	0.27	4.94	58.52
<i>Diores youngai</i>	0.69	0.45	0.29	3.49	62.01
Sciaridae sp.1	1.19	0.44	0.22	3.36	65.37
<i>Centrobolus</i> sp.1	0.91	0.43	0.18	3.31	68.68
<i>Camponotus</i> sp.2	0.87	0.42	0.23	3.23	71.92
<i>Crematogaster peringueyi</i>	1.06	0.42	0.21	3.19	75.10
<i>Microhodotermes viator</i>	2.40	0.37	0.13	2.87	77.97
<i>Xestobium</i> sp.	0.88	0.28	0.18	2.17	80.14
Pipunculidae sp.1	1.02	0.27	0.15	2.10	82.24
<i>Linepithema humile</i>	0.80	0.23	0.14	1.74	83.98
<i>Cecidomyia</i> sp. 1	1.77	0.18	0.09	1.40	85.38
Sphecidae sp.2	0.31	0.10	0.16	0.76	86.14
Hopliini sp.	0.67	0.09	0.10	0.73	86.87
<i>Oxyopes</i> sp.	0.28	0.09	0.14	0.68	87.55
<i>Camillina</i> sp.1	0.26	0.09	0.15	0.68	88.23
<i>Tenebrio</i> sp.1	0.36	0.09	0.10	0.67	88.90
<i>Endomia elongates</i>	0.46	0.08	0.11	0.59	89.49
<i>Cophogryllus</i> sp.	0.19	0.06	0.12	0.45	89.94
Carabidae sp.1	0.29	0.05	0.11	0.40	90.33

Species typifying CIATs vegetation type

Table 3.12 shows the contributions of each species to the Bray-Curtis similarity within the CIATs vegetation type. The Bray-Curtis similarity between all pairs of sites in the CIATs vegetation type was 19.27, made up mainly of the contributions from just seven species: *M. viator* (3.74, i.e.18.93%), *Centrobolus* sp.2 (2.73, i.e. 13.82%), *Camponotus* sp.2 (2.72, i.e. 13.75%), *Z. boei* (1.68, i.e. 8.47%), *L. humile* (1.30, i.e. 6.58%), *Porcello* sp. (1.07, i.e. 5.42%), *T. capense* (1.07, i.e. 5.39%), with a cumulative contribution of 72.35 of the total similarity of 19.79. However, the list was truncated when 90% was reached. These arthropod species can be identified as typical of CIATs. The higher ratio (Sim/SD) of their contribution was an indication that they also have a consistently large presence in CIATs compared to those with lower Sim/SD ratio (Clarke and Gorley, 2006). *Camponotus* sp.2 and *Centrobolus* sp.2 are also typical of fynbos vegetation which had a within-group average similarity of 16.32, which is why they do not top the list of discriminating species between CIATs and fynbos (Appendix 3b).

Table 3.12. Average similarities between arthropod samples at CIATs vegetation type. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 19.79.

Species	Av.		Sim/SD	Contrib%	Cum.%
	Abund	Sim			
<i>Microhodotermes viator</i>	11.88	3.74	0.41	18.93	18.93
<i>Centrobolus</i> sp.2	5.66	2.73	0.49	13.82	32.75
<i>Camponotus</i> sp.2	5.11	2.72	0.62	13.75	46.50
<i>Zophosis boei</i>	3.90	1.68	0.53	8.47	54.97
<i>Linepithema humile</i>	3.18	1.30	0.45	6.58	61.55
<i>Porcello</i> sp.	2.92	1.07	0.31	5.42	66.97
<i>Tetramorium capense</i>	2.76	1.07	0.38	5.39	72.35
<i>Crematogaster peringueyi</i>	1.98	0.68	0.31	3.46	75.81
<i>Gryllus bimaculatus</i>	1.03	0.64	0.53	3.22	79.03
<i>Daspletis</i> sp.	1.42	0.53	0.37	2.67	81.69
<i>Diores youngai</i>	1.14	0.44	0.43	2.22	83.91
Hopliini sp.	0.90	0.21	0.21	1.09	85.00
<i>Achmaedera</i> sp.	0.57	0.14	0.18	0.71	85.71
<i>Evarcha</i> sp.1	0.49	0.12	0.23	0.63	86.34
<i>Paramelita nigroculus</i>	0.77	0.11	0.13	0.57	86.91
<i>Cophogryllus</i> sp.	0.45	0.11	0.21	0.53	87.44
<i>Galeodes</i> sp.2	0.33	0.10	0.22	0.52	87.96
<i>Phalangium</i> sp.1	1.01	0.10	0.15	0.51	88.48
<i>Cleptria rufipes</i>	0.34	0.10	0.22	0.51	88.99
Halictidae sp.1	0.35	0.10	0.20	0.50	89.49
<i>Centrobolus</i> sp.1	1.06	0.09	0.09	0.46	89.94
<i>Microlestia tabida</i>	0.71	0.09	0.15	0.46	90.40

Species typifying vineyard vegetation type

Table 3.13 shows the contributions of each species to the Bray-Curtis similarity within the vineyard. The Bray-Curtis similarity between all pairs of sites in the vineyard was 12.09, made up mainly of the contributions from ten species: *L.humile* (1.64, i.e. 13.61%), *Z. boei* (1.55, i.e. 12.82%), *Centrobolus* sp.2 (0.92, i.e. 7.63%), *Camponotus* sp.2 (0.82, i.e. 6.81%), *G. bimaculatus* (0.52, i.e. 4.26%), *Xestobium* sp. (0.51, i.e. 4.24%), Halictidae sp.1 (0.44, i.e. 3.62%), Curculionidae sp.1 (0.43, i.e. 3.6%), *T. capense* (0.42, i.e. 3.45%), *Porcello* sp. (0.42, i.e. 3.44%), with a cumulative contribution of over 63% of the total similarity of 12.09. However, the list only includes

the top species before 90% cumulative percentage was reached. These arthropod species can be identified as typical of vineyards.

Table 3.13. Average similarities between arthropod samples at vineyard vegetation type. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 12.09.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Linepithema humile</i>	3.71	1.64	0.35	13.61	13.61
<i>Zophosis boei</i>	2.90	1.55	0.35	12.82	26.42
<i>Centrobolus</i> sp.2	3.31	0.92	0.22	7.63	34.05
<i>Camponotus</i> sp.2	2.01	0.82	0.27	6.81	40.85
<i>Gryllus bimaculatus</i>	0.73	0.52	0.37	4.26	45.12
<i>Xestobium</i> sp.	1.88	0.51	0.17	4.24	49.36
Halictidae sp.1	0.94	0.44	0.29	3.62	52.98
Curculionidae sp.1	0.57	0.43	0.28	3.60	56.58
<i>Tetramorium capense</i>	1.26	0.42	0.24	3.45	60.03
<i>Porcello</i> sp.	1.70	0.42	0.19	3.44	63.47
<i>Diores youngai</i>	0.95	0.41	0.32	3.36	66.83
<i>Galeodes</i> sp.2	0.47	0.25	0.28	2.06	68.88
<i>Cheilomenes lunata</i>	0.55	0.24	0.18	1.98	70.87
Cleridae sp.1	0.82	0.20	0.14	1.69	72.55
<i>Formicomus coeruleus</i>	0.48	0.18	0.22	1.48	74.04
<i>Trigonopus</i> sp.1	0.53	0.17	0.22	1.42	75.46
Noctuidae sp.2	0.39	0.17	0.17	1.41	76.86
<i>Cynthia cardui</i>	0.41	0.16	0.19	1.35	78.22
<i>Oxyopes</i> sp.	0.51	0.14	0.17	1.15	79.37
<i>Trabea purcelli</i>	0.36	0.12	0.19	1.00	80.37
<i>Evarcha</i> sp.1	0.57	0.12	0.14	0.97	81.34
<i>Forticula senegalensis</i>	0.48	0.10	0.18	0.86	82.20
<i>Poophilus</i> sp.1	0.31	0.10	0.13	0.83	83.03
<i>Caminara</i> sp.1	0.29	0.10	0.17	0.80	83.84
Sphecidae sp.2	0.33	0.09	0.15	0.77	84.61
<i>Galeodes</i> sp.1	0.32	0.09	0.16	0.73	85.34
<i>Pterotricha varia</i>	0.31	0.09	0.16	0.73	86.07
<i>Centrobolus</i> sp.1	0.67	0.09	0.11	0.72	86.78
<i>Helicoverpa armigera</i>	0.25	0.07	0.11	0.56	87.35
Lycosidae sp.1	0.20	0.07	0.12	0.56	87.91
<i>Apis mellifera</i>	0.22	0.06	0.14	0.53	88.44
<i>Proevippa</i> sp.1	0.29	0.06	0.13	0.50	88.94
Paederinae sp.	0.25	0.06	0.12	0.49	89.43
<i>Lithobius</i> sp.2	0.39	0.06	0.09	0.47	89.90
<i>Microhodotermes viator</i>	0.94	0.06	0.05	0.47	90.36

Determining discriminating species

The average of the Bray-Curtis dissimilarities between all pairs of sites (i.e. CIATs and fynbos vegetation type) was 82.98, which was made up of 7.45 from *M. viator*, 5.18 from *Camponotus* sp.2, 5.13 from *Centrobolus* sp.2, 3.91 from *Porcello* sp., 3.23 from *L. humile* etc, indicated in the third column of the Appendix 3b. The *M. viator* contribution was 8.98% of the total 82.98, *Camponotus* sp.2 and *Centrobolus* sp.2 contributed 6.2 each to this total, etc. (column 5) and these percentages were cumulated in column 6, until the cut-off of >90% was reached. Column 4 is the ratio of the average contribution (column 3) divided by the standard deviation (SD) of all contributions making up this average. However, a good discriminating species is the one which contributes consistently to that distinction, i.e. with a low SD and thus higher ratio (Clarke and Gorley, 2006). Columns 1 and 2 provide the average abundance of species in CIATs and fynbos vegetation types respectively. 4th-root transformation was used and therefore, the means in columns 1 and 2 were calculated on these transformed abundances.

Species which are likely to be good discriminators of CIATs and fynbos are *M. viator*, *Camponotus* sp.2, *Centrobolus* sp.2, *L. humile*, *Z. boei*, *T. capense*, *Daspletis* sp., and *D. youngai* since they have both high average dissimilarity and the ratio (Diss/SD). For fynbos and vineyards, they are *Camponotus* sp.2, *L. humile*, *Z. boei* and *G. bimaculatus*. For IATs and vineyards, they are *Centrobolus* sp.2, *Porcello* sp., *L. humile*, *Camponotus* sp.2, and *G. bimaculatus*. For CIATs and vineyards, they are *Centrobolus* sp.2, *Camponotus* sp.2, *Camponotus* sp.2, *L. humile*, and *G. bimaculatus*; and for CIATs and IATs, they are *M. viator*, *Centrobolus* sp.2, *Porcello* sp., *Camponotus* sp.2, *L. humile*, and *Z. boei*. Detailed lists of discriminating species between different sampled vegetation types are given in Appendices 3b–3f.

DISCUSSION

Species richness in the different vegetation types

As none of the species accumulation curves in the various individual vegetation types reached an asymptote, species estimates were considered underestimates. Nevertheless, the overall species accumulation curve did reach an asymptote, indicating that the sample size (>25 000 individuals) was enough to estimate surface-active arthropod species richness using both number of samples or individuals. All

species estimators indicated highest species richness in fynbos, with CIATs and vineyards, in that order, having the next highest estimates. Lowest species richness was in the IATs, indicating that conversion of fynbos for agriculture has less impact on arthropod species richness than does that of invasive alien trees. Furthermore, vineyards do not automatically result in very low arthropod diversity as did IATs. As CIATs species richness was comparable to that of fynbos, the clearing of alien trees increased species richness, an encouraging sign for restoration.

Arthropods restricted to a particular vegetation type(s)

More arthropod species were sampled from fynbos than from IATs. In short, IATs had the highest number of absentees compared to fynbos, CIATs and vineyards. However, some species such as *Ozyptila* sp., *Spintharina* sp., and *Dorylus helvolus* were only sampled from IATs. This clearly indicates that although the majority of arthropod cannot tolerate IATs conditions, some can only survive under such conditions, at least in the sampled area. Presumably in the wild, they are restricted to heavily shaded, possibly highly local habitats. In contrast, certain species such as *Chrysomya chloropyga* and *Ciphalilus* sp. preferred other vegetation types except IATs.

Vegetation comparison and classification

CIATs sites were very similar in their arthropod species richness to those of natural fynbos (Table 3.3), supporting 90% and 85% of the total sampled arthropod species respectively, while vineyard and IATs supported 80% and 77% of the total species respectively. There was a considerable overlap of species across fynbos, IATs, CIATs and vineyards, possibly partly due to the vegetation types being adjacent to one another. All vegetation types had fewer unique arthropod species, with only five species unique to natural vegetation, four to vineyards, and three to each of IATs and CIATs. Species richness was greatest in the indigenous vegetation (fynbos) compared to IATs, supporting earlier findings in the CFR (Ratsirarson et al., 2002). Fynbos had very high species abundance, the highest recorded in this study. Although surface-active arthropod species richness was significantly higher in vineyards than in IATs, species abundance was similar within vineyard and IATs. IATs and vineyards did not cluster together in terms of species richness (Figure 3.4). However, this was not the case for species abundance (Figure 3.6). Surprisingly, even sites in very close proximity (e.g. Vergelegen and Helderberg) did not cluster together (Figure 3.5), showing considerable localization of arthropod species richness.

Impact of invasive alien tree removal

Arthropod species richness was much higher in CIATs than in IATs. Moreover, species diversity in CIATs was more similar to those in fynbos and significantly higher than in IATs. Moreover, both species richness estimates for CIATs were significantly higher than that of disturbed invasive alien tree vegetation. Effectiveness of clearing varies with density of the original plant infestation, species type, and time that the site has been invaded (Holmes and Richardson, 1999). This apparent ability of certain invertebrate species (e.g. *Cantharis* sp., *Leptinotarsa* sp., *Chrysomya chloropyga*, *Ectrichodia crux*, *Laurhervasia setacea* and *Comptonotus maculatus*) to recover when IATs are removed confirms other findings for aquatic fauna recovery following IAT removal (Magoba and Samways, 2010)

Millipedes and invasive alien trees

Centrobolus millipedes were abundant in IATs (especially plantation forestry) and CIATs, and much more so than in fynbos and vineyards. These millipedes were dominant throughout the year in the invaded areas, living in soil, leaf litter, and under logs, where they benefited from the decaying vegetation, both in terms of habitat structure and nutrients (Smit and Van Aarde, 2001).

Arthropod species and environmental variable correlations

Species composition of surface-active arthropods was strongly influenced by vegetation type (natural fynbos, IATs, CIATs or vineyard), and soil characteristics such as leaf litter depth. Natural fynbos, IATs, CIATs had similar % soil moisture, which appeared to drive these results. In contrast, vineyards had relatively lower % soil moisture than fynbos, or CIATs, with resultant fewer arthropods but greater than in IATs. However, soil moisture was not statistically correlated with arthropod species richness.

Different vegetation types had different leaf litter depths, with IATs having the deepest, and vineyards the lowest. CIATs had significantly lower leaf litter compared to either IATs or natural vegetation, possibly due to the high abundance of decomposers rapidly reducing the leaf litter once it was cleared.

Arthropod variation over sampling period

Surface-active arthropod species richness and abundance in fynbos and vineyards were highest during sampling period 1 (August-October) and low during sampling period 2

(November-January) and 3 (May-July) respectively. Although this was the same pattern for IATs and CIATs, species richness declined sharply during sampling period 2 and increased during sampling period 3. Sampling period 1, the time of peak flowering, was favourable time for most arthropods, because of high resource availability, in comparison with other sampling periods. Generally, there was a steady decline in the number of species from sampling period 1 to 2, although there was a slight decline from sampling period 2 to 3. During sampling period 2, species richness was very similar in fynbos, CIATs and vineyards. However, IATs had significantly lower species richness during sampling period 2, although becoming similar to vineyards in sampling period 3. The increase in species richness in IATs during the wet and cold period (sampling 3) compared to dry and hot period (sampling 2), indicates that there are several species which are surface-active only during the cooler, moisture period. IATs and vineyards were more similar and had significantly lower species richness during cold months (i.e. May-July). Generally, fynbos and CIATs had higher species richness while IATs supported the lowest species richness throughout the study period.

Effect of commercial forestry plantation on biodiversity

Here, soil leaf litter and hence habitat structure under forestry-type IATs were altered, as found elsewhere (Armstrong and van Hensbergen, 1996). Such changes may influence arthropod diversity in various ways. Generally, there was lower arthropod species richness in plantation forestry than in fynbos in the same area (e.g. Jonkershoek and Hottentots Holland Nature Reserves). This may have been due to reduced plant species richness and increased shade canopy, characteristic of invaded areas (Cowling et al., 1976; Richardson and van Wilgen, 1986; Richardson et al., 1989). Because IAT types used in plantation forestry in this area are often sited in relatively open vegetation, it is more likely that sun-loving species are eliminated from the plantations owing to lack of sufficient sunlight under the closed canopy. Nevertheless, some pine plantations can be nurseries for shade-loving understorey species and enhance their local expansion (Geldenhuys et al., 1986). Arthropod species that are likely to decrease most are those that have specific sunny habitat requirements.

Arthropod assemblage composition

There was clear distinction between different vegetation types based on their arthropod assemblage composition. Cluster and MDS analyses resulted in similar grouping of vegetation types. Fynbos and vineyard had comparable arthropod composition, whereas

CIATs had different arthropod composition compared to other vegetation types. Moreover, ANOSIM test showed that the arthropod assemblage composition between vegetation types was not similar, such that assemblage groups could be distinguished based on arthropod assemblage composition. Although fynbos (natural) and CIATs vegetation types grouped together in terms of arthropod species richness, they were very different in terms of assemblage composition. This indicates that different vegetation types, supporting equal number of species, can differ substantially in species assemblage composition. Vineyards supported fewer species, most of which were also recorded from fynbos vegetation, and thus more comparable in terms of species assemblage composition.

When rare species were excluded from the analyses (i.e. in PCA), to retain comparable species between vegetation types, fynbos and CIATs vegetation types were no longer comparable in terms of assemblage composition. PCA-ordination indicates that IATs differed in arthropod assemblage composition more than any other vegetation type. In fact, different vegetation types had different arthropod assemblage composition irrespective of the number of species they supported. Moreover, fynbos and CIATs supported a different arthropod assemblage, despite having comparable species richness. The open vegetation types (vineyard and CIATs) were differed little in terms of arthropod assemblage composition, and it is likely that both supported species that preferred open areas rather than dense and shady fynbos vegetation.

The various diversity indices used here (Margalef's index, Pielou's evenness index, Shannon diversity index, and Simpson index), gave similar results. Generally, they indicated higher diversity for fynbos vegetation and lowest for IATs.

Species typifying vegetation types

Several arthropod species were identified as typical of a particular vegetation type (Clarke and Gorley, 2006). *Camponotus* species and *Linepithema humile* were typical fynbos vegetation species. A *Centrobolus* species was found to be a more typical IATs vegetation type species. *M. viator* was distinctly a typical CIATs vegetation species, while *L. humile* and *Z. boei* were closely associated with vineyards.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

Different vegetation types performed differently in supporting arthropods. The majority of arthropod species preferred fynbos than any other vegetation type. It is therefore crucial to maintain as many natural fynbos patches as possible adjacent to any transformed areas, which would then serve as refuge areas. There was a significant correlation between habitat characteristics such as leaf litter depth and species richness. Surprisingly, soil moisture content was not correlated with species richness, despite varying significantly between different vegetation types. The observed correlation between arthropods and leaf litter was an indication of the influence of habitat characteristics on species diversity. Therefore, care should be taken during invasive alien tree clearing to minimize erosion and washing away of dead organic material during rainfall, which in turn, would affect surface-active arthropod species diversity.

IATs significantly reduced species richness and changed assemblage composition in comparison with fynbos, CIATs, and vineyards. Thus, IATs are more important in threatening arthropod diversity than vineyards. This lends support to the national IAT clearing programme, with arthropods being major beneficiaries. This was further emphasized by CIATs species richness being comparable to fynbos, with abundance even higher. The results here indicate that this is not a one-way process, with recovery being achieved on the one hand by removal of IATs, or, as shown by Gaigher (2008), by converting traditional vineyards to organic ones (Gaigher, 2008).

Arthropod species diversity was strongly influenced by sampling period, with most species being active during August-October, the peak flowering time. In turn, very few arthropods were active during May-July sampling period. Generally, IATs had lower species richness than vineyards. However, IATs and vineyards had similar species richness during May-July months. Vineyard species richness differs significantly from one sampling period to the other. These results suggest that when assessing the impact of alien vegetation or the success of its removal, the counts should take place during flowering time.

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APPENDICES

Appendix 3a Arthropod species and their sampled abundance from different vegetation types: fynbos, invasive alien trees (IATs), cleared of invasive alien trees (CIATs) and vineyard sites. Numbers refers to number of surface-active individuals captured.

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Araneae						
Amaurobiidae	<i>Chresiona</i> sp.1	12	4	1	1	18
Gallieniellidae	<i>Drassodella</i> sp.1	30	24	19	15	88
Gnaphosidae	<i>Camillina</i> sp.1	28	32	35	26	121
Gnaphosidae	<i>Camillina</i> sp.2		3		2	5
Gnaphosidae	<i>Pterotricha varia</i>	24	15	19	39	97
Gnaphosidae	<i>Zelotes lightfooti</i>	31	19	35	18	103
Lycosidae	<i>Geolycosa</i> sp.1	12		5		17
Lycosidae	<i>Hogna</i> sp.1	8	5	2	14	29
Lycosidae	<i>Proevippa schreineri</i>	41	22	32	38	133
Lycosidae	<i>Proevippa</i> sp.1	45	12	33	37	127
Lycosidae	<i>Proevippa</i> sp.2	11	2	11	6	30
Lycosidae	Sp.1	29	5	5	25	64
Lycosidae	<i>Trabea purcelli</i>	34	16	22	45	117
Nemesiidae	<i>Pionothele straminae</i>	3		4	2	9
Oxyopidae	<i>Oxyopes</i> sp.	78	35	41	64	218
Palpimanidae	<i>Diaphorocellus</i> sp.1	41	12	20	7	80
Philodromidae	<i>Tibellus</i> sp.	3	5	3	9	20
Pisauridae	<i>Rothus purpurissatus</i>	31	23	21	14	89
Salticidae	<i>Aelurillus</i> sp.1	38	15	41	10	104
Salticidae	<i>Evarcha</i> sp.1	53	9	56	72	190
Salticidae	<i>Habrocestrum</i> sp.1	29	4	14	9	56
Salticidae	<i>Langona</i> sp.1	47	18	32	25	122
Salticidae	<i>undertermined genus</i>	6		4	1	11
Theridiidae	<i>Steatoda capensis</i>	13	21	20	8	62
Theridiidae	<i>Steatoda</i> sp.2	40	19	27	12	98
Theridiidae	<i>Theridion</i> sp.	11			63	74
Thomisidae	<i>Ozyptila</i> sp.		3			3
Thomisidae	<i>Synema imitator</i>	115	3	16	34	168
Trochanteriidae	<i>Platyoides</i> sp.	3				3
Zodariidae	<i>Diores capensis</i>	8	9	7	15	39
Zodariidae	<i>Diores youngai</i>	186	85	131	120	522
Blattodea						
Blaberidae	<i>Aptera fusca</i>	27	3	3	2	35
Blattidae	<i>Temnopteryx phalerata</i>	43	15	29	11	98
Coleoptera						
Anobiidae	<i>Xestobium</i> sp.	42	109	45	237	433
Anthricidae	<i>Endomia elongates</i>	62	57	64	22	205
Anthricidae	<i>Formicomus coeruleus</i>	11	4	9	61	85
Buprestidae	<i>Achmaedera</i> sp.	85	21	66	30	202
Cantharidae	<i>Cantharis</i> sp.1	4		15	6	25
Carabidae	<i>Acanthoscelis ruficornis</i>	5	3	11	1	20
Carabidae	<i>Anthia decemquttata</i>	16	3			19
Carabidae	<i>Caminara</i> sp.1	15	6	16	36	73
Carabidae	<i>Graphipterus trilineatus</i>	15			1	16
Carabidae	<i>Microlestia tabida</i>	125	3	82	12	222
Carabidae	<i>Passalidius fortipes</i>	1				1

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Carabidae	Sp.1	18	36	4	12	70
Chrysomelidae	<i>Leptinotarsa</i> sp.1	1		2	2	5
Cicindelinae	<i>Platydelia quadriguttata</i>	2				2
Cleridae	Sp.1	68	28	14	103	213
Coccinellidae	<i>Cheilomenes lunata</i>	10	9	30	69	118
Coccinellidae	<i>Harmonia axyridis</i>		5	1	29	35
Curculionidae	Sp.1	26	13	16	72	127
Curculionidae	Sp.2	5	1	4	1	11
Curculionidae	Sp.3	3	1	6	20	30
Curculionidae	Sp.4	4	9	13	4	30
Dermestidae	<i>Dermestes maculatus</i>	10	7	9		26
Dermestidae	<i>Dermestes</i> sp.	4	2	6	42	54
Elateridae	<i>Cardiotarsus acuminatus</i>	4	3	4		11
Lampyridae	<i>Lampyris disticollis</i>		2	1		3
Lampyridae	<i>Luciola</i> sp.	1		3		4
Lycidae	Sp.1	1				1
Meloidae	<i>Decapotoma lunata</i>	4	1		2	7
Melyridae	<i>Apalochrus</i> sp.1	8	1	17		26
Mordellidae	Sp.1			8	1	9
Mordellidae	Sp.2	22	7	16	15	60
Nitidulidae	Sp.1	32	18	27	1	78
Nitidulidae	Sp.2	11	22	21	52	106
Scarabaeidae	<i>Schizonycha</i> sp.	2	2	2	1	7
Scarabaeidae	Sp.2	4	4	4	7	19
Scarabaeidae	<i>Aphodius</i> sp.	9		5	17	31
Scarabaeidae	Hopliini sp.	116	83	104	25	328
Scarabaeidae	<i>Scarabaeus rigosus</i>	38	11	18	1	68
Scarabidae	Sp.1	5	1	1	8	15
Silvanidae	<i>Oryzaephilus</i> sp.	1		2	8	11
Staphylinidae	<i>Paederinae</i> sp.	46	23	35	31	135
Tenebrionidae	<i>Psammodes striatus</i>				1	1
Tenebrionidae	Sp.1	78	17	8	8	111
Tenebrionidae	Sp.2	69	5	10	13	97
Tenebrionidae	Sp.3	1		3	1	5
Tenebrionidae	<i>Tenebrio</i> cf. <i>molitor</i>	15	7	3	7	32
Tenebrionidae	<i>Tenebrio</i> sp.1	44	45	23	38	150
Tenebrionidae	<i>Trigonopus</i> sp.1	34	16	21	67	138
Tenebrionidae	<i>Zophosis boei</i>	317	25	449	365	1156
Tenebrionidae	<i>Zophosis</i> sp.1	7		1		8
Collembola						
Sminthuridae	<i>Sminthurus viridis</i>	23	39	63	58	183
Crustacea						
Amphipoda	<i>Paramelita nigroculus</i>	119	21	89	6	235
Dermaptera						
Forficulidae	<i>Forficula senegalensis</i>	39	17	32	60	148
Diptera						
Anthomyiidae	<i>Anthomyia</i> sp.	33	37	26	11	107
Asilidae	<i>Daspletis</i> sp.	155	41	163	35	394
Bombyliidae	<i>Exoprosopa</i> sp.	2	2			4
Calliphoridae	<i>Chrysomya chloropyga</i>	11		13	2	26
Calliphoridae	Sp.1	13	9		5	27
Calliphoridae	Sp.2	35	10	22	21	88
Cecidomyiidae	<i>Cecidomyia</i> sp.1	36	220	13		269
Heleomyzidae	<i>Helomyza picta</i>	2	10	4		16
Muscidae	Sp.1	13	8	24	6	51
Pipunculidae	Sp.1	34	127	60	54	275
Sarcophagidae	Sp.1	3	4	1	7	15
Sciaridae	Sp.1	15	148	63	40	266

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Syrphidae	Sp.1	1	5	12	1	19
Tachinidae	<i>Dejeania</i> sp.	30	16	18	2	66
Tachinidae	<i>Gonia</i> sp.	3	1	2	5	11
Tachinidae	Sp.1	14	1	5	8	28
Tachinidae	Sp.2	8	7	3	4	22
Tachinidae	Sp.3	8	6	16	8	38
Tachinidae	Sp.4	3	2	4	7	16
Therividae	<i>Therevid</i> sp.	5	3		2	10
Tipulidae	<i>Tipula jocose</i>	2	4	2	2	10
Hemiptera						
Anthocoridae	<i>Anthocoris</i> sp.1	8	6	18	21	53
Cercopidae	<i>Poophilus</i> sp.1	30	14	26	39	109
Cicadellidae	<i>Ciphalilus</i> sp.	2		3	1	6
Cicindelidae	<i>Cicindela quadriguttata</i>	3	1	17		21
Cydnidae	Sp.1	7	49		2	58
Miridae	<i>Deraeocoris</i> sp.	2	1	2	1	6
Pentatomidae	<i>Agonoscelis</i> sp.1	2		3	5	10
Pentatomidae	<i>Agonoscelis</i> sp.2	1				1
Pentatomidae	<i>Antestia lymphata</i>	3	1	5	5	14
Pentatomidae	<i>Bagrada hilaris</i>	2			23	25
Pentatomidae	<i>Coenomorpha</i> sp.	2		1		3
Pyrrhocoridae	<i>Scantius forsteri</i>	15	11	12	4	42
Reduviidae	<i>Acanthaspis sanguinosa</i>	1				1
Reduviidae	<i>Cleptria rufipes</i>	49	17	39	10	115
Reduviidae	<i>Ectrichodia crux</i>	2		3	1	6
Reduviidae	<i>Holoptilus</i> sp.1			1		1
Reduviidae	? <i>Oncocephalinae</i> sp.	1	27		2	30
Piesmatidae	Sp.1				2	2
Hymenoptera						
Anthophoridae	<i>Thyreus delumbatus</i>	2	2	4	1	9
Anthophoridae	<i>Xylocopa</i> sp.	1	1			2
Apidae	<i>Apis mellifera</i>	12	2	13	28	55
Apidae	<i>Meliponula</i> sp.1	5		4	26	35
Braconidae	<i>Charops</i> sp.1		2	3		5
Chalcididae	<i>Brachymeria kassalensis</i>	1	2	1	1	5
Chrysididae	<i>Spintharina</i> sp.		3			3
Formicidae	<i>Camponotus maculatus</i>	96		14	5	115
Formicidae	<i>Camponotus</i> sp.1	28	4	17	2	51
Formicidae	<i>Camponotus</i> sp.2	758	108	588	253	1707
Formicidae	<i>Camponotus</i> sp.3	21	30	9		60
Formicidae	<i>Crematogaster peringueyi</i>	300	132	228	18	678
Formicidae	<i>Dorylus helvolus</i>		1			1
Formicidae	<i>Linepithema humile</i>	359	99	366	467	1291
Formicidae	<i>Messor capensis</i>	62	27	34	14	137
Formicidae	Sp.1	16	15	8	19	58
Formicidae	Sp.2	4				4
Formicidae	Sp.3	135	12	22	18	187
Formicidae	<i>Tetramorium capense</i>	241	157	317	159	874
Halictidae	<i>Nomia amabilis</i>	54	15	18		87
Halictidae	Sp.1	36	21	40	118	215
Halictidae	Sp.2	5			2	7
Halictidae	Sp.3		2	2	2	6
Masaridae	<i>Ceramius</i> sp.1			1		1
Melittidae	Sp.1	29	1	12		42
Melittidae	Sp.2	10	4	7	12	33
Mutillidae	Sp.1	7	1	8	17	33

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Mutillidae	Sp.2	9	1	4	4	18
Pompilidae	Sp.1	43	15	29	14	101
Pompilidae	Sp.2	8	3	14	5	30
Scoliidae	<i>Campsomeriella</i> sp.1	14	3	11		28
Sphecidae	Sp.1	6	2	4	6	18
Sphecidae	Sp.2	56	39	44	42	181
Sphecidae	Sp.3	6		3	6	15
Sphecidae	Sp.4			1		1
Sphecidae	<i>Sphex bonaspei</i>	41	4	36	3	84
Sphecidae	<i>Sphex tydei</i> var <i>capensis</i>	12		5	1	18
Tiphiidae	Sp.1	4	7	5	4	20
Isopoda						
Porcellionidae	<i>Porcello</i> sp.	309	569	336	214	1428
Isoptera						
Hodotermitidae	<i>Microhodotermes viator</i>	776	298	1366	118	2558
Lepidoptera						
Arctiidae	<i>Rhodogastria amasis</i>	6	19	2	4	31
Geometridae	Sp.1	6		4	4	14
Lasiocampidae	<i>Eutricha capensis</i>			4	1	5
Lycaenidae	<i>Aloeides</i> sp.	1	2	3	12	18
Noctuidae	<i>Helicoverpa armigera</i>	18	51	6	32	107
Noctuidae	Sp.1		2	64	17	83
Noctuidae	Sp.2	6	12	5	49	72
Nymphalidae	<i>Cynthia cardui</i>	21	32	13	52	118
Psychidae	Sp.1	2		1		3
Pyalidae	Sp.1	4	8	3	4	19
Sphingidae	<i>Coelonia fulvnotata</i>				4	4
Tineidae	Sp.1	9	3		1	13
Tortricidae	Sp.1		3			3
Lithobiida						
Lithobiidae	<i>Lithobius</i> sp.1	26	4	20	16	66
Lithobiidae	<i>Lithobius</i> sp.2	9	10	12	49	80
Mantodea						
Hymenopodidae	<i>Harpagomantis tricolor</i>	5		1		6
Mantidae	<i>Polyspilota aeruginosa</i>				1	1
Mantidae	Sp.1	3	4	11	5	23
Mantidae	Sp.2	3		13	7	23
Mantidae	Sp.3	14	7	7		28
Mecoptera						
Bittacidae	<i>Anomalobittacus gracilipes</i>	17	44	9	4	74
Neuroptera						
Nemopteridae	<i>Laurhervasia setacea</i>	2		2	1	5
Lithobiomorpha						
Peripatopsidae	<i>Peripatopsis</i> sp.	7	6	3		16
Opiliones						
Phalangidae	<i>Phalangium</i> sp.1	31	14	116	6	167
Orthoptera						
Gryllidae	<i>Cophogryllus</i> sp.	63	24	52	22	161
Gryllidae	<i>Gryllus bimaculatus</i>	196	93	119	92	500
Scolopendromorpha						
Scolopendromorphae	<i>Scolopendra cingulata</i>	13	2	3	24	42
Scorpiones						
Buthidae	<i>Uroplectes lineatus</i>	21	1	1	1	24
Scorpionidae	<i>Opisthophthalmus macer</i>	12	22	22		56
Solifugae						
Eremobatidae	<i>Eremobates</i> sp.1	2		4		6

Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Galeodidae	<i>Galeodes</i> sp.1	47	16	26	40	129
Galeodidae	<i>Galeodes</i> sp.2	45	4	38	59	146
Spirobolida						
Pachybolidae	<i>Centrobolus</i> sp.1	67	113	122	85	387
Pachybolidae	<i>Centrobolus</i> sp.2	406	734	651	417	2208
Total sampled individuals		7746	4725	7563	5221	25255
Total sampled species		179	153	169	159	198
%Total sampled species		90	77	85	80	

Appendix 3b Discriminating species between vegetation cleared of invasive alien trees (CIATs) and fynbos vegetation types, ordered in decreasing contribution.

Average dissimilarity = 82.98.

Species	CIATs	Fynbos		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Microhodotermes viator</i>	11.88	5.35	7.45	0.85	8.98	8.98
<i>Camponotus</i> sp.2	5.11	5.23	5.18	1.06	6.24	15.22
<i>Centrobolus</i> sp.2	5.66	2.80	5.13	0.82	6.19	21.40
<i>Porcello</i> sp.	2.92	2.13	3.91	0.66	4.71	26.11
<i>Linepithema humile</i>	3.18	2.48	3.23	0.88	3.89	30.00
<i>Zophosis boei</i>	3.90	2.19	2.89	0.86	3.48	33.48
<i>Tetramorium capense</i>	2.76	1.66	2.83	0.82	3.41	36.90
<i>Crematogaster peringueyi</i>	1.98	2.07	2.62	0.69	3.15	40.05
<i>Gryllus bimaculatus</i>	1.03	1.35	1.94	0.72	2.34	42.39
<i>Daspletis</i> sp.	1.42	1.07	1.56	0.80	1.88	44.27
<i>Diores youngai</i>	1.14	1.28	1.51	0.79	1.82	46.09
Hopliini sp.	0.90	0.80	1.48	0.45	1.79	47.88
<i>Microlestia tabida</i>	0.71	0.86	1.09	0.54	1.32	49.19
<i>Centrobolus</i> sp.1	1.06	0.46	1.08	0.43	1.30	50.49
<i>Synema imitator</i>	0.14	0.79	1.06	0.38	1.27	51.76
Formicidae sp.3	0.19	0.93	1.04	0.31	1.25	53.01
<i>Achmaedera</i> sp.	0.57	0.59	1.04	0.48	1.25	54.26
<i>Paramelita nigroculus</i>	0.77	0.82	1.03	0.43	1.24	55.50
<i>Endomia elongates</i>	0.56	0.43	0.99	0.38	1.19	56.69
<i>Cophogryllus</i> sp.	0.45	0.43	0.75	0.64	0.90	57.60
<i>Oxyopes</i> sp.	0.36	0.54	0.74	0.61	0.89	58.48
Sphecidae sp.2	0.38	0.39	0.70	0.40	0.85	59.33
<i>Phalangium</i> sp.1	1.01	0.21	0.69	0.45	0.83	60.16
<i>Langona</i> sp.1	0.28	0.32	0.63	0.52	0.75	60.91
<i>Sminthurus viridis</i>	0.55	0.16	0.63	0.26	0.75	61.66
<i>Camponotus maculatus</i>	0.12	0.66	0.62	0.19	0.75	62.41
<i>Cleptria rufipes</i>	0.34	0.34	0.62	0.62	0.74	63.16
<i>Galeodes</i> sp.2	0.33	0.31	0.61	0.61	0.74	63.90
<i>Messor capensis</i>	0.30	0.43	0.60	0.32	0.72	64.62
Pipunculidae sp.1	0.52	0.23	0.60	0.32	0.72	65.34
<i>Evarcha</i> sp.1	0.49	0.37	0.59	0.63	0.71	66.05
Paederinae sp.	0.30	0.32	0.59	0.49	0.71	66.76
<i>Proevippa schreineri</i>	0.28	0.28	0.57	0.56	0.69	67.45
<i>Proevippa</i> sp.1	0.29	0.31	0.57	0.52	0.69	68.13
<i>Sphex bonaspei</i>	0.31	0.28	0.55	0.32	0.66	68.79
<i>Galeodes</i> sp.1	0.23	0.32	0.54	0.51	0.65	69.45
<i>Aelurillus</i> sp.1	0.36	0.26	0.54	0.54	0.65	70.10
<i>Xestobium</i> sp.	0.39	0.29	0.54	0.42	0.65	70.75
<i>Steatoda</i> sp.2	0.23	0.28	0.53	0.48	0.63	71.38
<i>Forficula senegalensis</i>	0.28	0.27	0.52	0.49	0.63	72.01
Tenebrionidae sp.1	0.07	0.54	0.52	0.41	0.62	72.63
Nitidulidae sp. 1	0.23	0.22	0.51	0.29	0.61	73.24
<i>Anthomyia</i> sp.	0.23	0.23	0.50	0.50	0.60	73.85

Species	CIATs	Fynbos	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Halictidae sp.1</i>	0.35	0.25	0.48	0.59	0.58	74.43
<i>Pompilidae sp.1</i>	0.25	0.30	0.47	0.58	0.57	75.00
<i>Sciaridae sp.1</i>	0.55	0.10	0.46	0.22	0.55	75.55
<i>Poophilus sp.1</i>	0.23	0.21	0.45	0.56	0.54	76.09
<i>Camillina sp.1</i>	0.30	0.19	0.44	0.40	0.53	76.63
<i>Cleridae sp.1</i>	0.12	0.47	0.44	0.38	0.53	77.15
<i>Rothus purpurissatus</i>	0.18	0.21	0.43	0.39	0.52	77.68
<i>Scarabaeus rigosus</i>	0.16	0.26	0.43	0.36	0.52	78.20
<i>Temnopteryx phalerata</i>	0.25	0.30	0.42	0.58	0.50	78.70
<i>Diaphorocellus sp.1</i>	0.17	0.28	0.41	0.51	0.49	79.20
<i>Tenebrio sp.1</i>	0.20	0.30	0.40	0.50	0.49	79.68
<i>Zelotes lightfooti</i>	0.30	0.21	0.40	0.55	0.48	80.17
<i>Drassodella sp. 1</i>	0.17	0.21	0.39	0.35	0.47	80.63
<i>Cecidomyia sp. 1</i>	0.11	0.25	0.39	0.22	0.46	81.10
<i>Tenebrionidae sp.2</i>	0.09	0.48	0.38	0.29	0.46	81.56
<i>Trigonopus sp.1</i>	0.18	0.23	0.38	0.42	0.46	82.02
<i>Lithobius sp.1</i>	0.17	0.18	0.38	0.45	0.45	82.47
<i>Nomia amabilis</i>	0.16	0.37	0.36	0.26	0.44	82.91
<i>Trabea purcelli</i>	0.19	0.23	0.36	0.47	0.44	83.35
<i>Cheilomenes lunata</i>	0.26	0.07	0.36	0.26	0.43	83.78
<i>Lycosidae sp.1</i>	0.04	0.20	0.35	0.26	0.42	84.20
<i>Curculionidae sp.1</i>	0.14	0.18	0.34	0.38	0.41	84.60
<i>Calliphoridae sp.2</i>	0.19	0.24	0.33	0.44	0.40	85.00
<i>Habrocestrum sp.1</i>	0.12	0.20	0.31	0.44	0.37	85.37
<i>Pterotricha varia</i>	0.17	0.17	0.29	0.41	0.35	85.72
<i>Muscidae sp.</i>	0.21	0.09	0.29	0.21	0.35	86.07
<i>Melittidae sp.1</i>	0.10	0.20	0.28	0.33	0.34	86.41
<i>Camponotus sp.1</i>	0.15	0.19	0.26	0.28	0.32	86.73
<i>Lithobius sp.2</i>	0.10	0.06	0.26	0.24	0.32	87.05
<i>Aptera fusca</i>	0.03	0.19	0.25	0.29	0.30	87.35
<i>Cynthia cardui</i>	0.11	0.14	0.25	0.19	0.30	87.65
<i>Dejeania sp.</i>	0.16	0.21	0.24	0.39	0.30	87.94
<i>Steatoda capensis</i>	0.17	0.09	0.24	0.43	0.29	88.23
<i>Mordellidae sp. 2</i>	0.14	0.15	0.24	0.33	0.29	88.52
<i>Nitidulidae sp. 2</i>	0.18	0.08	0.23	0.28	0.27	88.79
<i>Pompilidae sp.2</i>	0.12	0.06	0.21	0.34	0.26	89.05
<i>Opisththalmus macer</i>	0.19	0.08	0.21	0.35	0.26	89.31
<i>Cantharis sp.1</i>	0.13	0.03	0.21	0.15	0.25	89.56
<i>Anthocoris sp.1</i>	0.16	0.06	0.21	0.28	0.25	89.81
<i>Camponotus sp.3</i>	0.08	0.14	0.21	0.22	0.25	90.06

Appendix 3c Discriminating species between fynbos vegetation and vineyard, ordered in decreasing contribution. Average dissimilarity = 88.65.

Species	Fynbos	Vineyard	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Camponotus</i> sp.2	5.23	2.01	5.09	1.00	5.74	5.74
<i>Centrobolus</i> sp.2	2.80	3.31	4.55	0.61	5.14	10.88
<i>Linepithema humile</i>	2.48	3.71	4.34	0.81	4.89	15.77
<i>Zophosis boei</i>	2.19	2.90	3.81	0.70	4.30	20.07
<i>Microhodotermes viator</i>	5.35	0.94	3.60	0.54	4.06	24.13
<i>Porcello</i> sp.	2.13	1.70	3.22	0.58	3.64	27.76
<i>Tetramorium capense</i>	1.66	1.26	2.38	0.69	2.68	30.44
<i>Xestobium</i> sp.	0.29	1.88	2.36	0.43	2.66	33.11
<i>Gryllus bimaculatus</i>	1.35	0.73	2.11	0.74	2.37	35.48
<i>Diores youngai</i>	1.28	0.95	1.76	0.66	1.98	37.46
<i>Crematogaster peringueyi</i>	2.07	0.14	1.65	0.49	1.86	39.32
Cleridae sp.1	0.47	0.82	1.35	0.42	1.52	40.85
<i>Synema imitator</i>	0.79	0.27	1.32	0.39	1.49	42.34
Halictidae sp.1	0.25	0.94	1.24	0.56	1.40	43.73
Curculionidae sp.1	0.18	0.57	1.22	0.58	1.37	45.11
<i>Daspletis</i> sp.	1.07	0.28	1.12	0.67	1.27	46.37
<i>Cheilomenes lunata</i>	0.07	0.55	1.11	0.40	1.25	47.62
Formicidae sp.3	0.93	0.14	1.05	0.30	1.19	48.81
<i>Oxyopes</i> sp.	0.54	0.51	1.02	0.56	1.15	49.97
Hopliini sp.	0.80	0.20	1.00	0.38	1.13	51.10
<i>Centrobolus</i> sp.1	0.46	0.67	0.90	0.48	1.01	52.11
<i>Galeodes</i> sp.2	0.31	0.47	0.86	0.66	0.97	53.08
<i>Evarcha</i> sp.1	0.37	0.57	0.85	0.44	0.96	54.04
<i>Microlestia tabida</i>	0.86	0.10	0.81	0.47	0.91	54.95
<i>Trigonopus</i> sp.1	0.23	0.53	0.80	0.43	0.91	55.85
<i>Achmaedera</i> sp.	0.59	0.24	0.80	0.47	0.90	56.75
Sphecidae sp.2	0.39	0.33	0.79	0.47	0.89	57.65
Noctuidae sp.2	0.04	0.39	0.76	0.39	0.86	58.50
<i>Lithobius</i> sp.2	0.06	0.39	0.75	0.24	0.84	59.35
Pipunculidae sp.1	0.23	0.43	0.75	0.30	0.84	60.19
<i>Cynthia cardui</i>	0.14	0.41	0.73	0.45	0.83	61.01
<i>Sminthurus viridis</i>	0.16	0.46	0.73	0.27	0.82	61.83
<i>Formicomus coeruleus</i>	0.08	0.48	0.71	0.46	0.80	62.63
<i>Poophilus</i> sp.1	0.21	0.31	0.69	0.48	0.78	63.41
Paederinae sp.	0.32	0.25	0.69	0.46	0.78	64.19
<i>Proevippa schreineri</i>	0.28	0.30	0.68	0.44	0.76	64.95
Sciaridae sp.1	0.10	0.32	0.67	0.23	0.76	65.71
<i>Cophogryllus</i> sp.	0.43	0.17	0.66	0.55	0.75	66.46
<i>Galeodes</i> sp.1	0.32	0.32	0.64	0.52	0.73	67.18
<i>Langona</i> sp.1	0.32	0.20	0.64	0.50	0.73	67.91
<i>Endomia elongatus</i>	0.43	0.17	0.64	0.36	0.72	68.63
<i>Messor capensis</i>	0.43	0.11	0.64	0.30	0.72	69.35
<i>Forficula senegalensis</i>	0.27	0.48	0.63	0.51	0.72	70.07
<i>Trabea purcelli</i>	0.23	0.36	0.62	0.46	0.70	70.77

Species	Fynbos	Vineyard		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
Nitidulidae sp.2	0.08	0.41	0.62	0.25	0.70	71.47
Lycosidae sp.1	0.20	0.20	0.62	0.40	0.70	72.17
<i>Camponotus maculatus</i>	0.66	0.04	0.61	0.18	0.69	72.86
<i>Proevippa</i> sp.1	0.31	0.29	0.58	0.51	0.65	73.51
<i>Tenebrio</i> sp.1	0.30	0.30	0.58	0.40	0.65	74.16
<i>Helicoverpa armigera</i>	0.12	0.25	0.55	0.36	0.62	74.78
<i>Caminara</i> sp.1	0.10	0.29	0.55	0.36	0.62	75.40
<i>Paramelita nigroculus</i>	0.82	0.05	0.52	0.27	0.59	75.99
Calliphoridae sp.2	0.24	0.17	0.51	0.43	0.57	76.56
<i>Cleptria rufipes</i>	0.34	0.08	0.49	0.50	0.55	77.11
<i>Pterotricha varia</i>	0.17	0.31	0.48	0.47	0.54	77.65
<i>Steatoda</i> sp.2	0.28	0.10	0.48	0.42	0.54	78.19
Tenebrionidae sp.1	0.54	0.06	0.47	0.39	0.53	78.72
<i>Zelotes lightfooti</i>	0.21	0.14	0.46	0.45	0.52	79.24
<i>Camillina</i> sp.1	0.19	0.21	0.45	0.45	0.51	79.75
<i>Lithobius</i> sp.1	0.18	0.13	0.44	0.43	0.50	80.25
Pompilidae sp.1	0.30	0.11	0.42	0.44	0.48	80.73
<i>Aelurillus</i> sp.1	0.26	0.08	0.40	0.46	0.45	81.18
<i>Theridion</i> sp.	0.08	0.50	0.39	0.24	0.44	81.62
<i>Apis mellifera</i>	0.08	0.22	0.39	0.40	0.44	82.05
Tenebrionidae sp.2	0.48	0.10	0.38	0.27	0.43	82.49
<i>Anthomyia</i> sp.	0.23	0.09	0.38	0.40	0.43	82.92
<i>Harmonia axyridis</i>	0.00	0.23	0.38	0.30	0.43	83.34
<i>Drassodella</i> sp.1	0.21	0.12	0.38	0.41	0.43	83.77
<i>Dermestes</i> sp.	0.03	0.33	0.37	0.22	0.42	84.19
<i>Rothus purpurissatus</i>	0.21	0.11	0.37	0.34	0.42	84.61
<i>Diaphorocellus</i> sp.1	0.28	0.06	0.36	0.45	0.40	85.02
<i>Temnopteryx phalerata</i>	0.30	0.09	0.35	0.48	0.39	85.41
Carabidae sp.1	0.12	0.10	0.33	0.23	0.37	85.78
<i>Scarabaeus rigosus</i>	0.26	0.01	0.33	0.28	0.37	86.15
Formicidae sp.1	0.11	0.15	0.32	0.28	0.36	86.51
<i>Aphodius</i> sp.	0.06	0.13	0.32	0.26	0.36	86.87
<i>Phalangium</i> sp.1	0.21	0.05	0.31	0.32	0.35	87.23
<i>Scolopendra cingulata</i>	0.09	0.19	0.30	0.30	0.34	87.57
<i>Sphex bonaspei</i>	0.28	0.02	0.30	0.19	0.33	87.90
<i>Nomia amabilis</i>	0.37	0.00	0.30	0.21	0.33	88.23
Nitidulidae sp.1	0.22	0.01	0.29	0.22	0.33	88.57
Mordellidae sp.2	0.15	0.12	0.29	0.25	0.33	88.90
<i>Hogna</i> sp.1	0.06	0.11	0.29	0.23	0.32	89.22
<i>Anthocoris</i> sp.1	0.06	0.17	0.27	0.23	0.31	89.53
<i>Cecidomyia</i> sp.1	0.25	0.00	0.27	0.20	0.31	89.83
<i>Meliponula</i> sp.1	0.03	0.21	0.26	0.30	0.29	90.13

Appendix 3d Discriminating species between invasive alien tree (IATs) vegetation and vineyard, ordered in decreasing contribution. Average dissimilarity = 90.82.

Species	IATs	Vineyard		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Centrobolus</i> sp.2	5.92	3.31	7.12	0.84	7.84	7.84
<i>Porcello</i> sp.	4.59	70	5.57	0.73	6.14	13.97
<i>Linepithema humile</i>	0.80	3.71	3.92	0.68	4.31	18.29
<i>Zophosis boei</i>	0.20	2.90	3.44	0.59	3.78	22.07
<i>Xestobium</i> sp.	0.88	1.88	3.03	0.52	3.33	25.40
<i>Camponotus</i> sp.2	0.87	2.01	2.92	0.67	3.21	28.62
<i>Tetramorium capense</i>	1.27	1.26	2.73	0.58	3.00	31.62
<i>Microhodotermes viator</i>	2.40	0.94	2.70	0.40	2.97	34.59
<i>Centrobolus</i> sp.1	0.91	0.67	2.06	0.48	2.27	36.86
<i>Gryllus bimaculatus</i>	0.75	0.73	1.98	0.67	2.18	39.05
Sciaridae sp.1	1.19	0.32	1.88	0.49	2.08	41.12
Pipunculidae sp.1	1.02	0.43	1.83	0.42	2.01	43.14
<i>Diores youngai</i>	0.69	0.95	1.82	0.60	2.00	45.14
<i>Crematogaster peringueyi</i>	1.06	0.14	1.52	0.46	1.67	46.81
<i>Cecidomyia</i> sp.1	1.77	0.00	1.43	0.28	1.57	48.39
Cleridae sp.1	0.23	0.82	1.40	0.37	1.54	49.92
Curculionidae sp.1	0.10	0.57	1.34	0.55	1.48	51.40
Halictidae sp.1	0.17	0.94	1.24	0.56	1.37	52.77
Hopliini sp.	0.67	0.20	1.16	0.29	1.28	54.05
<i>Cheilomenes lunata</i>	0.07	0.55	1.15	0.42	1.27	55.31
<i>Oxyopes</i> sp.	0.28	0.51	1.11	0.42	1.22	56.53
<i>Tenebrio</i> sp.1	0.36	0.30	1.05	0.32	1.16	57.69
<i>Cynthia cardui</i>	0.26	0.41	1.00	0.42	1.10	58.79
<i>Sminthurus viridis</i>	0.31	0.46	0.93	0.31	1.03	59.82
Sphecidae sp.2	0.31	0.33	0.86	0.47	0.94	60.76
Noctuidae sp.2	0.10	0.39	0.83	0.42	0.92	61.68
Nitidulidae sp.2	0.18	0.41	0.80	0.29	0.88	62.56
<i>Endomia elongatus</i>	0.46	0.17	0.79	0.29	0.87	63.43
<i>Helicoverpa armigera</i>	0.41	0.25	0.79	0.42	0.87	64.30
<i>Trigonopus</i> sp.1	0.13	0.53	0.78	0.41	0.85	65.15
<i>Lithobius</i> sp.2	0.08	0.39	0.77	0.24	0.85	66.00
<i>Evarcha</i> sp.1	0.07	0.57	0.71	0.36	0.78	66.78
<i>Galeodes</i> sp.2	0.03	0.47	0.67	0.54	0.74	67.52
<i>Formicomus coeruleus</i>	0.03	0.48	0.67	0.43	0.73	68.25
<i>Poophilus</i> sp.1	0.11	0.31	0.66	0.44	0.73	68.99
<i>Pterotricha varia</i>	0.12	0.31	0.66	0.44	0.73	69.72
<i>Daspletis</i> sp.	0.33	0.28	0.65	0.41	0.71	70.43
<i>Forficula senegalensis</i>	0.14	0.48	0.64	0.47	0.71	71.14
<i>Trabea purcelli</i>	0.13	0.36	0.63	0.43	0.70	71.84
<i>Camillina</i> sp.1	0.26	0.21	0.63	0.47	0.69	72.53
<i>Proevippa schreineri</i>	0.18	0.30	0.62	0.36	0.69	73.22
Paederinae sp.	0.19	0.25	0.62	0.34	0.68	73.90
<i>Zelotes lightfooti</i>	0.15	0.14	0.61	0.36	0.67	74.57
<i>Achmaedera</i> sp.	0.17	0.24	0.59	0.31	0.65	75.22

Species	IATs	Vineyard		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Caminara</i> sp.1	0.05	0.29	0.55	0.34	0.60	75.82
Carabidae sp.1	0.29	0.10	0.52	0.31	0.58	76.40
<i>Cophogryllus</i> sp.	0.19	0.17	0.51	0.40	0.56	76.96
<i>Galeodes</i> sp.1	0.13	0.32	0.50	0.44	0.55	77.52
<i>Synema imitator</i>	0.02	0.27	0.48	0.21	0.52	78.04
<i>Proevippa</i> sp.1	0.10	0.29	0.47	0.39	0.51	78.55
<i>Anthomyia</i> sp.	0.30	0.09	0.46	0.30	0.51	79.06
<i>Langona</i> sp.1	0.15	0.20	0.46	0.42	0.50	79.57
Calliphoridae sp.2	0.08	0.17	0.45	0.36	0.50	80.07
<i>Rothus purpurissatus</i>	0.19	0.11	0.45	0.30	0.49	80.56
Lycosidae sp.1	0.04	0.20	0.44	0.40	0.49	81.05
<i>Harmonia axyridis</i>	0.04	0.23	0.42	0.33	0.47	81.51
<i>Messor capensis</i>	0.22	0.11	0.41	0.26	0.45	81.97
Formicidae sp.1	0.12	0.15	0.40	0.32	0.44	82.40
<i>A. gracilipes</i>	0.35	0.03	0.39	0.32	0.42	82.83
Pompilidae sp.1	0.12	0.11	0.39	0.31	0.42	83.25
<i>Drassodella</i> sp.1	0.19	0.12	0.37	0.37	0.40	83.66
<i>Dermestes</i> sp.	0.02	0.33	0.37	0.21	0.40	84.06
<i>Apis mellifera</i>	0.02	0.22	0.35	0.36	0.39	84.45
Formicidae sp.3	0.10	0.14	0.35	0.19	0.39	84.84
<i>Rhodogastria amasis</i>	0.15	0.03	0.33	0.34	0.37	85.20
<i>Cleptria rufipes</i>	0.14	0.08	0.33	0.37	0.36	85.57
<i>Aelurillus</i> sp.1	0.12	0.08	0.33	0.38	0.36	85.93
<i>Steatoda</i> sp.2	0.15	0.10	0.33	0.35	0.36	86.29
<i>Phalangium</i> sp.1	0.11	0.05	0.31	0.26	0.34	86.64
<i>Anthocoris</i> sp.1	0.05	0.17	0.31	0.25	0.34	86.97
<i>Temnoptery phalerata</i>	0.12	0.09	0.30	0.35	0.33	87.30
<i>Scolopendra cingulata</i>	0.02	0.19	0.29	0.27	0.32	87.62
<i>Steatoda capensis</i>	0.17	0.06	0.29	0.30	0.32	87.93
<i>Hogna</i> sp.1	0.04	0.11	0.29	0.22	0.32	88.25
<i>Lithobius</i> sp.1	0.03	0.13	0.28	0.31	0.31	88.56
Cydnidae sp.	0.40	0.02	0.28	0.18	0.30	88.86
<i>Diores capensis</i>	0.07	0.12	0.28	0.38	0.30	89.16
<i>Camponotus</i> sp.3	0.24	0.00	0.27	0.15	0.30	89.47
<i>Paramelita nigroculus</i>	0.17	0.05	0.26	0.21	0.29	89.75
Tenebrionidae sp.1	0.14	0.06	0.26	0.27	0.28	90.04

Appendix 3e Discriminating species between vegetation cleared of invasive alien tree (CIATs) vegetation and vineyard, ordered in decreasing contribution. Average dissimilarity = 87.84.

Species	CATs	Vineyard		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Microhodotermes viator</i>	11.88	0.94	6.54	0.74	7.45	7.45
<i>Centrobolus</i> sp.2	5.66	3.31	5.84	0.80	6.65	14.09
<i>Camponotus</i> sp.2	5.11	2.01	4.38	0.88	4.99	19.08
<i>Zophosis boei</i>	3.90	2.90	4.37	0.80	4.98	24.06
<i>Linepithema humile</i>	3.18	3.71	4.34	0.80	4.95	29.01
<i>Porcello</i> sp.	2.92	1.70	3.63	0.63	4.14	33.14
<i>Tetramorium capense</i>	2.76	1.26	2.74	0.72	3.13	36.27
<i>Xestobium</i> sp.	0.39	1.88	2.32	0.42	2.64	38.91
<i>Crematogaster peringueyi</i>	1.98	0.14	1.70	0.56	1.93	40.84
<i>Gryllus bimaculatus</i>	1.03	0.73	1.45	0.85	1.65	42.49
<i>Diores youngai</i>	1.14	0.95	1.44	0.58	1.63	44.13
Halictidae sp.1	0.35	0.94	1.27	0.59	1.45	45.58
<i>Daspletis</i> sp.	1.42	0.28	1.25	0.65	1.43	47.01
<i>Cheilomenes lunata</i>	0.26	0.55	1.21	0.42	1.37	48.38
Cleridae sp.1	0.12	0.82	1.19	0.37	1.36	49.74
Curculionidae sp.1	0.14	0.57	1.16	0.56	1.32	51.06
<i>Centrobolus</i> sp.1	1.06	0.67	1.12	0.43	1.28	52.34
Hopliini sp.	0.90	0.20	1.08	0.43	1.23	53.58
Pipunculidae sp.1	0.52	0.43	0.96	0.34	1.09	54.67
<i>Achmaedera</i> sp.	0.57	0.24	0.95	0.39	1.08	55.75
<i>Sminthurus viridis</i>	0.55	0.46	0.95	0.29	1.08	56.83
Sciaridae sp.1	0.55	0.32	0.94	0.28	1.07	57.90
<i>Evarcha</i> sp.1	0.49	0.57	0.92	0.47	1.05	58.94
<i>Oxyopes</i> sp.	0.36	0.51	0.83	0.49	0.94	59.89
<i>Galeodes</i> sp.2	0.33	0.47	0.82	0.66	0.94	60.82
Sphecidae sp.2	0.38	0.33	0.81	0.38	0.93	61.75
<i>Trigonopus</i> sp.1	0.18	0.53	0.76	0.40	0.87	62.62
<i>Lithobius</i> sp.2	0.10	0.39	0.76	0.24	0.87	63.49
Noctuidae sp.2	0.04	0.39	0.76	0.39	0.87	64.35
<i>Poophilus</i> sp.1	0.23	0.31	0.71	0.49	0.81	65.17
<i>Cynthia cardui</i>	0.11	0.41	0.71	0.41	0.81	65.97
<i>Endomia elongatus</i>	0.56	0.17	0.70	0.31	0.80	66.77
Nitidulidae sp. 2	0.18	0.41	0.69	0.28	0.79	67.56
<i>Formicomus coeruleus</i>	0.08	0.48	0.68	0.45	0.78	68.34
<i>Paramelita nigroculus</i>	0.77	0.05	0.66	0.36	0.75	69.09
<i>Forficula senegalensis</i>	0.28	0.48	0.66	0.51	0.75	69.84
<i>Proevippa schreineri</i>	0.28	0.30	0.62	0.40	0.71	70.55
<i>Galeodes</i> sp.1	0.23	0.32	0.62	0.48	0.70	71.25
Paederinae sp.	0.30	0.25	0.61	0.44	0.69	71.94
<i>Phalangium</i> sp.1	1.01	0.05	0.61	0.37	0.69	72.63
<i>Trabea purcelli</i>	0.19	0.36	0.60	0.45	0.68	73.31
<i>Proevippa</i> sp. 1	0.29	0.29	0.58	0.51	0.65	73.97
<i>Synema imitator</i>	0.14	0.27	0.57	0.25	0.65	74.62

Species	CATs	Vineyard		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Caminara</i> sp.1	0.14	0.29	0.56	0.36	0.64	75.26
<i>Microlestia tabida</i>	0.71	0.10	0.53	0.35	0.61	75.87
<i>Pterotricha varia</i>	0.17	0.31	0.52	0.47	0.59	76.46
<i>Zelotes lightfooti</i>	0.30	0.14	0.49	0.47	0.56	77.02
<i>Helicoverpa armigera</i>	0.05	0.25	0.49	0.33	0.55	77.57
<i>Cophogryllus</i> sp.	0.45	0.17	0.48	0.53	0.55	78.12
<i>Tenebrio</i> sp.1	0.20	0.30	0.48	0.35	0.55	78.67
<i>Camillina</i> sp.1	0.30	0.21	0.47	0.38	0.54	79.20
Calliphoridae sp.2	0.19	0.17	0.46	0.40	0.52	79.72
<i>Langona</i> sp. 1	0.28	0.20	0.43	0.52	0.49	80.22
<i>Apis mellifera</i>	0.11	0.22	0.42	0.44	0.48	80.70
<i>Cleptria rufipes</i>	0.34	0.08	0.41	0.51	0.46	81.16
<i>Anthomyia</i> sp.	0.23	0.09	0.40	0.37	0.45	81.61
<i>Anthocoris</i> sp.1	0.16	0.17	0.39	0.29	0.45	82.06
<i>Lithobius</i> sp.1	0.17	0.13	0.39	0.40	0.44	82.50
<i>Harmonia axyridis</i>	0.01	0.23	0.38	0.31	0.44	82.94
Lycosidae sp.1	0.04	0.20	0.38	0.38	0.43	83.37
<i>Aelurillus</i> sp.1	0.36	0.08	0.37	0.44	0.42	83.80
Pompilidae sp.1	0.25	0.11	0.37	0.45	0.42	84.22
<i>Dermestes</i> sp.	0.05	0.33	0.36	0.21	0.42	84.63
<i>Messor capensis</i>	0.30	0.11	0.34	0.22	0.39	85.02
Noctuidae sp.1	0.56	0.13	0.34	0.26	0.39	85.41
<i>Drassodella</i> sp.1	0.17	0.12	0.34	0.29	0.39	85.80
<i>Sphex bonaspei</i>	0.31	0.02	0.32	0.38	0.36	86.16
<i>Steatoda</i> sp.2	0.23	0.10	0.31	0.45	0.35	86.51
Formicidae sp.1	0.07	0.15	0.31	0.27	0.35	86.86
Formicidae sp.3	0.19	0.14	0.28	0.21	0.32	87.18
Mordellidae sp.2	0.14	0.12	0.28	0.24	0.32	87.50
<i>Rothus purpurissatus</i>	0.18	0.11	0.28	0.41	0.31	87.81
<i>Diaphorocellus</i> sp.1	0.17	0.06	0.27	0.37	0.31	88.13
Nitidulidae sp.1	0.23	0.01	0.27	0.22	0.31	88.43
Muscidae sp.	0.21	0.05	0.27	0.20	0.31	88.74
<i>Temnopteryx phalerata</i>	0.25	0.09	0.27	0.47	0.31	89.04
<i>Meliponula</i> sp.1	0.03	0.21	0.26	0.30	0.30	89.34
Tachinidae Sp.3	0.14	0.06	0.26	0.28	0.29	89.63
<i>Scolopendra cingulata</i>	0.03	0.19	0.25	0.25	0.29	89.92
<i>Theridion</i> sp.	0.00	0.50	0.25	0.18	0.29	90.21

Appendix 3f Discriminating species between vegetation cleared of invasive alien tree (CIATs) and invasive alien trees (IATs), ordered in decreasing contribution. Average dissimilarity = 86.93.

Species	CATs	IATs		Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund	Av.Diss			
<i>Microhodotermes viator</i>	11.88	2.40	7.22	0.78	8.31	8.31
<i>Centrobolus</i> sp.2	5.66	5.92	7.03	0.97	8.09	16.40
<i>Porcello</i> sp.	2.92	4.59	5.99	0.80	6.89	23.29
<i>Camponotus</i> sp.2	5.11	0.87	3.90	0.85	4.49	27.78
<i>Tetramorium capense</i>	2.76	1.27	3.20	0.69	3.68	31.46
<i>Linepithema humile</i>	3.18	0.80	2.79	0.75	3.21	34.67
<i>Zophosis boei</i>	3.90	0.20	2.61	0.73	3.00	37.67
<i>Crematogaster peringueyi</i>	1.98	1.06	2.56	0.68	2.94	40.61
<i>Centrobolus</i> sp.1	1.06	0.91	2.21	0.49	2.54	43.16
<i>Gryllus bimaculatus</i>	1.03	0.75	1.85	0.66	2.13	45.28
Sciaridae sp.1	0.55	1.19	1.70	0.49	1.96	47.24
Pipunculidae sp.1	0.52	1.02	1.69	0.41	1.95	49.19
Hopliini sp.	0.90	0.67	1.66	0.37	1.91	51.09
<i>Diores youngai</i>	1.14	0.69	1.59	0.67	1.83	52.92
<i>Cecidomyia</i> sp.1	0.11	1.77	1.53	0.30	1.76	54.68
<i>Xestobium</i> sp.	0.39	0.88	1.40	0.42	1.61	56.29
<i>Daspletis</i> sp.	1.42	0.33	1.31	0.63	1.51	57.80
<i>Endomia elongatus</i>	0.56	0.46	1.13	0.35	1.30	59.10
<i>Tenebrio</i> sp.1	0.20	0.36	0.90	0.29	1.04	60.14
<i>Achmaedera</i> sp.	0.57	0.17	0.88	0.37	1.01	61.15
<i>Sminthurus viridis</i>	0.55	0.31	0.84	0.31	0.96	62.12
<i>Oxyopes</i> sp.	0.36	0.28	0.81	0.36	0.93	63.05
<i>Paramelita nigroculus</i>	0.77	0.17	0.81	0.39	0.93	63.98
Sphécidae sp.2	0.38	0.31	0.77	0.41	0.89	64.87
<i>Phalangium</i> sp.1	1.01	0.11	0.70	0.41	0.80	65.67
<i>Cophogryllus</i> sp.	0.45	0.19	0.63	0.51	0.73	66.40
<i>Camillina</i> sp.1	0.30	0.26	0.62	0.44	0.72	67.11
<i>Anthomyia</i> sp.	0.23	0.30	0.59	0.38	0.68	67.79
<i>Zelotes lightfooti</i>	0.30	0.15	0.56	0.36	0.65	68.44
<i>Cynthia cardui</i>	0.11	0.26	0.55	0.24	0.63	69.07
<i>Forficula senegalensis</i>	0.28	0.14	0.53	0.44	0.61	69.68
<i>Proevippa schreineri</i>	0.28	0.18	0.53	0.40	0.61	70.28
Paederinae sp.	0.30	0.19	0.52	0.33	0.60	70.88
<i>Rothus purpurissatus</i>	0.18	0.19	0.51	0.34	0.58	71.46
<i>Cleptria rufipes</i>	0.34	0.14	0.50	0.52	0.58	72.04
<i>Microlestia tabida</i>	0.71	0.02	0.49	0.33	0.57	72.61
<i>Aelurillus</i> sp.1	0.36	0.12	0.49	0.47	0.56	73.17
<i>Pterotricha varia</i>	0.17	0.12	0.48	0.35	0.56	73.72
Halictidae sp.1	0.35	0.17	0.48	0.56	0.55	74.28
Curculionidae sp.1	0.14	0.10	0.48	0.28	0.55	74.83
<i>Proevippa</i> sp. 1	0.29	0.10	0.46	0.39	0.53	75.36
<i>Evarcha</i> sp.1	0.49	0.07	0.46	0.50	0.53	75.90
Pompilidae sp.1	0.25	0.12	0.46	0.40	0.53	76.43

Species	CATs	IATs	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
Cleridae sp.1	0.12	0.23	0.46	0.20	0.53	76.95
<i>Langona</i> sp.1	0.28	0.15	0.45	0.46	0.51	77.47
<i>Poophilus</i> sp.1	0.23	0.11	0.43	0.46	0.49	77.96
<i>Helicoverpa armigera</i>	0.05	0.41	0.42	0.30	0.49	78.45
Nitidulidae sp.2	0.18	0.18	0.42	0.26	0.48	78.93
<i>Cheilomenes lunata</i>	0.26	0.07	0.41	0.28	0.47	79.41
Nitidulidae sp.1	0.23	0.15	0.41	0.31	0.47	79.88
<i>A. gracilipes</i>	0.08	0.35	0.40	0.34	0.46	80.34
<i>Steatoda</i> sp.2	0.23	0.15	0.39	0.43	0.45	80.79
<i>Galeodes</i> sp.1	0.23	0.13	0.39	0.43	0.45	81.24
<i>Temnopteryx phalerata</i>	0.25	0.12	0.39	0.45	0.44	81.68
Carabidae sp.1	0.03	0.29	0.38	0.32	0.44	82.12
<i>Drassodella</i> sp.1	0.17	0.19	0.38	0.32	0.43	82.56
<i>Galeodes</i> sp.2	0.33	0.03	0.37	0.47	0.43	82.99
<i>Messor capensis</i>	0.30	0.22	0.37	0.31	0.43	83.42
<i>Trabea purcelli</i>	0.19	0.13	0.36	0.36	0.42	83.83
Formicidae sp.3	0.19	0.10	0.35	0.25	0.40	84.24
<i>Sphex bonaspei</i>	0.31	0.03	0.34	0.40	0.39	84.63
<i>Trigonopus</i> sp.1	0.18	0.13	0.34	0.39	0.39	85.02
<i>Steatoda capensis</i>	0.17	0.17	0.34	0.35	0.39	85.41
<i>Camponotus</i> sp.3	0.08	0.24	0.34	0.18	0.39	85.79
<i>Opisththalmus macer</i>	0.19	0.18	0.33	0.24	0.38	86.17
Tenebrionidae sp.1	0.07	0.14	0.31	0.30	0.36	86.53
<i>Diaphorocellus</i> sp.1	0.17	0.10	0.31	0.42	0.36	86.89
<i>Scarabaeus rigosus</i>	0.16	0.09	0.30	0.34	0.34	87.23
Muscidae sp.	0.21	0.06	0.29	0.20	0.34	87.57
<i>Lithobius</i> sp.2	0.10	0.08	0.28	0.23	0.33	87.90
Calliphoridae sp.2	0.19	0.08	0.27	0.32	0.31	88.21
Cydnidae sp.	0.00	0.40	0.27	0.18	0.31	88.52
<i>Nomia amabilis</i>	0.16	0.12	0.27	0.25	0.31	88.82
<i>Rhodogastria amasis</i>	0.02	0.15	0.26	0.32	0.30	89.12
Syrphidae sp.1	0.10	0.04	0.25	0.21	0.29	89.41
<i>Anthocoris</i> sp.1	0.16	0.05	0.24	0.30	0.28	89.69
<i>Dejeania</i> sp.	0.16	0.13	0.22	0.36	0.26	89.95
Formicidae sp.1	0.07	0.12	0.22	0.34	0.25	90.21

Chapter 4 – Ecotone Effects on Arthropod Diversity in the Cape Floristic Region

INTRODUCTION

Habitat loss and fragmentation

Landscape fragmentation and associated habitat loss, along with impact of invasive alien organisms, are by far the two greatest threats to biodiversity (Saunders et al., 1991; Macdonald and Richardson, 1986; Wilcove et al., 1986; Clark and Downes, 1996; Edwards and Abivardi, 1998; Gibbs and Stanton, 2001; Magura et al., 2001; Niemelä, 2001; Wolters, 2001; Tschamntke et al., 2002; Freemark et al., 2002; Fahrig, 2003; Peintinger et al., 2003; Le Maitre et al., 2004; Schoereder et al., 2004; Hansen et al., 2005; Hoekstra et al., 2005; Radeloff et al., 2005; Bailey, 2007; Harvey et al., 2008), yet relatively few studies have examined the effects of fragmentation on ecological communities (Barrett and Bohlen, 1991; Saunders et al., 1991; Soulé et al., 1992; Andrén, 1994; Forman, 1995; Bolger et al., 1997; Collinge, 2000; Burel et al., 2000; Debinski and Holt, 2000; Davies et al., 2001; Crooks, 2002; Samways, 2005). Habitat fragmentation is often defined as a process by which “a large intact habitat (containing continuous habitat) is transformed into a number of smaller patches (fragments) of smaller total area, isolated from each other by a matrix of habitats unlike the original” (Wilcove et al., 1986; Bowers and Dooley, 1999; Groppe et al., 2001; Mossman and Waser, 2001; Cascante et al., 2002). This often takes place through the dominance of agricultural fields in an area (Ekbom, 2000; Kruess and Tschamntke, 2000). In turn, fragmentation often leads to a close juxtaposition of disturbed patches with natural patches. However, it remains unclear to what extent the edge effects of the disturbed patches influence arthropod diversity in the natural area.

An understanding of the distribution of individual species is important in order to understand the impacts of habitat fragmentation on species persistence (Tschamntke et al., 2002; Ries et al., 2004; Ranius, 2006; Fraser et al., 2008) and extinction thresholds (Fahrig, 2001), and whether there will be any conservation benefit to some animal species (Markovchick-Nicholls et al., 2008). However, there is a distinct lack of knowledge regarding the distribution of arthropod species living in fragmented

landscapes (Tschardt et al., 2002; Dolman and Fuller, 2003; Bowne and Bowers, 2004). This is especially true for fynbos vegetation, which is one of many natural areas that have become increasingly fragmented as a result of human activities, such as expansion of agricultural activities and urban development.

Habitat edges are an important feature in the Cape Floristic Region (CFR) landscapes, due to increasing rates of habitat loss and native fynbos vegetation fragmentation. As a result, there has been increasing landscape heterogeneity as continuous patches of native habitat are broken into many smaller, isolated patches surrounded by a matrix of different, often heavily disturbed habitats (Donaldson et al. 2002). As human communities and alien species spread into fynbos vegetation, the option of protecting natural vegetation and associated organisms becomes increasingly important. The boundaries or ecotones between the native fynbos vegetation and vineyards or plantation forestry are usually sharp, and often maintained by both natural and, more recently, anthropogenic fires (Donaldson et al. 2002).

Ecotones are defined as boundaries between two distinct fragments of vegetation or land-use classes (i.e., forests, meadows, scrub, agriculture, urban areas, etc.) within terrestrial landscapes (Forman, 1995; Ries et al., 2004), mainly as a result of vegetation disturbance (Johnston et al., 1992), although others occur naturally. Disturbance is any relatively discrete activity in time that disrupts ecosystems, assemblage or population structure and changes resources, substrate availability, or the physical environment (White and Pickett, 1985). At ecotones, environmental conditions usually differ from those of the surroundings (Johnson et al., 1992). However, an ecotone may or may not have distinct vegetation structure relative to the adjacent vegetation (Gordon and Forman, 1983), which may affect arthropod distribution (Gould and Stinner, 1984). The proportion of species negatively affected by anthropogenic disturbances is much higher than those that benefit from it (e.g. Bowers and Breland, 1996). In general, vegetation disturbance often favours communities which are recognized as pests (Woodwell, 1983), or at least are generalists.

Conservationists and land managers are increasingly charged with meeting multiple, often conflicting goals in landscapes undergoing significant change as a result of shifting land use. Such changes in the land use often have profound ecological impacts at the individual, population, and ecosystem levels (Wilcove et al., 1986). These

impacts occur both within, and adjacent to, the fragment under active management, forming new and extensive boundary between habitat types. Ecotones established between management areas, for example between plantation forestry patches agricultural fields and adjacent natural fynbos vegetation, inevitably lead to differences in the quality of habitats on either side of the boundary, which results in a habitat edge. Although boundaries are common components of undisturbed landscapes, the amount of boundaries proliferates rapidly as landscapes become fragmented (Sisk and Haddad, 2002).

The functional properties of landscapes depend mostly on a variety of interactions between patches and boundaries or edges (Hansen et al., 1992; Ries et al., 2004). For specialist species, the behaviour near vegetation boundaries has important consequences for the dispersion of individuals (Tischendorf and Wissel, 1997). The function of boundaries includes conduit (guiding structure), filter or barrier, source, sink or habitat providing resources (Hansen et al., 1988; Forman and Moore, 1992; Janzen, 1983; Wratten et al., 2003).

Insect dispersion across habitat fragments

Distribution of surface-dwelling arthropods between adjacent different habitat fragments has been studied in various parts of the world (Altieri and Schmidt, 1986; Mader, 1990; Duelli et al., 1990), including South Africa (Ingham and Samways, 1996; Kotze and Samways, 1999; Samways et al. 1996). Insect dispersion across agricultural fields and boundaries/ecotones is affected by the nature of the crop, the size of the field and structure or nature of the boundary (Wratten and van Emden, 1995). Tall barriers of plantation forestry may have a significant impact on flying insects by altering their flight paths (Wood and Samways, 1991), while small patches of pines in grassland limit the local distribution of grasshoppers (Samways and Moore, 1991). The variation in arthropod diversity within an agroecosystem often depends on the biodiversity of the surroundings, rather than on differing land-use or agricultural intensity (Duelli, et al., 1999; Tschardt et al., 2005).

Habitat specialists can respond strongly to the edges differing in structure, ranging from high contrast (treeline) to low-contrast (field) (Ries and Debinski, 2001). Therefore, specialist habitat individuals are less likely to escape habitat disturbance than habitat

generalist species (Ries and Debinskit, 2001; Sisk and Battin, 2002; Kotze and O'Hara, 2003).

The ecotone as arthropod habitat

As native habitats become smaller and more irregularly shaped, they become increasingly dominated by ecotone habitat. This means that, understanding the ecology of ecotones is critical for landscape ecology, biodiversity conservation and management decisions. Ecotones may provide unique habitats optimal to some arthropod species while inhospitable to others (Hansen et al., 1988). Being unique, ecotones do not necessarily have higher diversity of species compared to the neighboring environments (di Castri and Hansen, 1992).

More evidence and accumulating theory indicate that with increased fragmentation and expanded ecotones, there may be an increase in local species richness (Forman, 1995; Whittaker, 1972; Magura et al., 2001). Furthermore, ecotones tend to be sunnier, warmer, drier and more suitable for invasive alien species, shade intolerant species and the generalist predators at the expense of indigenous species (Harper et al., 2005).

Common assumptions about edge effects

The effects of habitat fragmentation may include loss of species from isolated fragments or changes in species abundances among habitats that differ in structure, or edge characteristics (Forman, 1995; Golden and Crist, 2000; Kruess and Tschardtke, 2000). Recent findings have raised concern on most of these assumptions, leaving the conservation manager with little scientific support for decision-making (Sisk and Haddad, 2002). Concerns are that edge effects are similar for all related species, and they extend some characteristics, a fixed distance from the habitat edge into a patch (i.e. species are assumed to respond similarly to edges, up to some fixed distance into a habitat patch). For a given arthropod species, edge effects are consistent, regardless of the type of edge (i.e. a particular species response to edges, is characteristic of the species in general), while species respond similarly to the same type of edge in different locations.

Arthropod response to ecotone

Many factors influence the distribution and abundance of arthropods throughout their native and introduced ranges. Ecotones may be attractive to many arthropod species

(Ries and Fagan, 2003), resulting in relatively higher species densities at the ecotone than in adjacent habitats. However, species dispersion across adjacent, distinct habitats may be influenced by the boundaries, even when they are clearly capable of crossing them (Sisk and Battin, 2002). Leopold (1933) popularized the term “edge effect”, a hypothesis that edges are beneficial to wildlife. However, it was later acknowledged that not all species respond similarly to habitat alterations and ecotones (Brittingham and Temple, 1983; Wilcove et al., 1986). Thus, there is much contention as to the role of ecotones with a distinct lack of information from southern Africa, especially the CFR.

Aims of this chapter

The response of individuals in response to patch boundaries is a crucial element in many dispersal models (Matthysen, 2002). However, it is not clear to what extent the edge effects of the disturbed patches influence arthropod diversity in the different vegetation types. As a result of competing interests for land-use, great emphasis needs to be focused on the efficiency of a particular network of vegetation representing relatively higher biodiversity. As a result, the main objectives of this study were: 1) to determine whether arthropod species richness, abundance and diversity are higher in the ecotone than in the adjacent habitat fragments, 2) to determine the significance of adjacent distinct vegetation types on arthropod family or species, and 3) to determine whether there is any correlation between varying distances from ecotone and associated arthropod species richness and abundance. In this regard, different groups of surface-active arthropods were used to test the edge-effect hypothesis (Leopold, 1933).

STUDY SITES

Study sites included habitat boundaries/ecotones between the major, well-represented habitat types in the Cape Winelands of the Cape Floristic Region (CFR). Based on the most common adjacent land and boundary characteristics of CFR fynbos, six ecotone types were chosen for study: alien cleared/fynbos, alien cleared/alien invaded, fynbos/alien invaded, alien cleared/vineyard, vineyard/alien invaded, and vineyard/fynbos. Plantation forestry (gums and pines) were considered here as alien trees.

➤ *See Chapter 2 for details of study sites*

MATERIAL AND METHODS

The diversity of surface-active arthropods between various landscape types (fynbos/alien invaded; vineyard/fynbos; alien cleared/fynbos; vineyard/alien invaded; alien cleared/vineyard and alien cleared/alien invaded) were studied. Surface-dwelling arthropod taxa were collected using pitfall traps, which allow for standardized sampling and comparative analysis. Multiple sampling stations were established along transects running orthogonal to transitions between distinct vegetation types. Layout of the pitfall traps followed a \log^2 scale away from the ecotones, thus giving increased focus in the region of the ecotone. Sampling stations extended 128 m into both the adjoining vegetation types. Arthropod sampling was from August 2006 to January 2008 (i.e. August-October 2006, May-July 2007 and November-January 2007/8 (November and December 07- January 08)). The collected surface-active arthropods were sorted and placed into families (Appendix 4.a), and where possible, they were further identified to species. Species identifications were undertaken or verified by taxonomic specialists: Araneae and Solifugae (Dr. Ansie Dippenaar-Schoeman of ARC-Plant Protection Research Institute), other arthropods (Prof. Henk Geertsema and Mr. Patrick Reavel of Stellenbosch University).

➤ *See Chapter 2 for details of Material and Methods*

DATA ANALYSIS

To gain the measure of the thoroughness of sampling, species accumulation curves were produced, using EstimateS version 8.0.0 with samples randomized 50 times (Colwell, 2006). These curves were plotted for all six ecotones (vegetation type combinations): fynbos/alien invaded; vineyard/fynbos; alien cleared/fynbos; vineyard/alien invaded; alien cleared/vineyard and alien cleared/alien invaded ecotones. An overall observed (Sobs (Mao Tau)) and estimated (Chao 2 mean) species accumulation curve for all ecotones combined were also produced. A variety of non-parametric species estimators were used to provide the best overall arthropod species estimates for all the vegetation types (Hortal et al., 2006). Incidence based Coverage Estimator (ICE) was considered a robust and accurate estimator of species richness (Chazdon et al., 1998), whereas Chao2 and Jackknife estimators provide the least biased estimates for insufficient sampling (Colwell and Coddington, 1994). Therefore,

these estimators were calculated using EstimateS (Colwell, 2006) for all the ecotones separately, and for combined ecotones. Mean species richness and abundance, standard deviation, variance and standard errors were calculated for each ecotone type using a statistical analysis program, SPSS version 15.0 for Window (SPSS Inc., 2006). One-way analysis of variance (ANOVA) was performed on the species and the log-transformed abundance data for the ecotones, with multiple comparisons of the means using the Bonferroni methods (Legendre and Legendre, 1998).

A variety of diversity indices (d = Margalef's index, a measure of the number of species present for a given number of individuals; H' = Shannon diversity index; and $1-\text{Lambda}'$ = Simpson index) was calculated for arthropods in the different ecotones using PRIMER v5.

Species-specific responses to the ecotone were expressed as ecotone response curves. The resultant arthropod data allowed fitting of a regression (ecotone response curve) of species richness and abundance for each ecotone type. Various ecotone response curves were used for dominant taxa, showing either declines, constant levels or increases in species richness and abundance across ecotones. An ecotone was considered to be the centre of the regression line, that is, distance 0 represented the ecotone. As a result, the response for selected species for each ecotone type was estimated. Correlations between species richness and abundance with the distance from the ecotone were calculated using Spearman's Rank Order Coefficient.

RESULTS

Species diversity

In total, 197 species from 107 families and 24 orders were recorded, from 24 239 individuals collected. The species estimators for all sites combined were: ICE = 202.7, Chao2 = 206.41 (SD \pm 7.25) and Jackknife2 = 214.96. Alien cleared/fynbos ecotone sites had 133 observed species from 92 families and 20 orders, with species estimator scores of: ICE = 143.91, Chao2 = 143.43 (SD \pm 6.25) and Jackknife2 = 155.89. Alien cleared/alien invaded ecotone sites had 104 observed species from 84 families and 21 orders, with species estimator scores of: ICE = 118.73, Chao2 = 115.39 (SD \pm 6.5) and Jackknife2 = 128.87. Fynbos/alien invaded ecotone sites had 99 observed species from 88 families and 22 orders, with species estimator scores of:

ICE = 108.57, Chao2 = 105.11 (SD \pm 4.34) and Jackknife2 = 113.03. Alien cleared/vineyard ecotone sites had 128 observed species from 93 families and 22 orders, with species estimator scores of: ICE = 141.29, Chao2 = 145.86 (SD \pm 9.85) and Jackknife2 = 160.64. Vineyard/alien invaded ecotone sites had 118 observed species from 85 families and 20 orders, with species estimator scores of: ICE = 144.72, Chao2 = 152.78 (SD \pm 16.16) and Jackknife2 = 169.32. Vineyard/fynbos ecotone sites had 108 observed species from 80 families and 20 orders, with species estimator scores of: ICE = 114.5, Chao2 = 112.06 (SD \pm 3.22) and Jackknife2 = 116.18.

Species accumulation curves for individual ecotones, fynbos/alien invaded; vineyard/fynbos; alien cleared/fynbos; vineyard/alien invaded; alien cleared/vineyard and alien cleared/alien invaded, although flattening slightly, did not reach asymptotes (Figure 4.1) The lowest species accumulation curve is the fynbos/alien invaded ecotone. However, alien cleared/fynbos had resulted in greater abundance than any other ecotone. Alien cleared/vineyard and vineyard/alien invaded ecotone sites had the highest species accumulation (Figure 4.1). In fact, it had doubled the species abundance from other ecotones. Overall observed and estimated species accumulation curves showed a very similar pattern, although the observed species richness was lower than estimated (Figure 4.2). Nevertheless, the overall observed species accumulation curve reached an asymptote after taking over 420 samples, and was comparable to the estimated species accumulation curve (Figure 4.2).

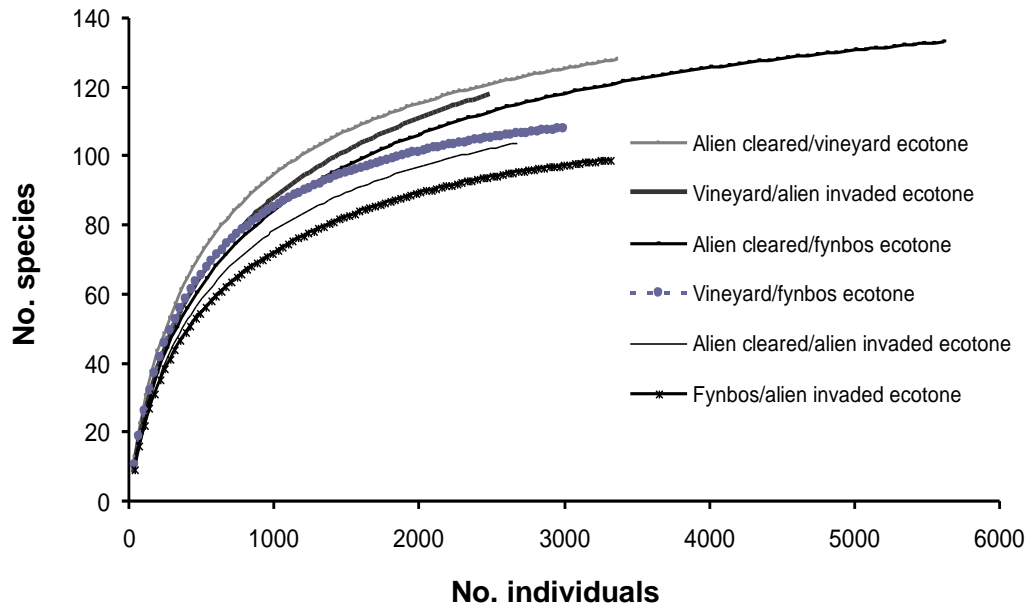


Figure 4.1 Species accumulation curves for the six ecotones: alien cleared/fynbos, alien cleared/alien invaded, fynbos/alien invaded, alien cleared/vineyard, vineyard/alien invaded, and vineyard/fynbos ecotones.

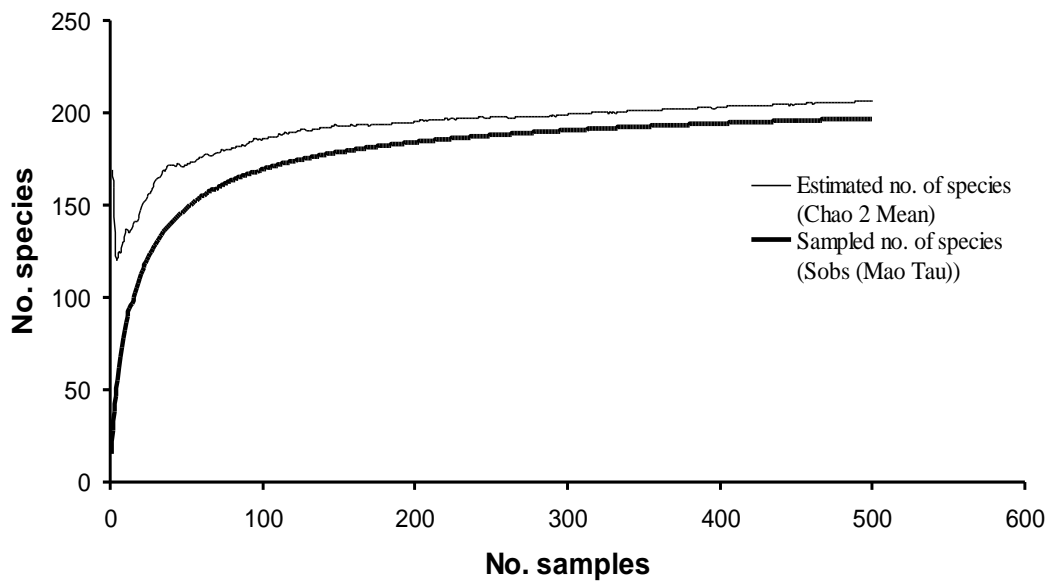


Figure 4.2 An overall estimated and observed species accumulation curves for the alien cleared/fynbos, alien cleared/alien invaded, fynbos/alien invaded, alien cleared/vineyard, vineyard/alien invaded, and vineyard/fynbos ecotone combined.

Distribution of arthropods between adjacent vegetation types

Different pairs of adjacent vegetation types supported considerably different numbers of species, with the highest mean species richness and abundance recorded for the alien cleared/fynbos ecotone (Table 4.1). Vineyard/alien invaded ecotone had the lowest mean species richness and abundance. Where aliens were adjacent to fynbos, both species richness and abundance increased (Table 4.1). However, where aliens were adjacent to the alien cleared sites, the difference in species richness was just above zero, although there was a sharp increase in abundance. The number of species per each vegetation pair ranged from 55-83 for alien cleared/fynbos, and 41-62 for vineyard/alien invaded ecotone (Table 4.1).

Overall differences between adjacent patches

Nested ANOVA among alien cleared/fynbos, alien cleared/alien invaded, fynbos/alien invaded, alien cleared/vineyard, vineyard/alien invaded, and vineyard/fynbos ecotones had indicated overall significant difference between adjacent vegetation types ($df = 36$, Wilks lambda = 0.32498, $F(10, 154) = 11.614$, $p = 0.000$) in terms of both species richness and abundance.

Figure 4.3 and Figure 4.4 show the comparison of individual ecotones in terms of arthropod species richness and abundance respectively. There were no statistically significant differences among some ecotones (i.e. alien cleared/alien invasive, fynbos/alien invasive and vineyard/fynbos ecotones) in terms of species richness. Both Figure 4.3 and Figure 4.4 have a similar pattern, such that species richness was reflected by abundance at each ecotone. Generally, alien invaded vegetation type was associated with lower species richness, whatever the adjacent vegetation type. However, the alien cleared/vineyard ecotone also had a significantly higher species richness compared to vineyard/alien invaded ecotone (Figure 4.3). In contrast, the alien cleared/fynbos ecotone had significantly higher species diversity than any other ecotone. Detailed species data and summary statistics are given in Appendices 4a-h.

Table 4.1 Species richness and abundance means, standard deviation (Std.Dev), variance and Standard error (Std.Err.) for the alien cleared/fynbos (AC/FB), alien cleared/alien invaded (AC/AI), fynbos/alien invaded (FB/AI), alien cleared/vineyard (AC/VY), vineyard/alien invaded (VY/AI), and vineyard/fynbos (VY/FB) ecotones.

Arthropod variable	Adjacent vegetation types	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean			
						Lower Bound	Upper Bound	Min.	Max.
Species richness	AC/FB	6	71.29	7.087	1.894	67.19	75.38	55	83
	AC/AI	6	50.50	9.843	2.631	44.82	56.18	33	66
	FB/AI	6	59.50	11.058	2.955	53.12	65.88	39	79
	AC/VY	6	63.43	6.676	1.784	59.57	67.28	52	74
	VY/AI	6	49.86	5.749	1.537	46.54	53.18	41	62
	VY/FB	6	59.64	7.772	2.077	55.16	64.13	48	75
	<i>Total</i>	36	<i>59.04</i>	<i>10.908</i>	<i>1.190</i>	<i>56.67</i>	<i>61.40</i>	<i>33</i>	<i>83</i>
Abundance	AC/FB	6	449.43	89.207	23.842	397.92	500.94	297	608
	AC/AI	6	236.14	73.308	19.593	193.82	278.47	123	408
	FB/AI	6	292.50	56.140	15.004	260.09	324.91	175	357
	AC/VY	6	280.21	50.301	13.443	251.17	309.26	190	371
	VY/AI	6	214.21	36.015	9.626	193.42	235.01	151	269
	VY/FB	6	273.79	66.829	17.861	235.20	312.37	186	441
	<i>Total</i>	36	<i>291.05</i>	<i>98.420</i>	<i>10.739</i>	<i>269.69</i>	<i>312.41</i>	<i>123</i>	<i>608</i>

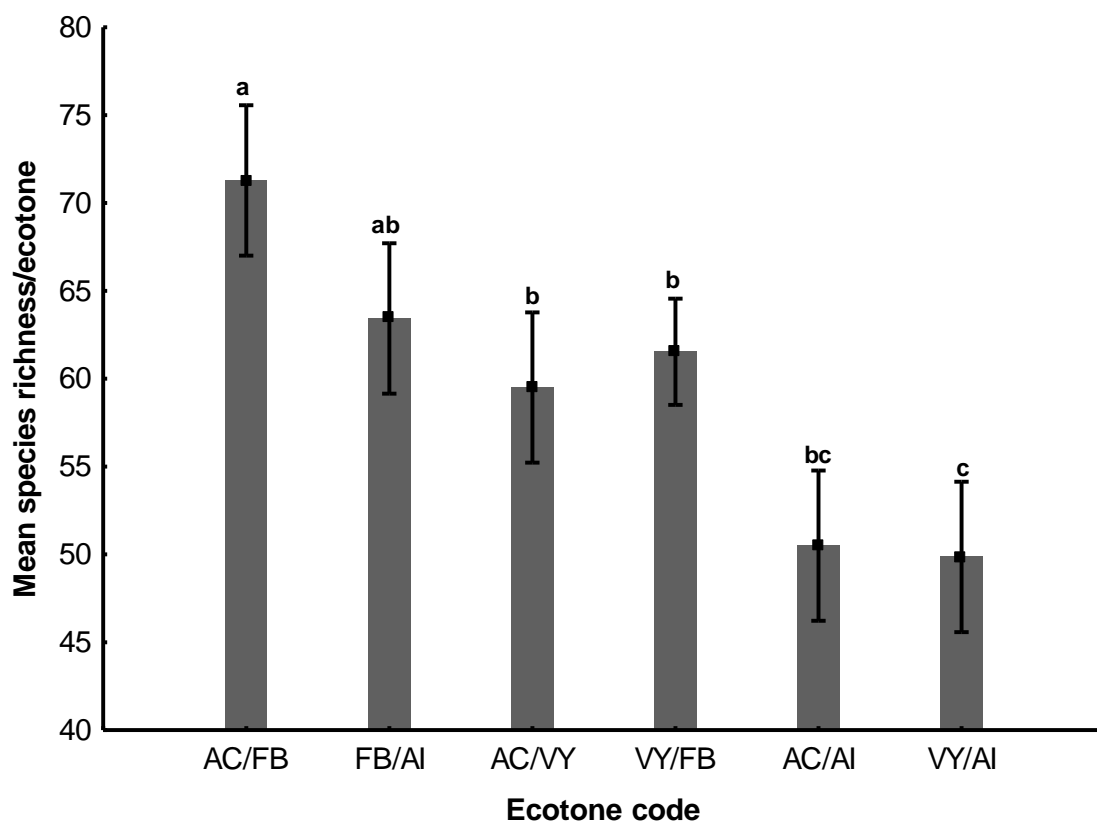


Figure 4.3 Mean arthropod species richness for each ecotone: alien cleared/fynbos (AC/FB), alien cleared/alien invaded (AC/AI), fynbos/alien invaded (FB/AI), alien cleared/vineyard (AC/VY), vineyard/alien invaded (VY/AI), and vineyard/fynbos (VY/FB) ecotones. Vertical bars denote \pm standard errors. Different letters above vertical bars indicate a significant difference (i.e. AC/FB and AC/AI).

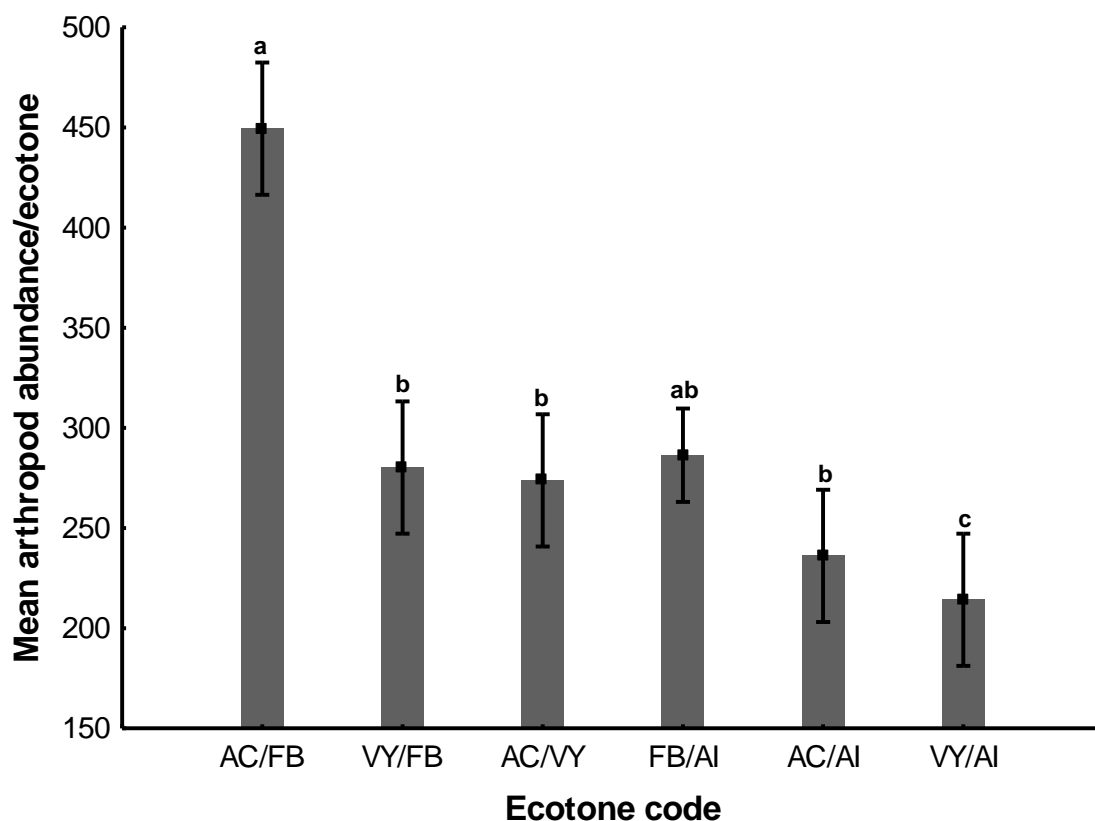


Figure 4.4 Mean arthropod abundance for each ecotone: alien cleared/fynbos (AC/FB), alien cleared/alien invaded (AC/AI), fynbos/alien invaded (FB/AI), alien cleared/vineyard (AC/VY), vineyard/alien invaded (VY/AI), and vineyard/fynbos (VY/FB) ecotones. Vertical bars denote \pm standard errors. Different letters above vertical bars indicate a significant difference.

Diversity measures

Generally, higher arthropod diversity was measured in alien cleared/fynbos ecotone, and the least in the vineyard/IATs. Both Shannon diversity index (H') and Simpson index ($1-\text{Lambda}'$) measured lower diversity for the vineyard/IATs ecotones (Table 4.2).

Table 4.2 Univariate diversity indices (d = Margalef's index, H' = Shanon diversity index and $1-\text{Lambda}'$ = Simpson index) for alien cleared/fynbos (AC/FB), alien cleared/alien invaded (AC/AI), fynbos/alien invaded (FB/AI), alien cleared/vineyard (AC/VY), vineyard/alien invaded (VY/AI), and vineyard/fynbos (VY/FB) ecotones.

Ecotone	d	H'(loge)	1-Lambda'
AC/FB	19.62	3.990	0.947
AC/AI	12.52	2.598	0.833
FB/AI	16.57	2.906	0.847
AC/VY	16.94	3.192	0.868
VY/AI	8.5	1.763	0.565
VY/FB	15.57	2.906	0.796

Arthropod species richness and abundance patterns along the ecotones

There was a remarkably similar 'bird wing' pattern for both species richness (Figures 4.5-4.6) and abundance (Figures 4.7-4.8) across all the ecotones, with a peak at the centre of the ecotone, then a slight decrease through either side, and then an overall increase further into each patch.

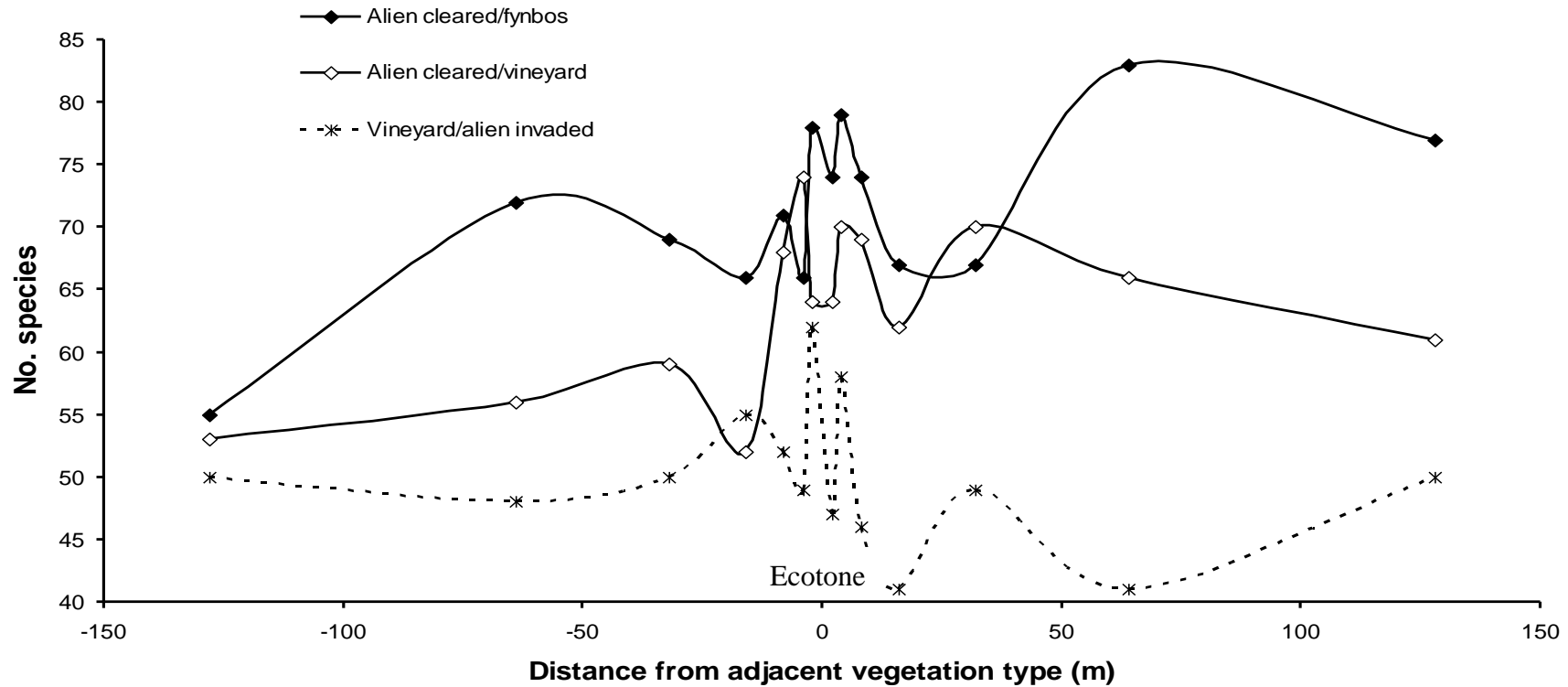


Figure 4.5 Arthropod species richness along the ecotones: alien cleared/fynbos, alien cleared/vineyard, and vineyard/alien invaded ecotones. Negative distance (-) represents the first mentioned vegetation type of the pair. For example, alien cleared/fynbos: negative distance represents alien cleared, while positive distance represent fynbos. 0 m = ecotone.

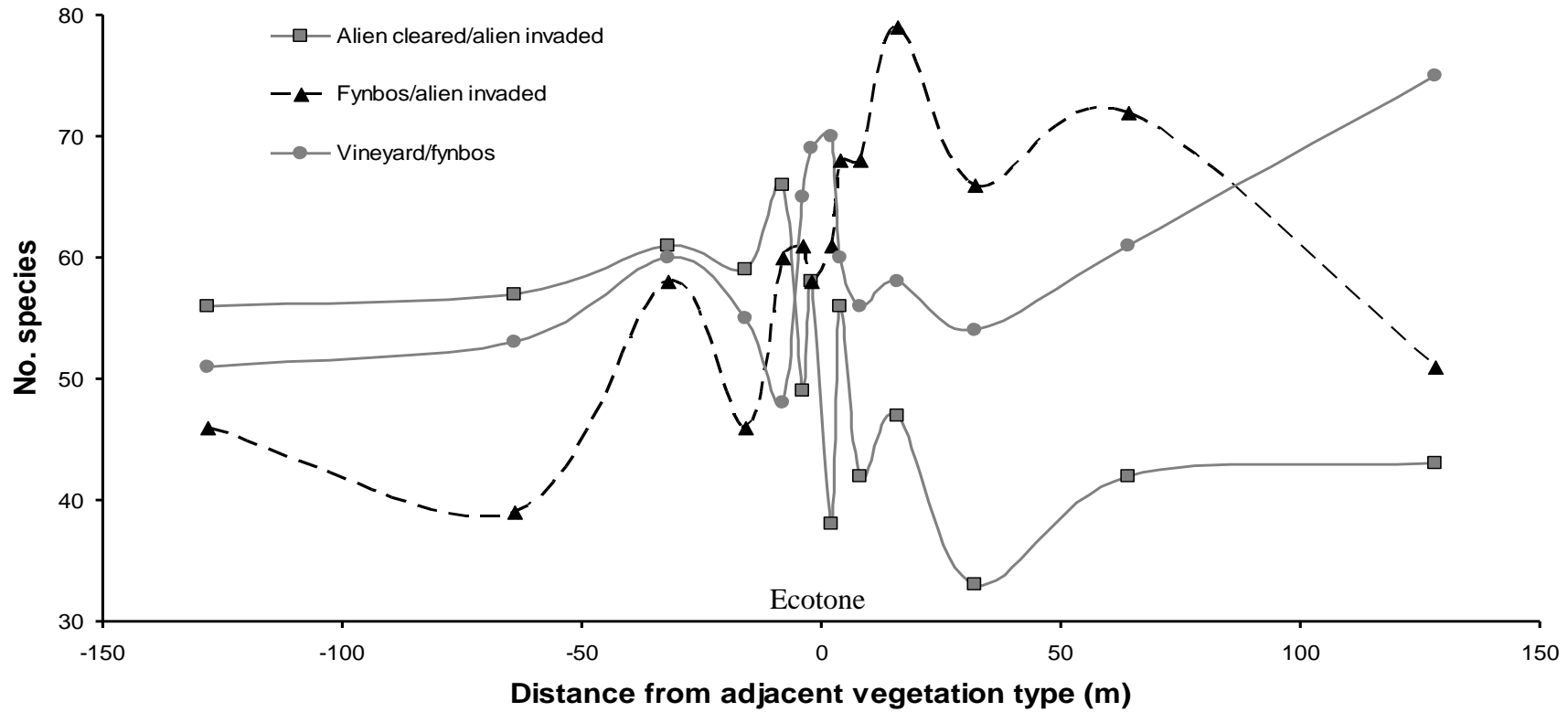


Figure 4.6 Arthropod species richness along the ecotones: alien cleared/alien invaded, fynbos/alien invaded, and vineyard/fynbos ecotones. Negative distance (-) represents the first mentioned vegetation type of the pair. For example, alien cleared/alien invaded: negative distance represents alien cleared, while positive distance represents alien invaded. 0 m = ecotone.

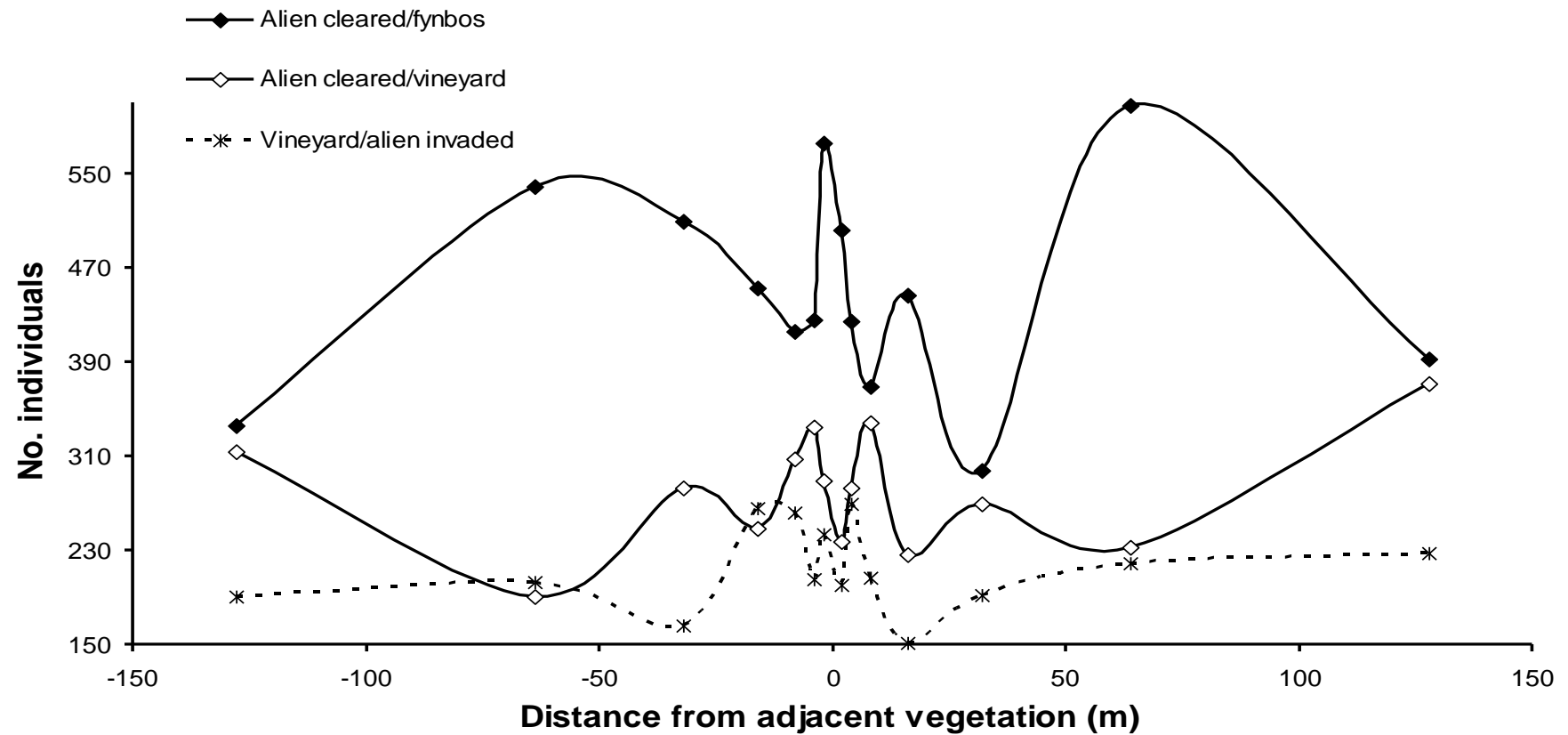


Figure 4.7 Arthropod abundance along alien cleared/fynbos, alien cleared/vineyard, and vineyard/alien invaded ecotones. Negative distance (-) represent the first mentioned vegetation type of the pair. For example, alien cleared/fynbos: negative distance represents alien cleared, while positive distance represents fynbos. 0 m = ecotone.

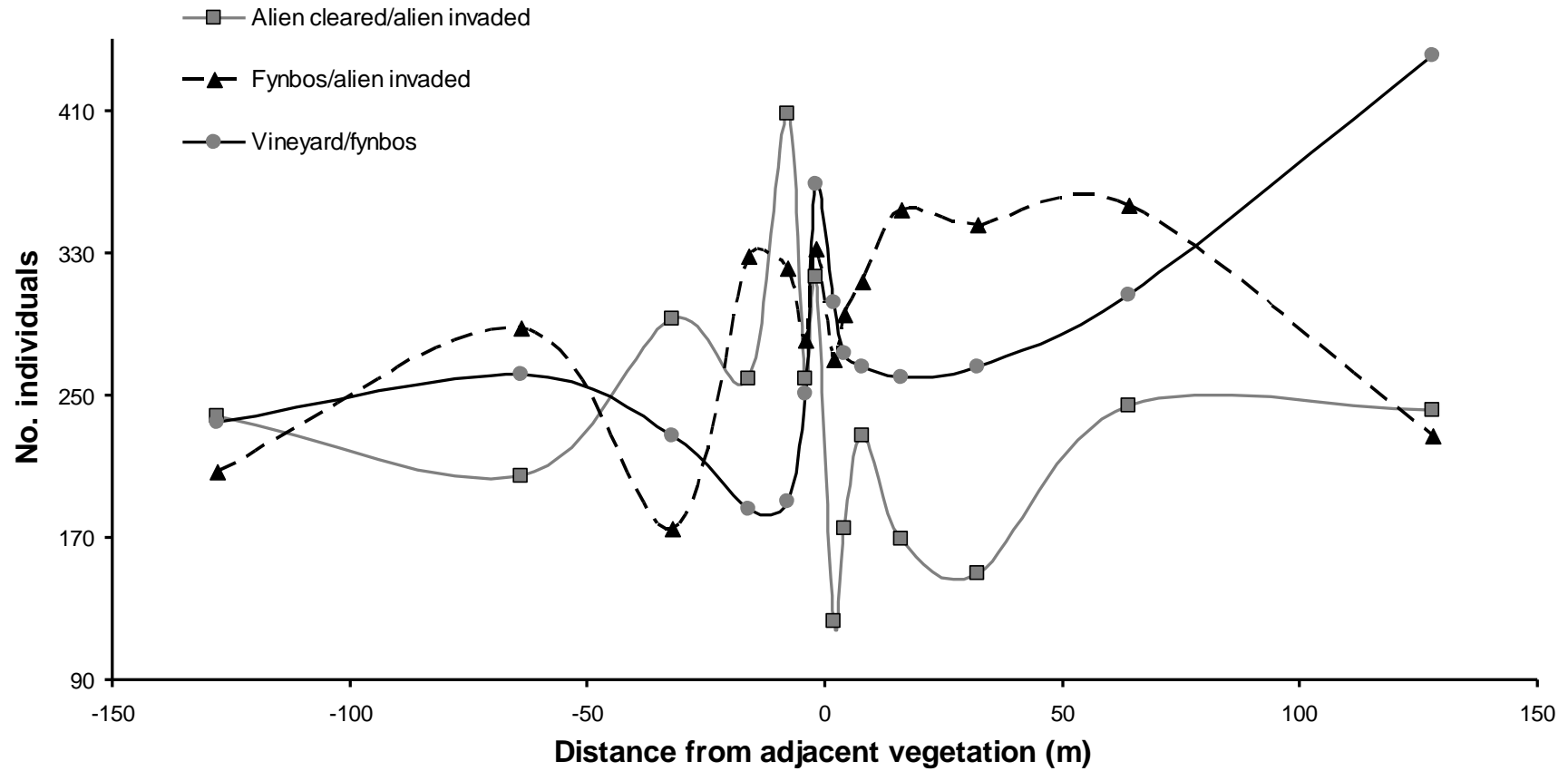


Figure 4.8 Arthropod abundance along alien cleared/alien invaded, fynbos/alien invaded and vineyard/fynbos ecotones. Negative distance (-) represents the first mentioned vegetation type of the pair. For example, alien cleared/fynbos: negative distance represents alien cleared, while positive distance represents alien invaded. 0 m = ecotone.

Taxonomic response to the ecotone

Fynbos/alien invaded ecotone

Arthropod groups responded strongly in terms of varying species richness to this (Figures 4.9-4.10). Generally, all groups showed a trough at the ecotone, although in the case of spiders, there was also a sharp peak (Figure 4.9). Different spider species responded differently to the alien cleared/alien invaded ecotone (Figure 4.11). However, beetle species (Anobiidae: *Xestobium* sp.) was more abundant in alien invaded than in fynbos vegetation type (Figure 4.12). Some fly species were sampled from the fynbos but not in the core of adjacent alien invaded vegetation type. (Figure 4.13).

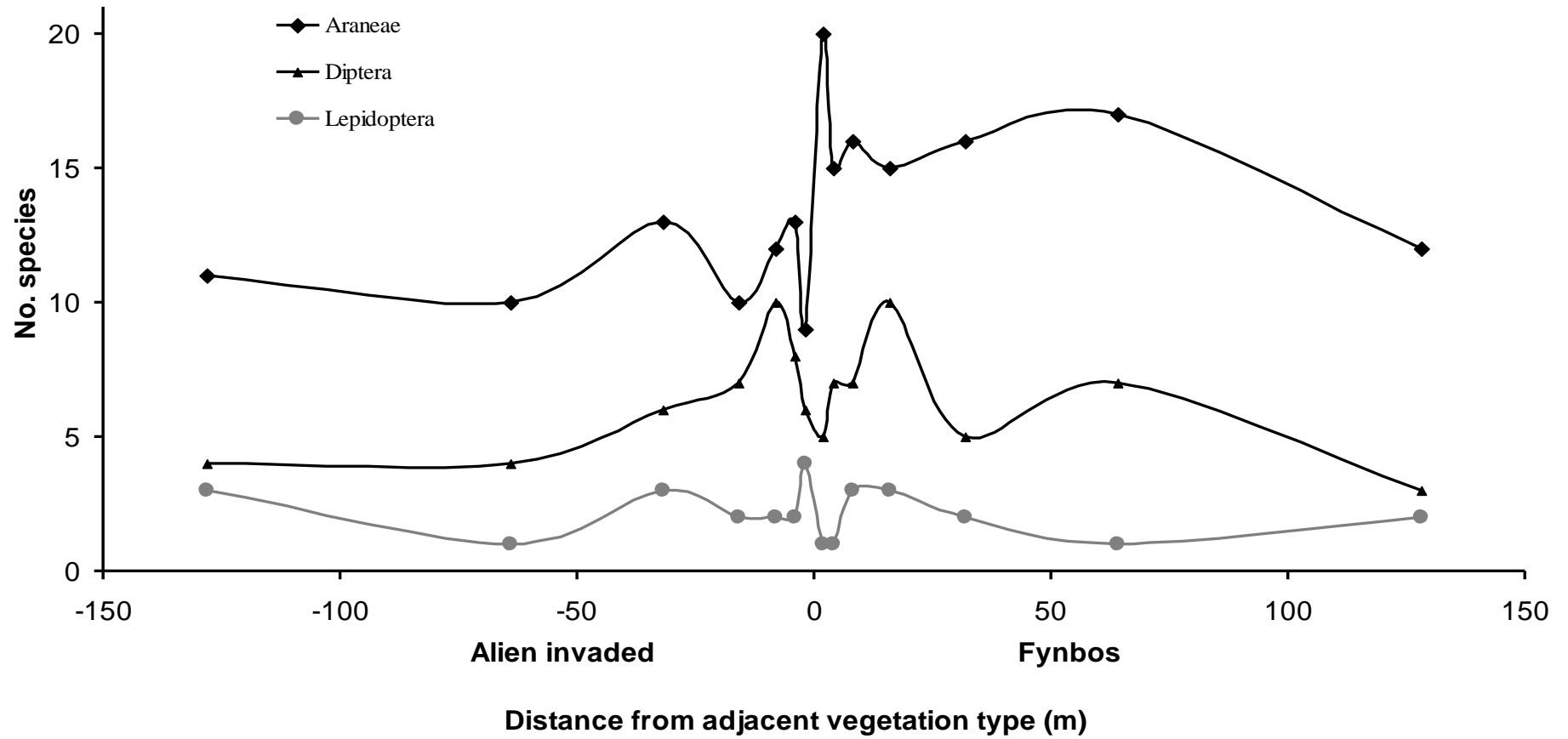


Figure 4.9 Species richness for the dominant taxa: Araneae, Diptera, and Lepidoptera along alien invaded/fynbos ecotone. 0 m = ecotone.

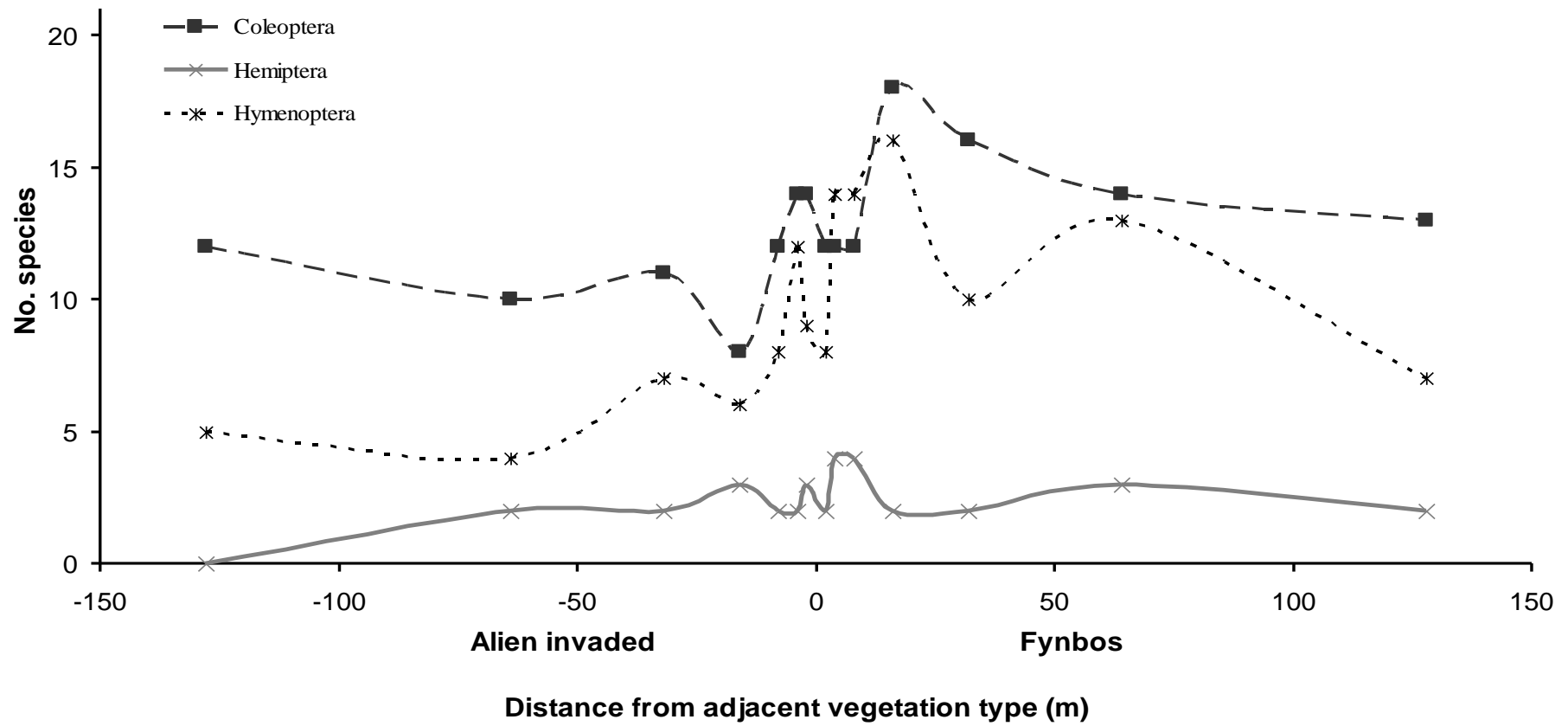


Figure 4.10 Species richness for the dominant taxa: Coleoptera, Hemiptera, and Hymenoptera along alien invaded/fynbos ecotone. 0 m = ecotone.

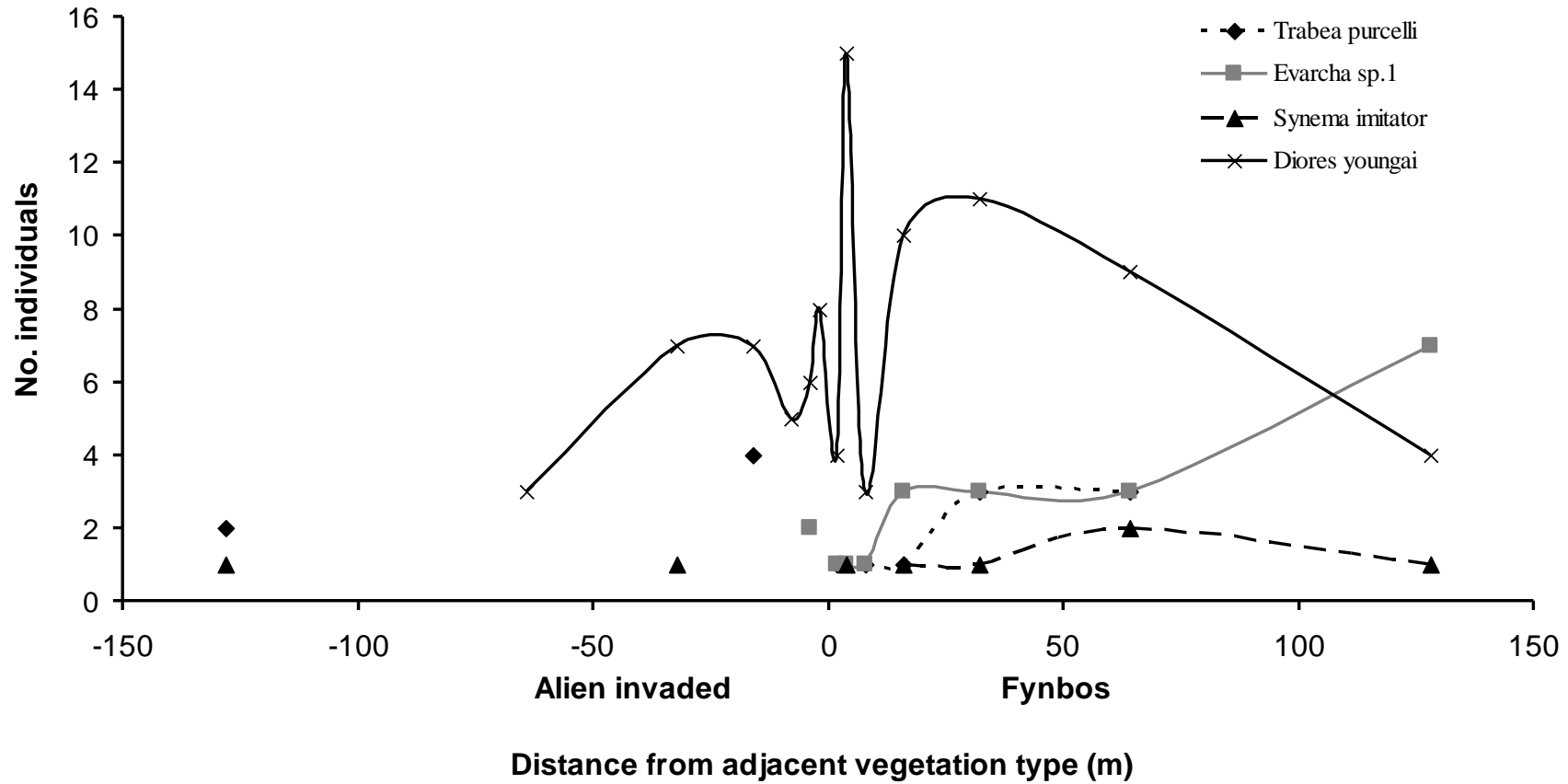


Figure 4.11 Selected spider species (Lycosidae: *Trabea purcelli*; Salticidae: *Evarcha sp.*; Thomisidae: *Synema imitator*; and Zodariidae: *Diores youngai*) abundance along alien invaded/fynbos ecotone. 0 m = ecotone.

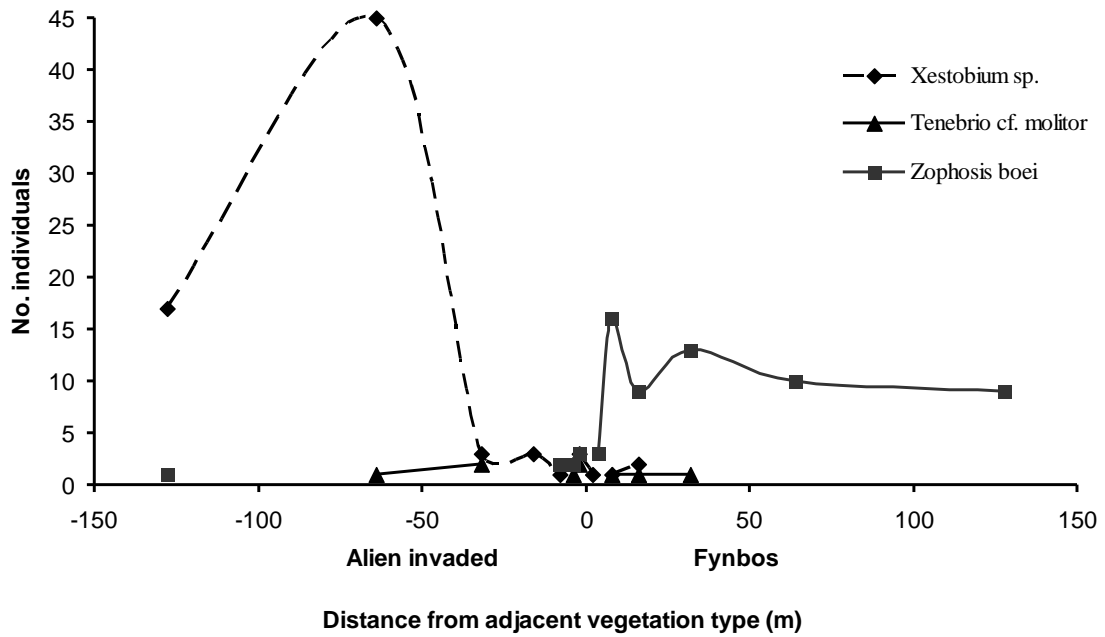


Figure 4.12 Selected beetle species (Anobiidae: *Xestobium* sp., Tenebrionidae: *Tenebrio* cf. *molitor* and *Zophosis boei*) abundance along alien invaded/fynbos ecotone. 0 m = ecotone.

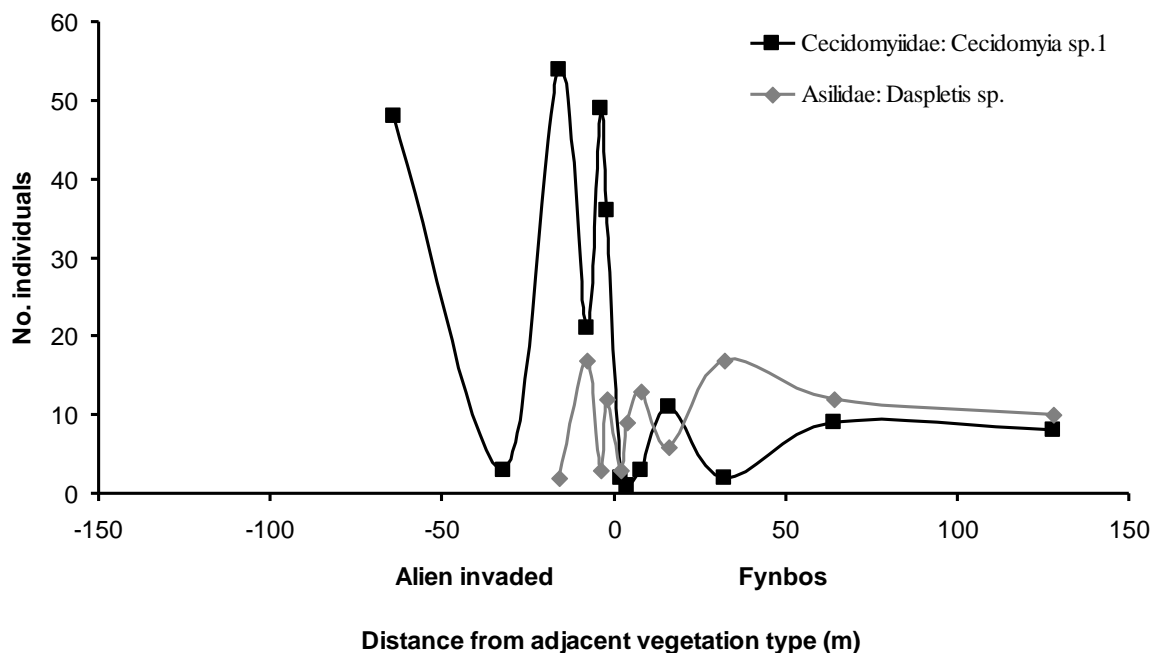


Figure 4.13 Dominant fly species (Cecidomyiidae: *Cecidomyia* sp.1 and Asilidae: *Daspletis* sp.) abundance along alien invaded/fynbos ecotone. 0 m = ecotone.

Alien cleared/fynbos ecotone

There was a remarkably similar pattern of species richness for all major taxa on either side of this ecotone (Figure 4.14-4.17). Although beetles and spiders were generally more species rich at the ecotone, most of the other taxa showed variation in species richness around the 0 m mark, although the bugs dropped at the precise transition zone while spiders increased.

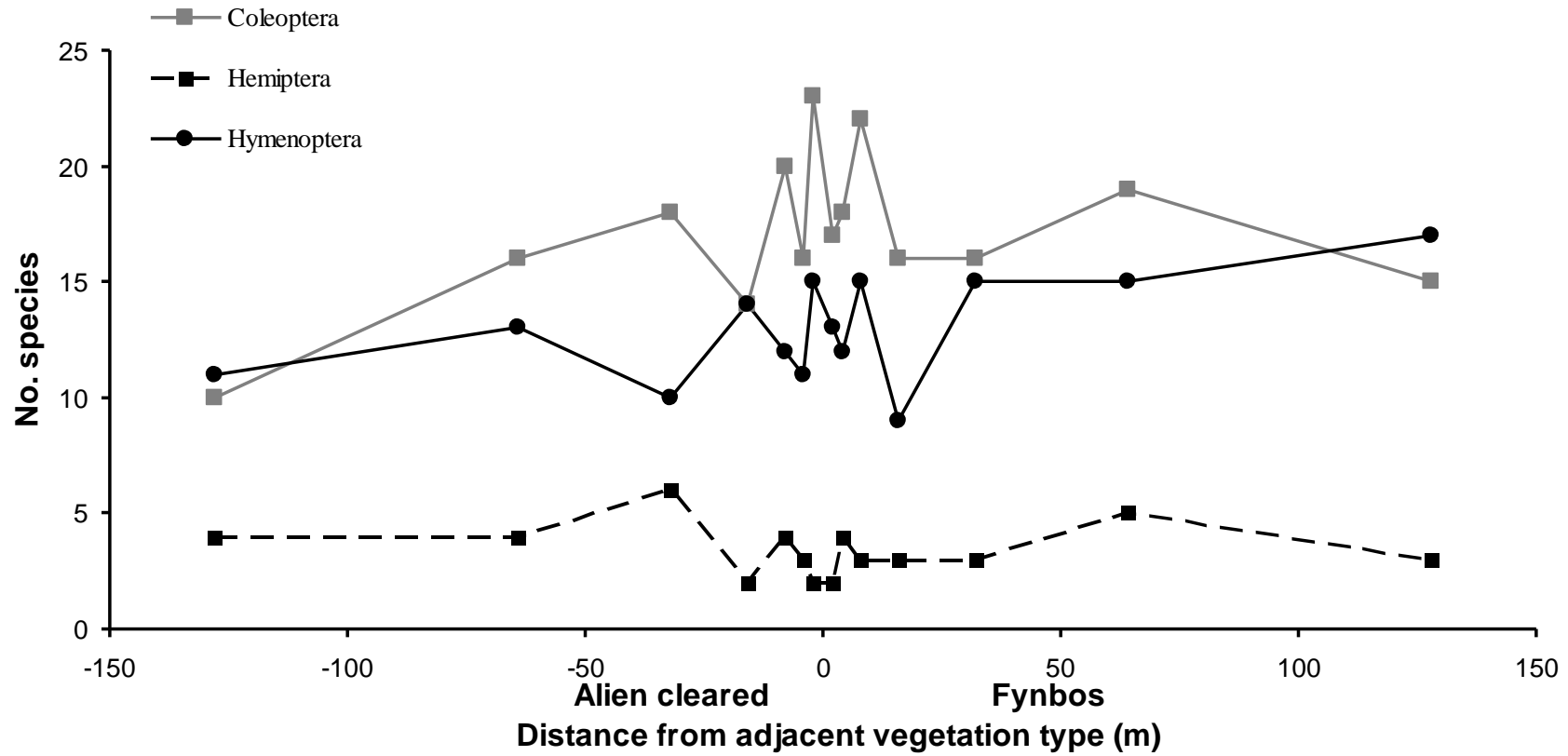


Figure 4.14 Species richness for the dominant taxa: Coleoptera, Hemiptera, and Hymenoptera along alien cleared /fynbos ecotone. 0 m = ecotone.

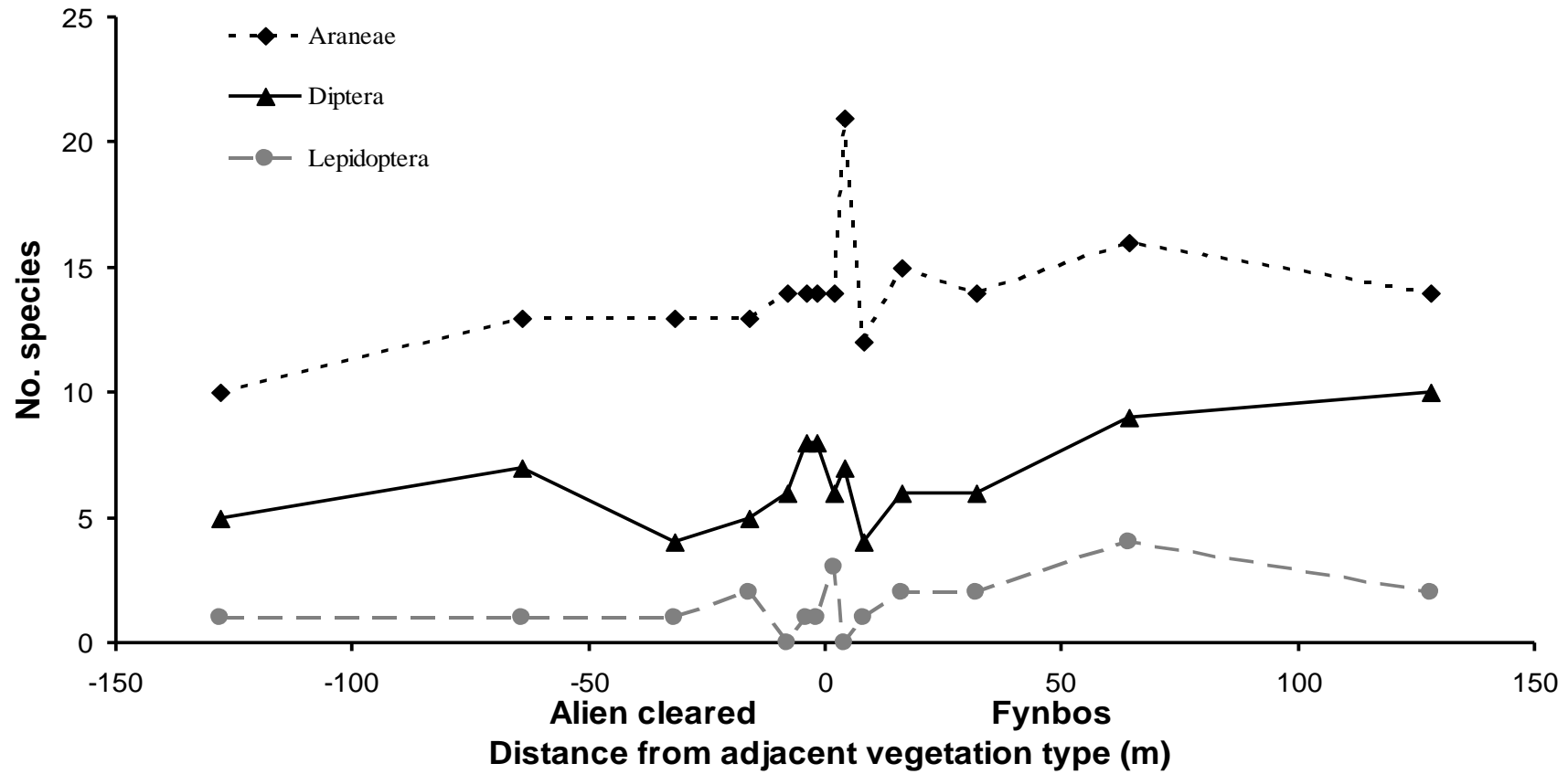


Figure 4.15 Species richness for the dominant taxa: Araneae, Diptera and Lepidoptera along alien cleared/fynbos ecotone. 0 m = ecotone.

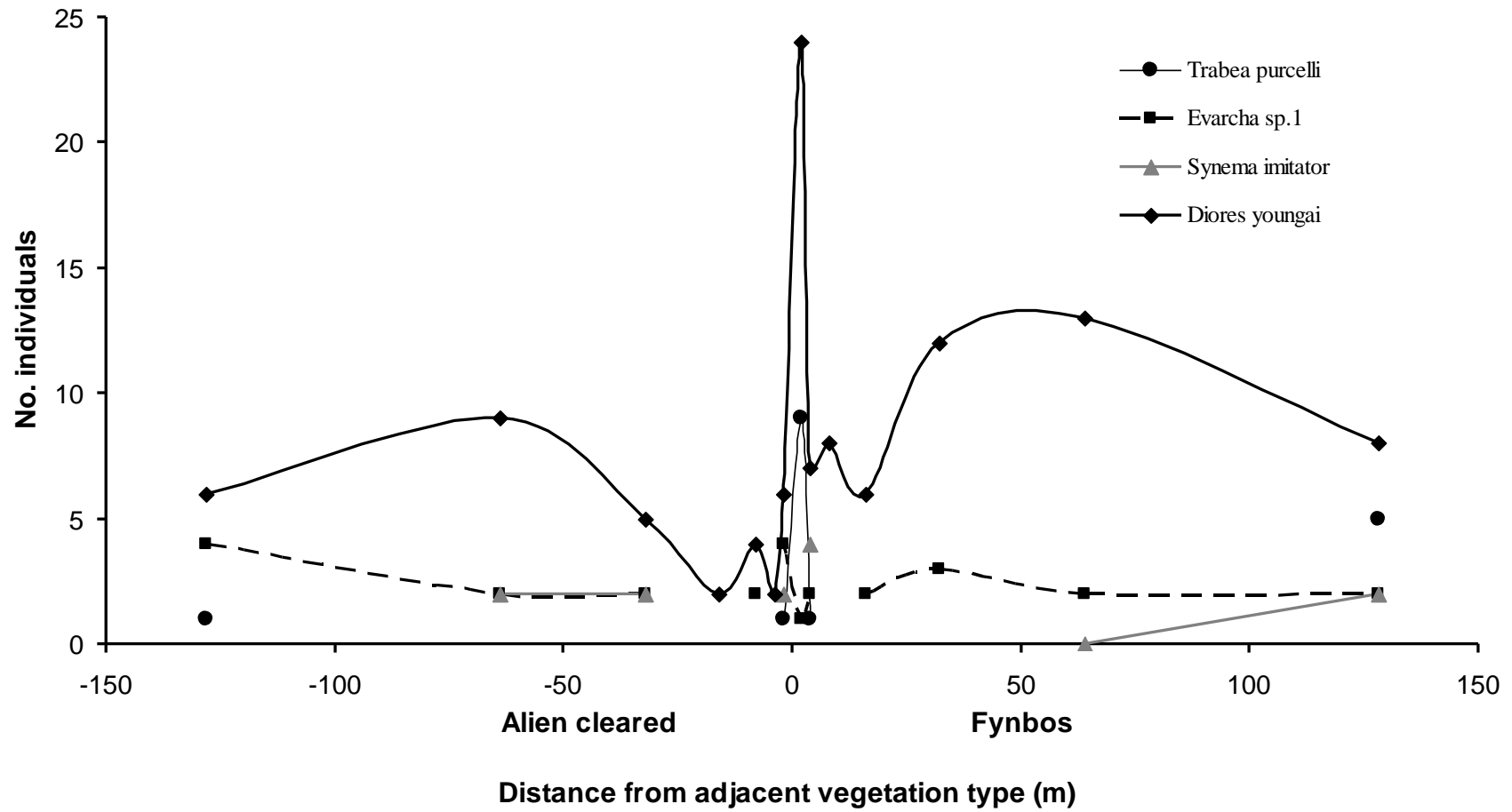


Figure 4.16 Selected spiders (Lycosidae: *Trabea purcelli*; Salticidae: *Evarcha sp.1*; Thomisidae: *Synema imitator*; and Zodariidae: *Diores youngai*) species abundance along alien cleared/fynbos ecotone. 0 m = ecotone.

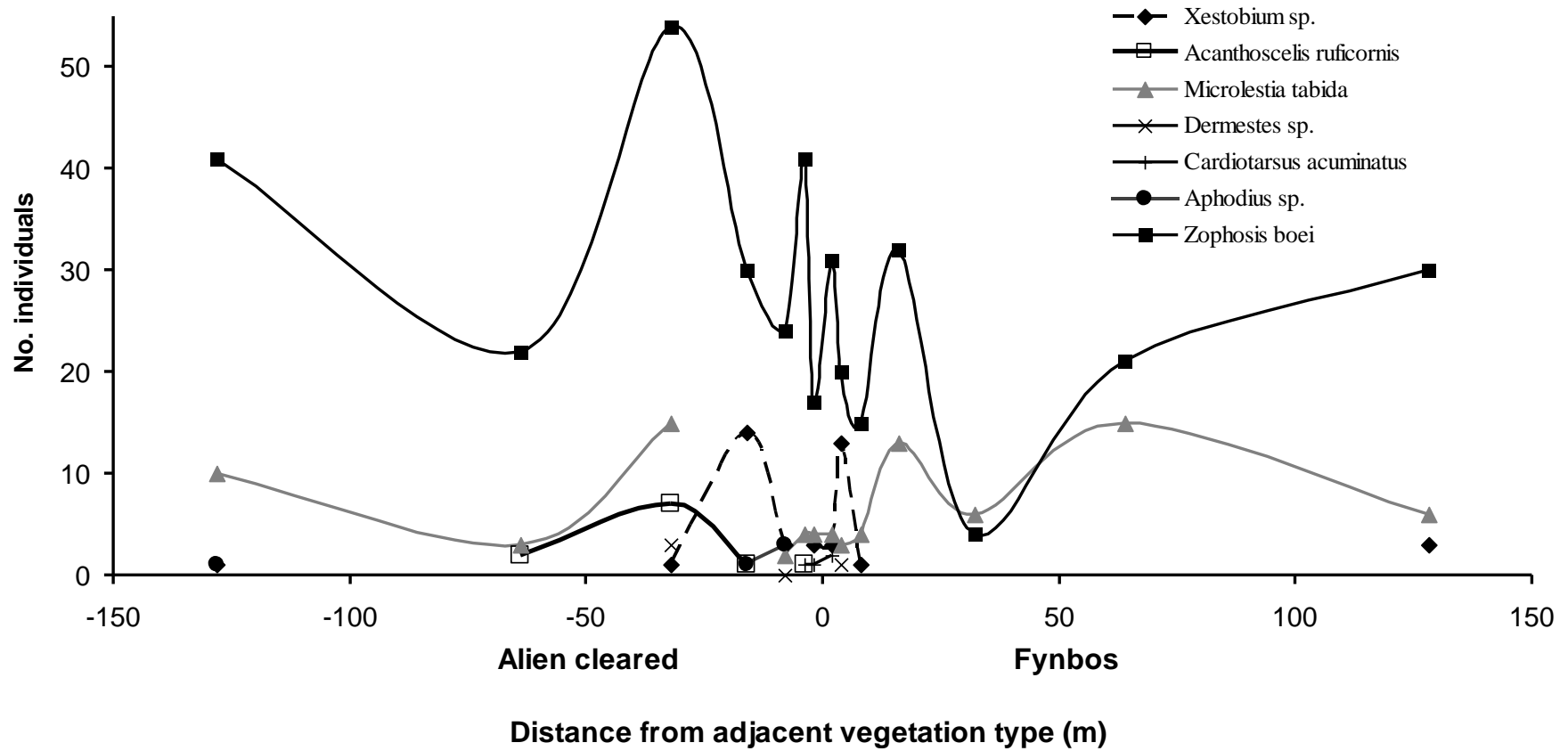


Figure 4.17 Selected beetle species (Anobiidae: *Xestobium* sp., Carabidae: *Acanthoscelis ruficornis* and *Microlestia tabida*, Dermestidae: *Dermestes* sp., Elateridae: *Cardiotarsus acuminatus*, Scarabaeidae: *Aphodius* sp., and Tenebrionidae: *Zophosis boei*) abundance along alien cleared/fynbos ecotone. 0 m represents ecotone.

Alien cleared/alien invaded ecotone

There was overall substantial difference in species richness either side of this ecotone, with the cleared of aliens side much richer in beetles, spiders, hymenopterans and flies (Figure 4.18-4.19). However, other spider species such as *Synema imitator* were rarely sampled away from the ecotone (Figure 4.20). Bugs and lepidopterans were similar on either side of the ecotone. For all taxa, the boundary had a remarkably strong effect on species richness with sharp changes in species richness even over small distances from the 0 m mark. Interestingly, most taxa showed a trough within 30 m on either side of the boundary.

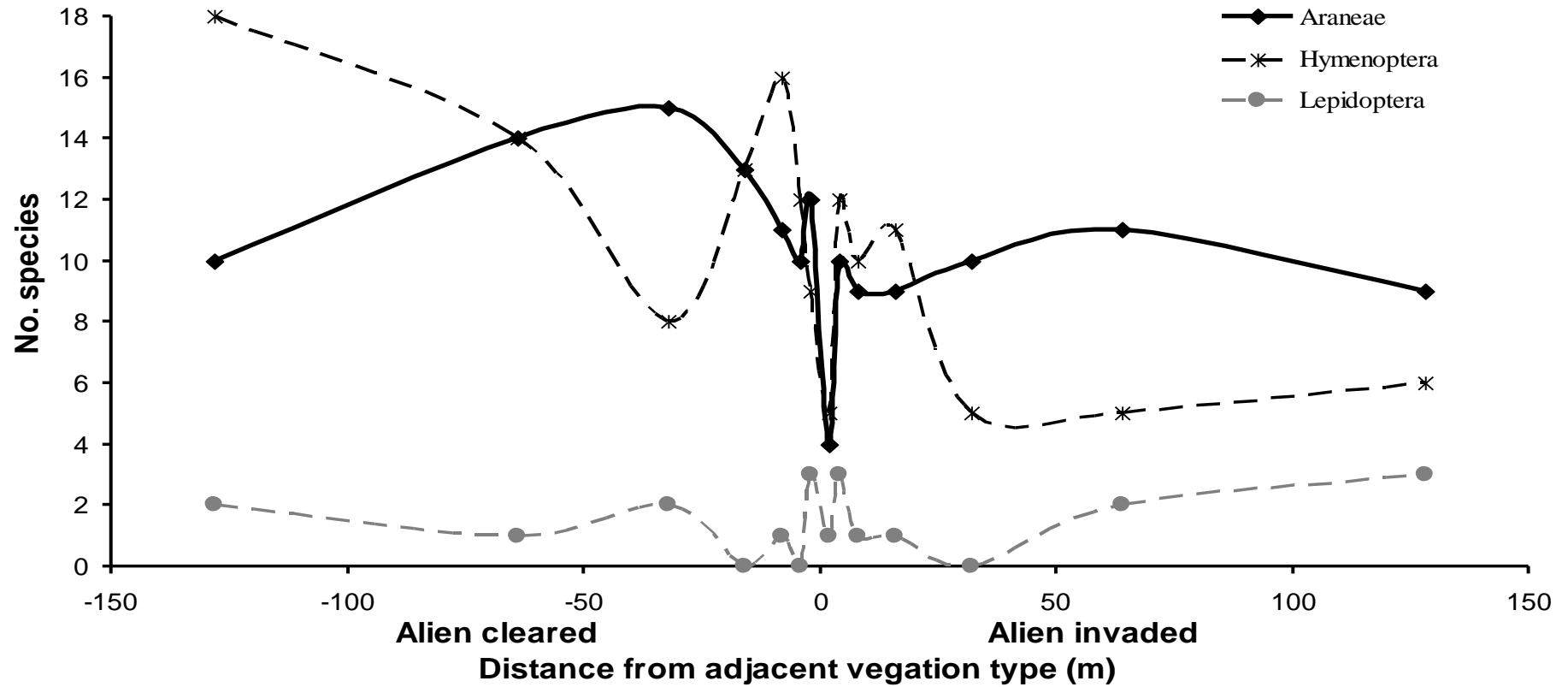


Figure 4.18 Species richness for the dominant taxa: Araneae, Hymenoptera, and Lepidoptera along alien cleared/invaded ecotone. 0 m represents ecotone.

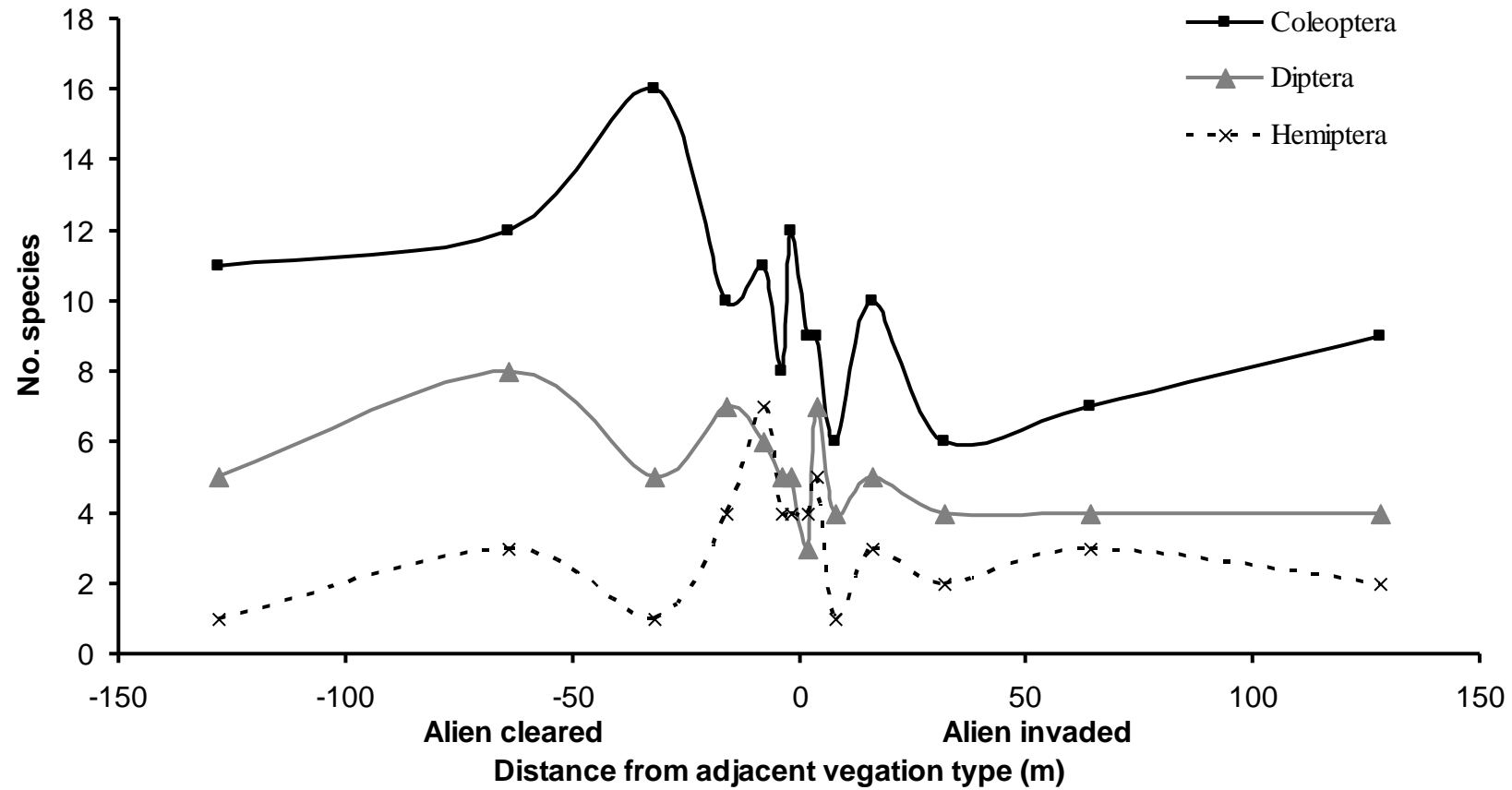


Figure 4.19 Species richness for the dominant taxa: Coleoptera, Diptera and Hemiptera along alien cleared/invaded ecotone. 0 m represents ecotone.

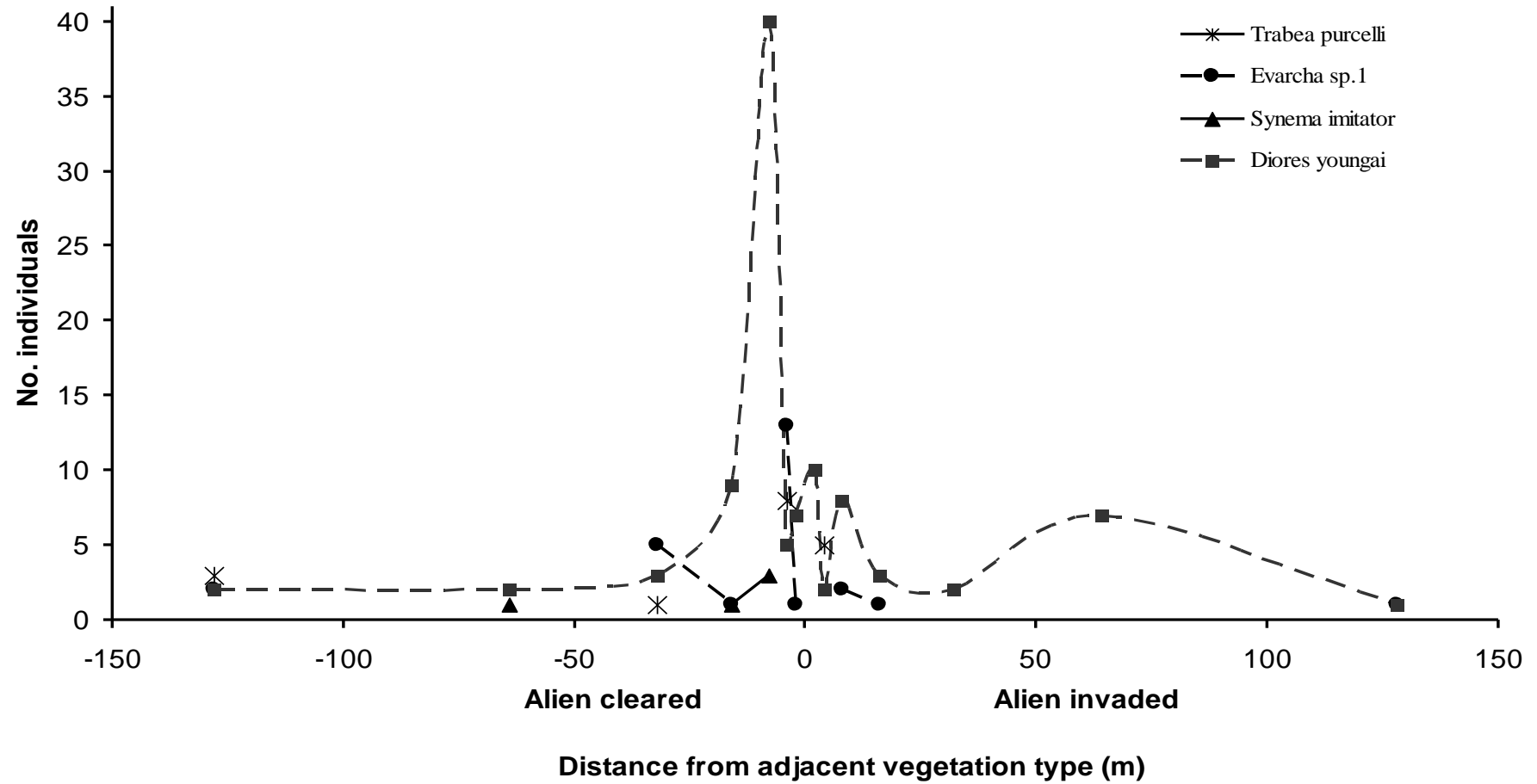


Figure 4.20 Selected spider species (Lycosidae: *Trabea purcelli*; Salticidae: *Evarcha* sp.1; Thomisidae: *Synema imitator* and Zodariidae: *Diores youngai*) abundance along alien cleared/alien invaded ecotone. 0 m represents ecotone.

Vineyard/fynbos ecotone

Different arthropod taxa responded differently towards vineyard/fynbos ecotone (Figure 4.21-4.22). Coleoptera, Araneae and Hymenoptera were the dominant taxa (Figure 4.21-4.22). However, these groups tended to decrease in species richness from ecotone into the core of adjacent vineyard, but increase from ecotone into the core of adjacent fynbos vegetation (Figure 4.21-4.22). Nevertheless, Lepidoptera species richness was high in the vineyard compared to the adjacent fynbos vegetation (Figure 4.22). Different species responded differently to vineyard/fynbos ecotone, even those belonging to the same taxa (Figure 4.23-Figure 4.24).

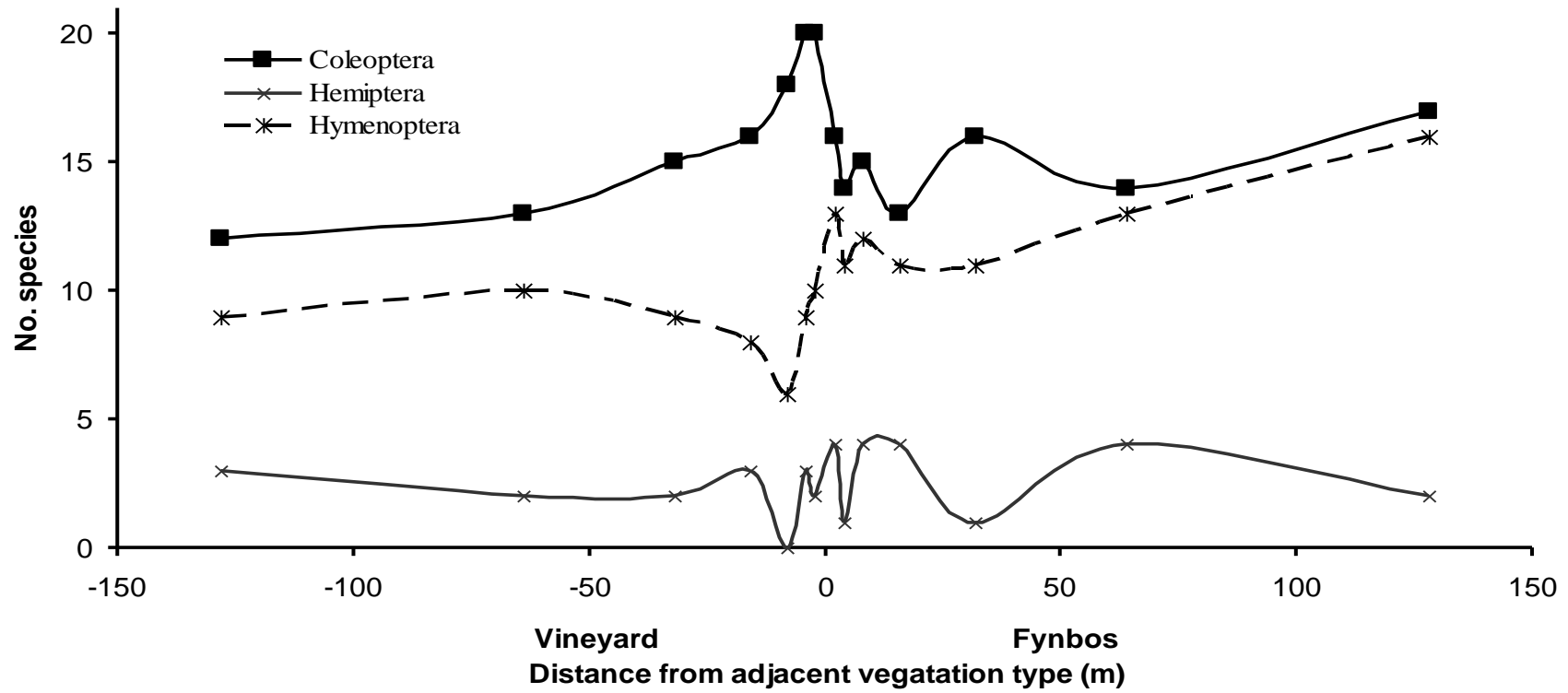


Figure 4.21 Species richness for the dominant taxa: Coleoptera, Hemiptera and Hymenoptera along vineyard/fynbos ecotone. 0 m represents ecotone.

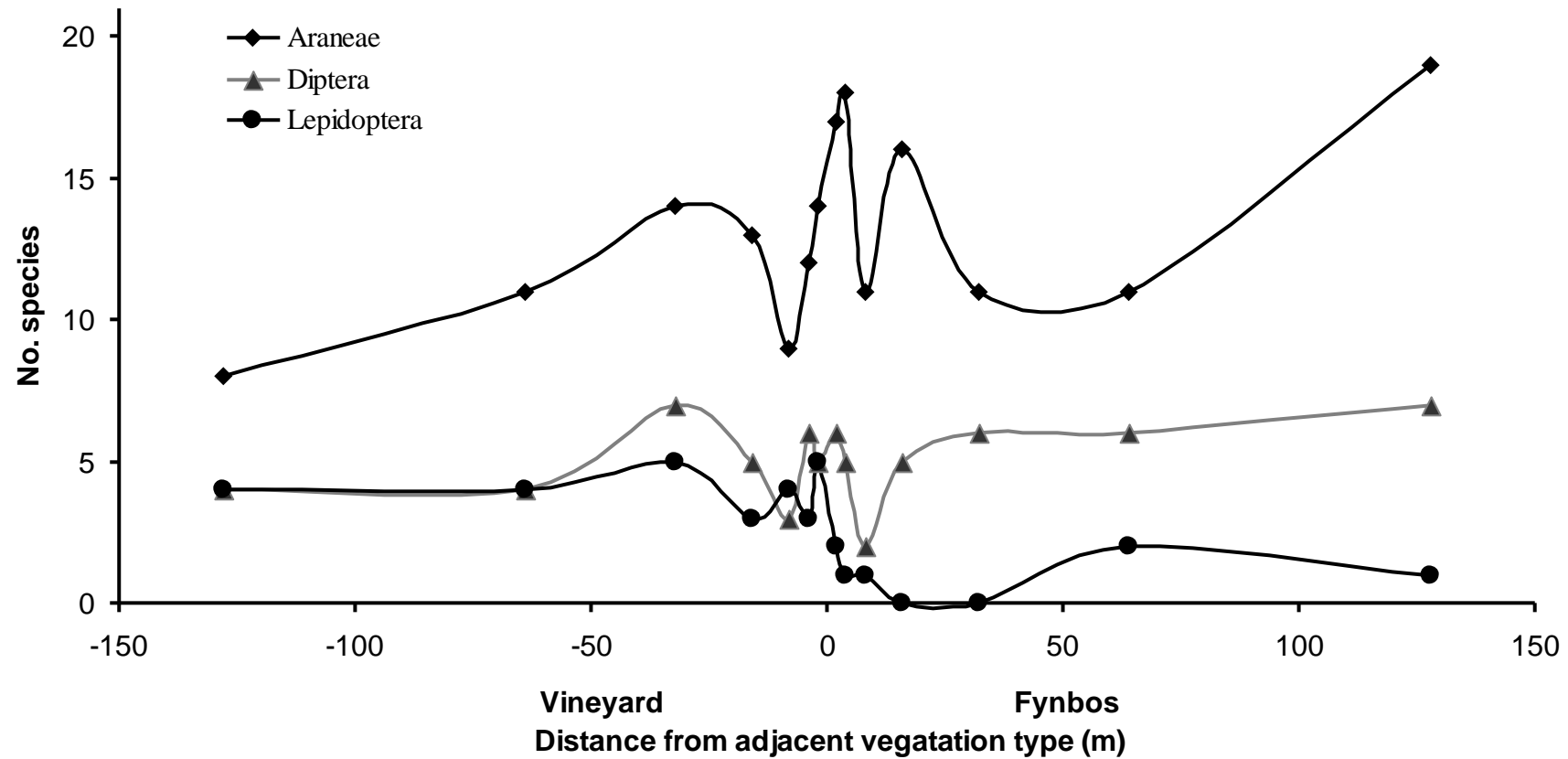


Figure 4.22 Species richness for the dominant taxa: Araneae, Diptera, and Lepidoptera along vineyard/fynbos ecotone. 0 m represents ecotone.

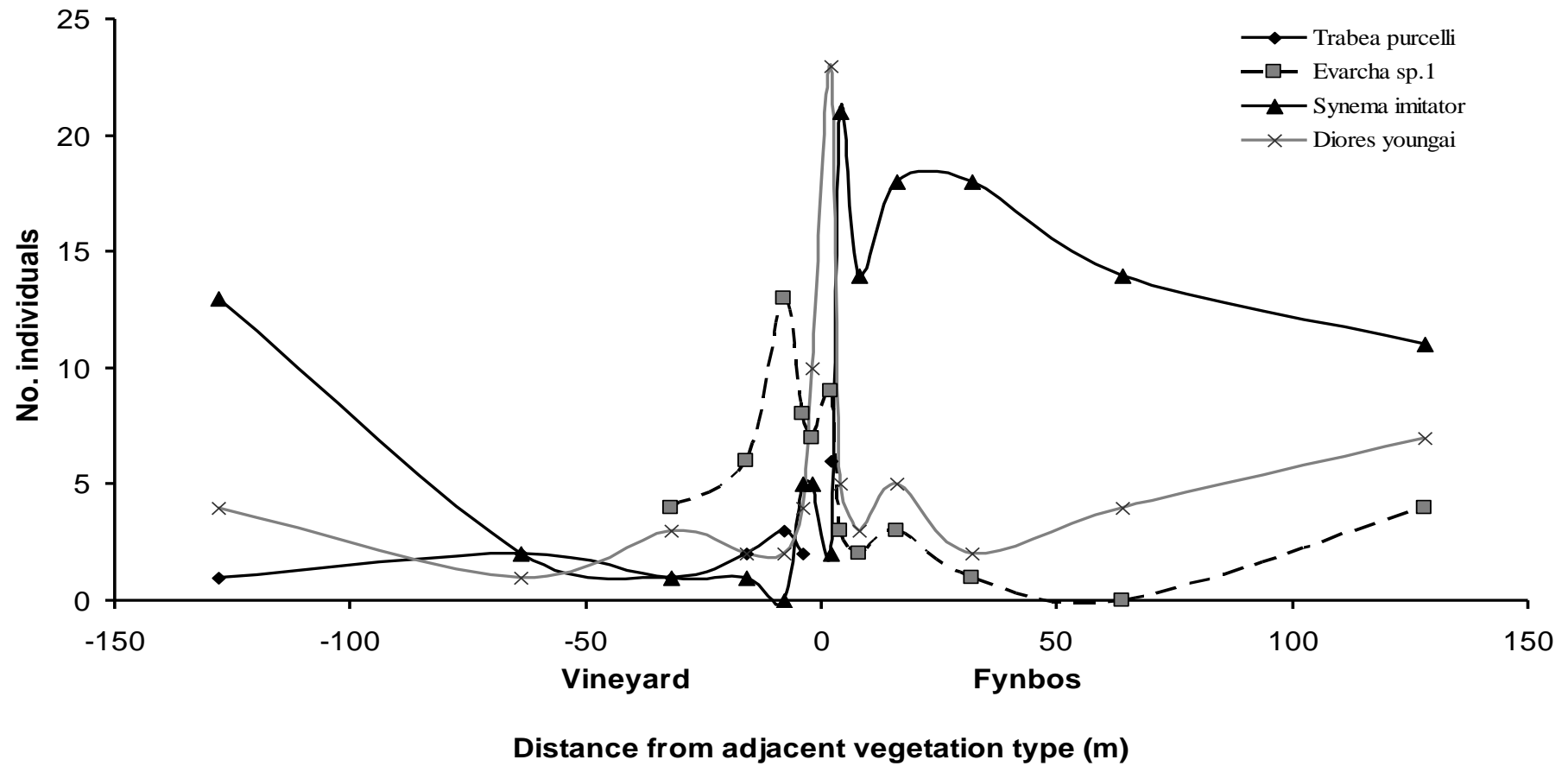


Figure 4.23 Selected spider species (Lycosidae: *Trabea purcelli*; Salticidae: *Evarcha sp.1*; Thomisidae: *Synema imitator*; and Zodariidae: *Diores youngai*) abundance along vineyard/fynbos ecotone. 0 m represents ecotone.

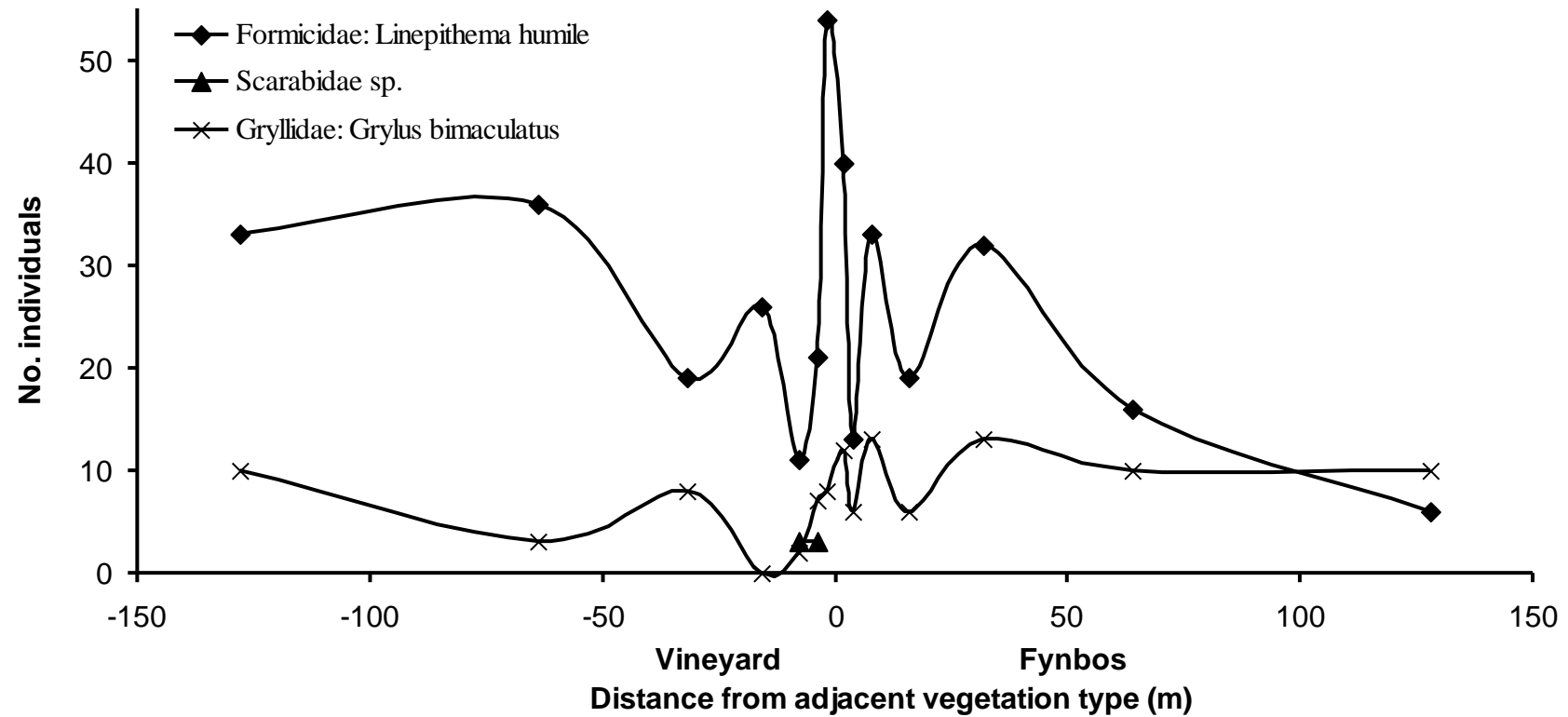


Figure 4.24 Selected arthropod species (Formicidae: *Linepithema humile*, Scarabidae sp., and Gryllidae: *Gryllus bimaculatus*) abundance along the vineyard/fynbos ecotone. 0 m represents ecotone.

Vineyard/alien invaded ecotone

Generally, different arthropod taxa showed similar patterns towards vineyard/alien invaded ecotone (Figure 4.25-4.26). Almost all arthropod taxa had higher species richness within the vineyard/alien invaded ecotone zone than in adjacent vegetation (Figure 4.25-4.26). Dominant taxa were Araneae, Coleoptera, and Hymenoptera on either side of the ecotone. Spider species had a “jump” of restricted distribution along vineyard/alien invaded ecotone (Figure 4.27). Coleopteran species (Anobiidae: *Xestobium* sp.) had dominated vineyard/alien invaded ecotone (Figure 4.28).

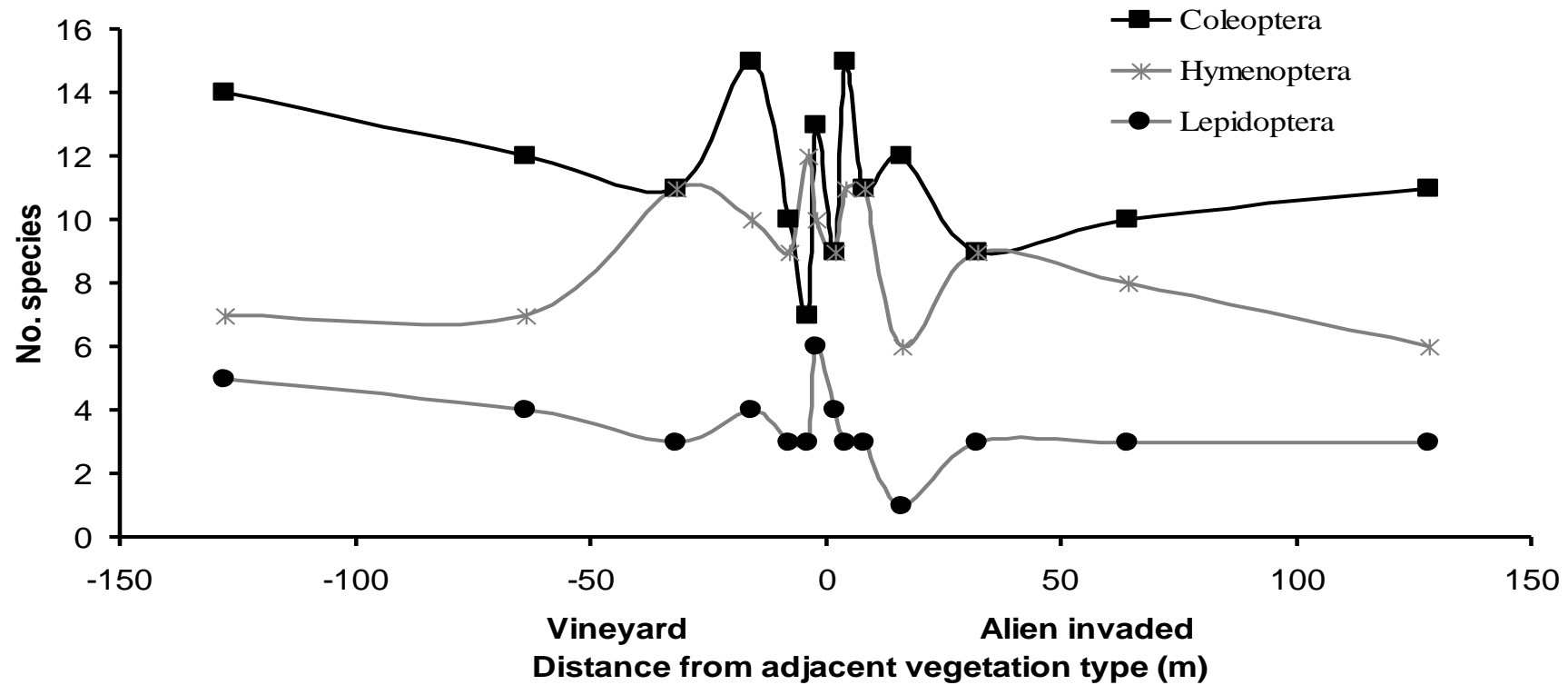


Figure 4.25 Species richness for the dominant taxa: Coleoptera, Hymenoptera, and Lepidoptera along vineyard/alien invaded ecotone. 0 m represents ecotone.

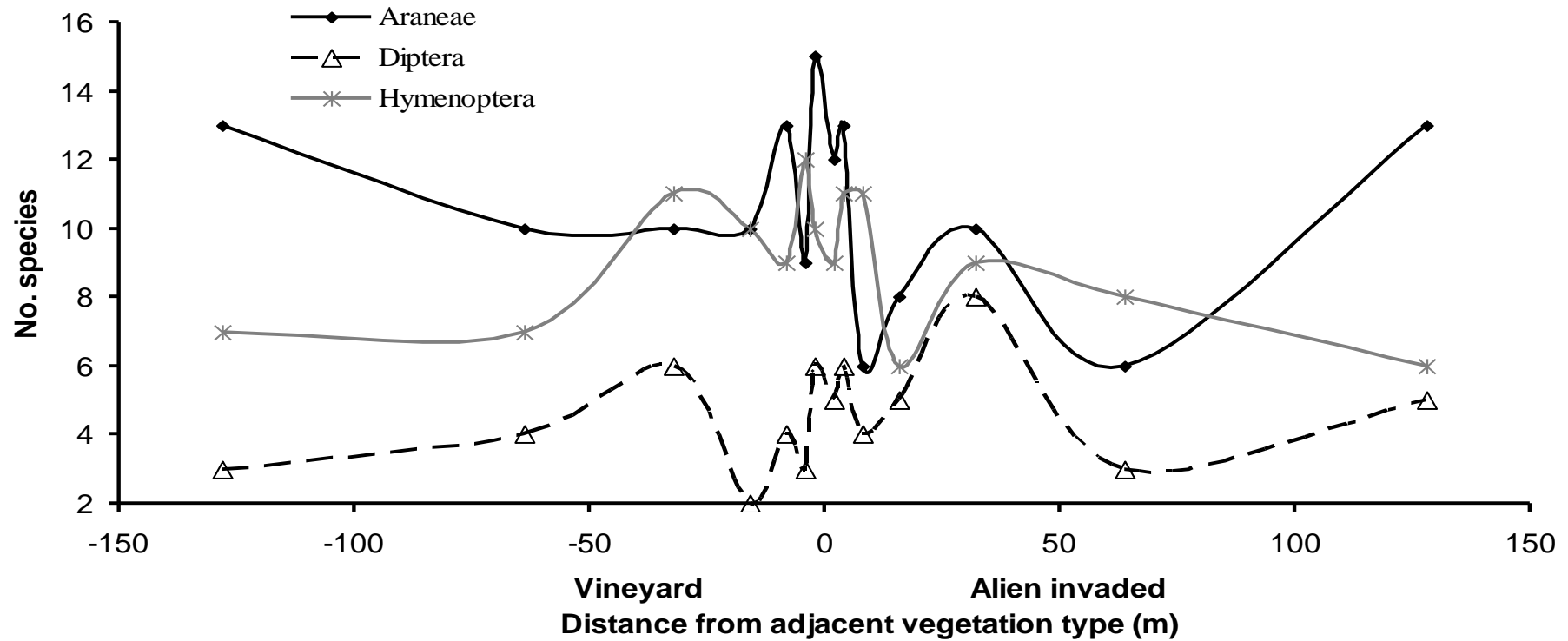


Figure 4.26 Species richness for the dominant taxa: Araneae, Diptera and Hymenoptera along vineyard/alien invaded ecotone. 0 m represents ecotone.

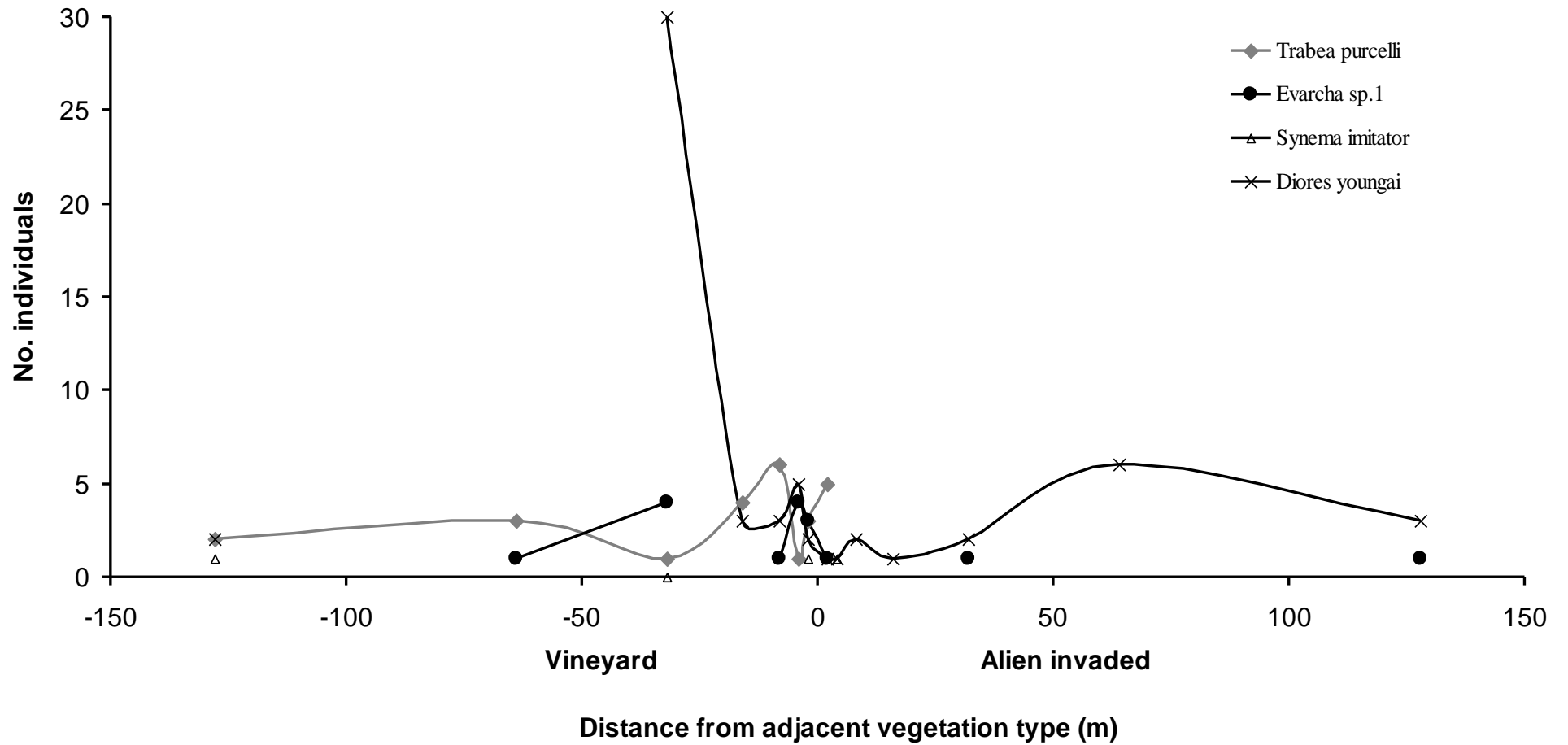


Figure 4.27 Selected spider species (Lycosidae: *Trabea purcelli*; Salticidae: *Evarcha* sp.1; Thomisidae: *Synema imitator*; and Zodariidae: *Diores youngai*) abundance along vineyard/alien invaded ecotone. 0 m represents ecotone.

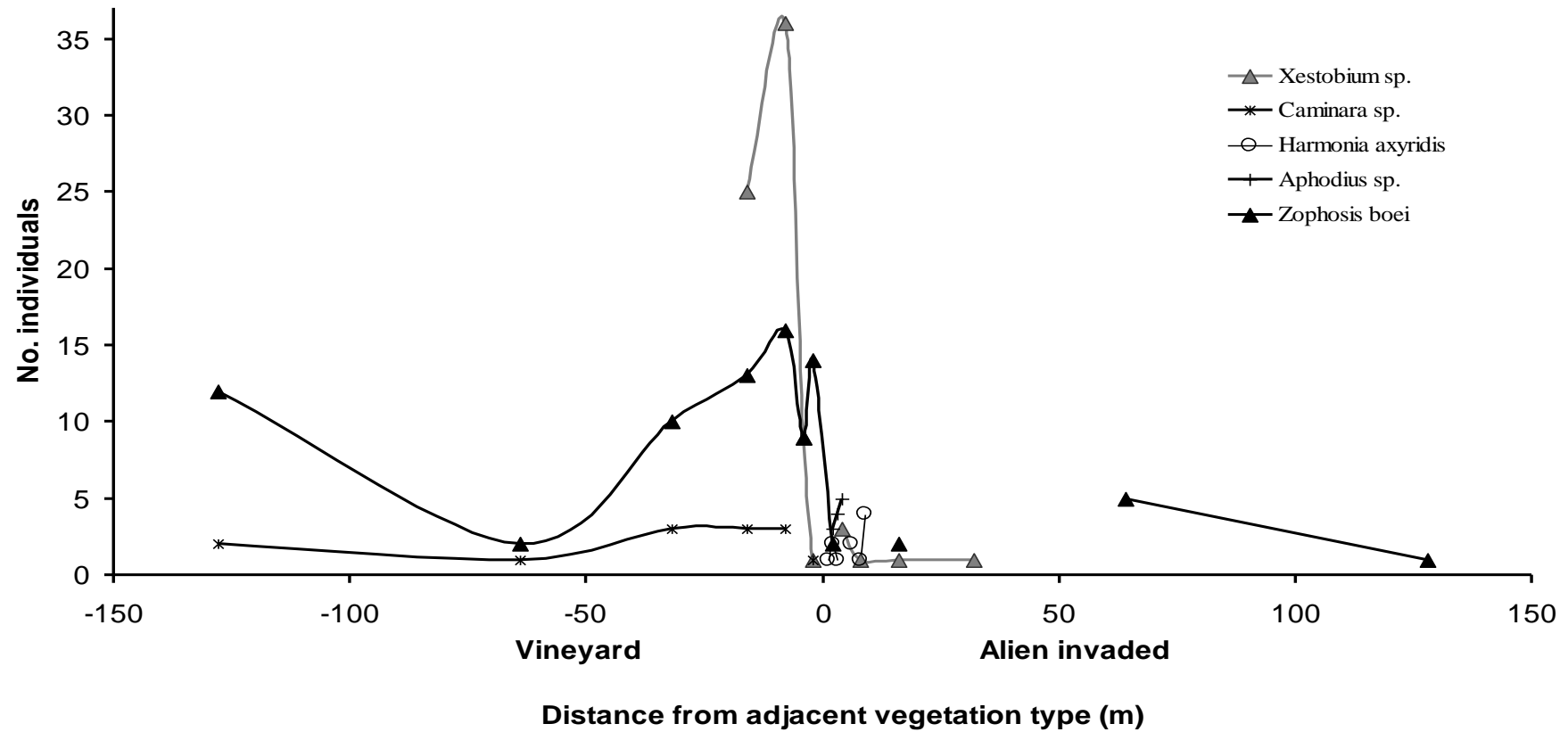


Figure 4.28 Selected coleopteran species (Anobiidae: *Xestobium* spp., Carabidae: *Caminara* spp., Cocinellidae: *Harmonia axyridis*, Scarabaeidae: *Aphodius* spp. and Tenebrionidae: *Zophosis boei*) abundance along vineyard/alien invaded ecotone. 0 m represents ecotone.

Alien cleared/vineyard ecotone

Generally, different arthropod groups had similar response patterns towards alien cleared/vineyard (AC/VY) ecotone (Figure 4.29-4.30). In fact, similar responses to that in alien cleared/fynbos ecotone (Figure 4.14) were observed. The dominant taxa: Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, and Lepidoptera did not show clear preference to particular adjacent vegetation. However, higher species richness was within the ecotone for almost all the taxa (Figure 4.29-4.30).

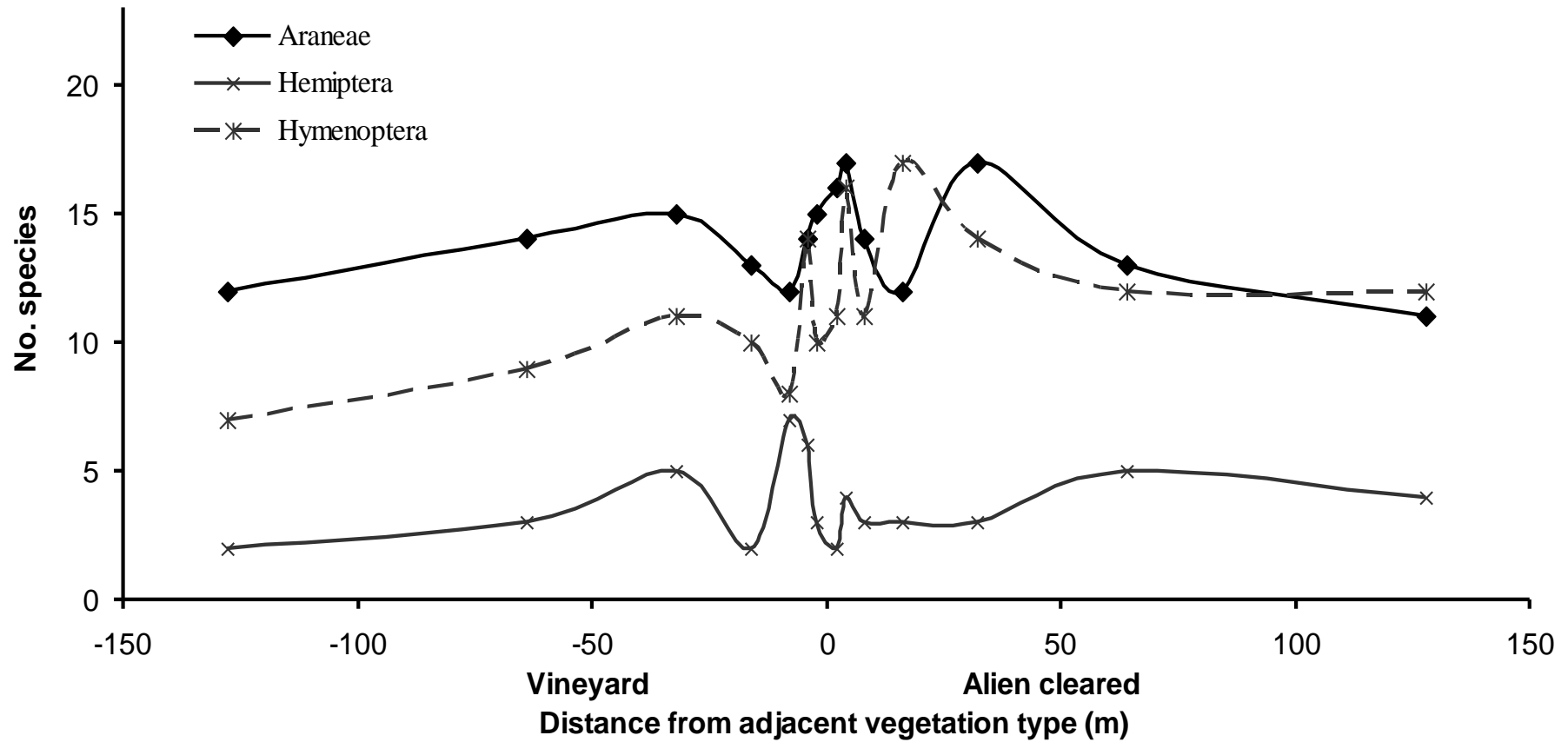


Figure 4.29 Species richness for the dominant taxa: Araneae, Hemiptera, and Hymenoptera along alien cleared/vineyard ecotone. 0 m represents ecotone.

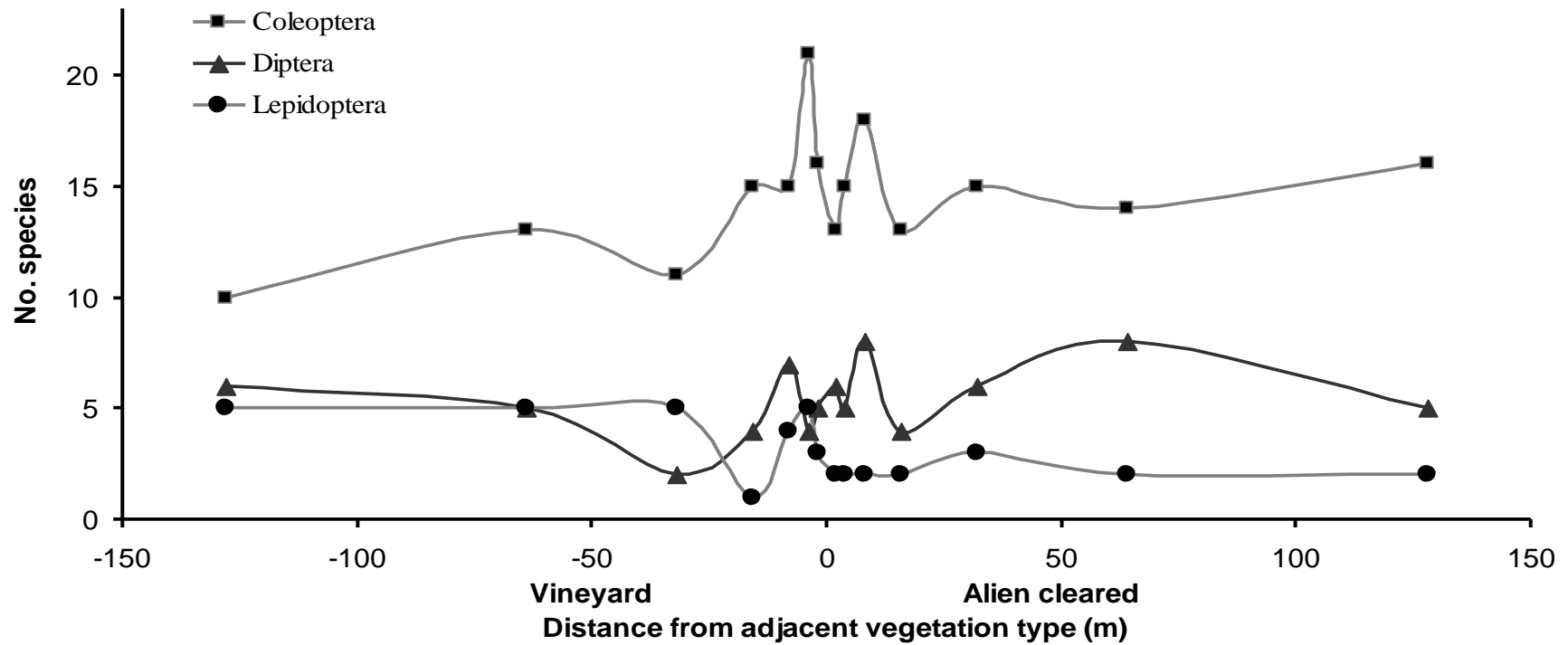


Figure 4.30 Species richness for the dominant taxa: Coleoptera, Diptera and Lepidoptera along alien cleared/vineyard ecotone. 0 m represents ecotone.

Correlation between habitat edge's distance and arthropods

Overall, there was no significant correlation between increasing distance from the ecotone in terms of species richness (Spearman's Coefficient = 0.05; $p = 0.919$, $n = 503$), as well as abundance (Spearman's Coefficient = 0.061; $p = 0.171$, $n = 503$). Nevertheless, individual arthropod species abundance (e.g. *Linepithema humile*) was significantly correlated (Spearman's Coefficient = 0.696; $p = 0.000$, $n = 42$), with the distance from the vineyard edge into the core on natural fynbos (Figure 4.24). Some species had continuous distributions along the ecotones (Figure 4.24) while others were restricted to one side of the ecotone (Figure 4.28).

DISCUSSION

The overall species diversity

As none of the individual ecotone species accumulation curves reached an asymptote, the species estimates are thus considered underestimates. This implies that none of the sampled ecotones alone was good enough to support all the expected CFR surface-active arthropod species. Nevertheless, the overall observed and estimated species accumulation curves were highly comparable, indicating that the sample size (over 24 000 individuals) was sufficient to provide an estimate of surface-active arthropod species richness.

The vineyard/alien invaded ecotone supported lower species richness, despite having the highest estimated richness than any other ecotone type. This clearly indicated that such an ecotone type is not inhabitable by the majority of local arthropods. Only 70% of the estimated species by Jackknife2 were encountered in vineyard/alien invaded ecotone. Generally, vineyards had higher species richness than alien invaded vegetation. The cumulative effect of invasive alien trees (IATs) was evident in the natural fynbos/alien invaded ecotone, where lowest species accumulation was obtained. In fact, all species estimators suggested lowest species richness for the natural fynbos/alien invaded ecotone. Nevertheless, this was not the case when IATs neighbouring natural fynbos were cleared. Both species richness and abundance improved significantly. About 85% of the estimated species using by Jackknife2 species estimator were encountered. This means that there was at least 15% increase in

species richness once IATs were removed. These implies that the clearing of IATs is not only beneficial to aquatic invertebrates (Magoba and Samways, 2010), but to terrestrial arthropod diversity as well. The vineyard/alien invaded ecotone had high species richness, but it was even greater once IATs were cleared. The removal of IATs adjacent to the vineyard not only resulted in higher species richness but also improved abundance as well. As a result, the highest species accumulation curve was for this ecotone type. This indicates that the effect of conversion of natural fynbos for agricultural activities on arthropod species richness is significantly smaller than that caused by invasive alien trees. Also, vineyards do not automatically result in very low arthropod diversity compared to IATs.

Arthropod responses to ecotone

Generally, there were species richness and abundance changes in response to different ecotones. However, different arthropod taxa responded differently, although the majority had their peak at the ecotone, indicating a positive response towards ecotones. Some arthropod species may have benefited from the microclimatic edge effects of the ecotone, and thus their densities were generally high in these areas. However, when these species penetrate an adjacent vegetation fragments, they may compete with arthropod species that depend on landscape element interior conditions. Therefore, interior species may be harmed through the ecological processes of predation, competition and parasitism (Forman, 1995).

Alien cleared/fynbos ecotone had by far the highest species richness and abundance, while alien cleared/alien invaded and vineyard/alien invaded ecotones, had the lowest. As expected, fynbos was associated with higher species richness, while alien invaded vegetation and vineyards were associated with lower species richness. Vineyard/alien invaded ecotone supported significantly lower species diversity. However, both species richness and abundance increased significantly when IATs neighbouring vineyards were removed. In any ecotone type that involved fynbos, fynbos sites always had the highest species richness in comparison with adjacent landscape types. In contrast, alien invaded vegetation always had a lower species richness compared to the adjacent landscape types. The exception was the vineyard/alien invaded ecotone, where arthropods were more equally distributed on either side of this ecotone. Overall, various groups of surface-active arthropods studied here confirmed the edge-effect

hypothesis (Leopold, 1933), where diversity and species abundance is higher in ecotones than in adjacent areas.

Fynbos/alien invaded ecotone

Different arthropod taxa responded differently to the fynbos/alien invaded ecotone, with similar responses to that in the alien cleared/alien invaded ecotone. As expected, fynbos supported higher species richness than the adjacent alien invaded vegetation for the majority of arthropod taxa. This was evident for spiders, beetles and bug species. The spider, *D. youngai* was particularly abundant in various ecotones. In contrast, spiders such as *T. purcelli*; *Evarcha* sp.; and *S. imitator*, were rarely more than a few meters away from the ecotone into the core of alien invaded vegetation. It is possible that these species were sampled by chance within alien invaded vegetation where they had discontinuous distributions, as opposed to continuous distribution in the adjacent natural fynbos.

The rarity of many species makes it difficult to conclude that any one site is important for their conservation. Nevertheless, the beetle species *Xestobium* sp. was common in alien invaded vegetation, and more so even than in fynbos, indicating that alien invaded vegetation can be important for the benefit of some arthropod species. Even the beetle *Tenebrio* cf. *molitor*, was sampled only within 50 m of the ecotone. This implies that alien cleared/alien invaded ecotone provides better living conditions for some arthropod species than the adjacent vegetation. Alternatively, arthropods may be more active at the ecotone.

Alien cleared/fynbos ecotone

Different arthropod taxa responded positively to the alien cleared/fynbos ecotone. More species of spider, beetle, and hymenopterans preferred the ecotone, although they were equally distributed on either side of it. Fly, lepidopteran and bug species, also had continuous distribution along the ecotone. Interestingly, there was a “mirror image” pattern of species richness on either side of the ecotone. This implies that clearing of IATs was beneficial to many arthropods, with species richness being comparable to that of fynbos. However, at the species level, species belonging to the same group responded differently to similar ecotone. These findings are in agreement with Sisk and Haddad (2002) that the edge impacts are not restricted to the population and community levels, but influence all levels of ecological organization. For example, *D. youngai* was

more common in the ecotone than were some spider species. Dwernychuk and Boag (1972), described such a zone of low quality habitat that organisms prefer over superior habitats as an ecological trap. However, another spider species *T. purcelli* was rarely sampled at any distance away from the ecotone. This was also the case for other beetle species. These findings are in contrast with the general assumption that edge effects are similar for all related species (Sisk and Haddad, 2002). However, these results support Gaston (1991) that different species will react differently to any particular conditions.

Alien cleared/alien invaded ecotone

The majority of taxa were strongly represented within the ecotone, and in the alien cleared vegetation compared to adjacent alien invaded vegetation. This implies that more arthropod species would prefer to stay within the ecotone than in the core of alien vegetation. It is important to note that the responses of arthropods are not mirror images, indicating differences in response between adjacent distinct vegetation fragments. Different spider species responded differently to alien cleared/alien invaded ecotone. Although *D. youngai* was common in ecotone, this was not the case for the other spiders *T. purcelli*; *Evarcha* sp.; and *S. imitator* which were rare even a few meters away from the ecotone. Indeed, there are no general predictions available of the future responses of species (Thomas et al., 2008). However, the distance to the nearest vegetation margin appears to be important for the diversity of some arthropods, confirming previous findings elsewhere outside Africa (Tscharrntke et al., 2005).

Vineyard/fynbos ecotone

Different arthropod taxa responded differently to vineyard/fynbos ecotone, acknowledging the fact that not all species respond similarly to habitat alterations and ecotones (Brittingham and Temple, 1983; Wilcove et al., 1986). The outer boundary of any habitat is not a fence, but rather a zone of varying structure and width depending on habitat type. In some cases, arthropods are attracted to the edge, which can sometimes function as an ecological trap (Gates and Gysel, 1978). Generally, the greater the structural contrast between adjacent terrestrial habitats (i.e. vineyard adjacent to fynbos), the more intense the edge effects (Samways and Moore, 1991). Generally, arthropods responded positively to this ecotone. Some species preferred to stay in a particular habitat rather than in the ecotone. This was the case for spider species *T. purcelli*, which was equally abundant in the vineyard, and rarely sampled from the adjacent natural fynbos. However, other species (e.g. Salticidae: *Evarcha* sp.;

and Thomisidae: *S. imitator*) were continuously distributed along the ecotone. This implies that vineyard/fynbos ecotone provides better habitat conditions for some arthropod species than the adjacent vegetation, and that different species require different habitat conditions, even among those species that belong to the same taxa.

The invasive alien Argentine ant (*Linepithema humile*) was abundant in vineyard/fynbos ecotone and adjacent vineyard, indicating that vineyards provide the best conditions for it. Indeed, its abundance was sharply reduced in the core of adjacent natural fynbos. Changes in temperature, light levels, humidity, wind, and soil properties at the edges may be influential to the associated arthropod diversity. However, Kotze and Samways (2001) studying surface-active amphipod, carabid and ant distribution patterns along Afromontane forest/grassland ecotones, found little evidence to support the biological edge effect. Here, a certain unidentified Scarabidae sp. was sampled only near the ecotone, some few meters into the adjacent vineyard, indicating that vineyard may be a preferred habitat than fynbos by a number of species. This indicated that vineyards may be of great importance for certain arthropod diversity conservation, and that they need to be considered in a biodiversity strategic management plan.

Vineyard/alien invaded ecotone

Generally, different arthropod groups had similar positive response patterns towards vineyard/alien invaded ecotone. The majority of arthropod taxa had highest species richness within the vineyard/alien invaded ecotone in comparison with adjacent vegetation, with more equal number of species on either side of the ecotone. This implies that the majority of arthropod species had an equal preference for alien invaded vegetation and vineyard habitats. This was the case for certain lepidopteran species, showing no clear preference for particular adjacent vegetation despite higher species richness within the ecotone. This ecotone type was dominated by species restricted to ecotone, vineyard or alien invaded area. Generally, spiders and beetles were having a discontinuous distribution or limited distribution, never continuous.

Determining where threatened species survive best is of great importance to habitat protection and management. The general perception is that species are likely to be persistent in central parts of geographical ranges where they tend to be relatively widespread and often in large abundance (Thomas et al., 1994; Thomas et al., 2008).

Nevertheless, more species within vineyard/alien invaded ecotone had relatively higher abundance. For most of these arthropod species, destructed alien invaded vegetation and vineyard habitats did not supported continuous distributions along an ecotone.

Alien cleared/vineyard ecotone

Generally, different arthropods respond differently towards ecotones (Brittingham and Temple, 1983; Wilcove et al., 1986). Nevertheless, some different arthropod taxa had similar response patterns towards the alien cleared/vineyard ecotone, with similar responses to that in the alien cleared/fynbos ecotone. The dominant taxa: spiders, beetles, flies, bugs, hymenopterans, and lepidopterans did not show clear preference for any particular adjacent vegetation. Nevertheless, more species had a greater abundance in the ecotone. This implies that different arthropod species show greater recruitment within an ecotone than in adjacent vegetation. Indeed, habitat fragmentation changes landscape structure (Burgess and Sharpe, 1981; Harris, 1984; Fischer and Lindenmayer, 2007) and does not affect all arthropod species equally (Kruess and Tschardtke, 2000).

Correlation between habitat edge distance and arthropod diversity

The overall non-significant correlation between increasing distances from the ecotone and the species richness and abundance was an indication of population dynamics along the ecotone, as most of the species richness decline took place in the first 32 m from the ecotone. It may also imply that the ecotone provided conditions comparable to those in adjacent habitat fragments. Contrary to Brown's (1984) findings that density is greatest at the habitat core and declines usually gradually towards the boundaries, here, more species showed an increased abundance in the ecotone. Some individual arthropod species abundance (e.g. *Linepithema humile*) had significant negative correlation with the distance from the vineyard edge into the core on natural fynbos. Nevertheless, the distance to the nearest vegetation margin appears to be of great importance for much arthropod diversity, such as bees and ants (Tschardtke et al., 2005). This indicated that individual species are more sensitive towards ecotone than are the higher taxonomic levels (e.g. order or family) and not stating the focal taxonomic level may give a misleading picture of arthropod response towards habitat edges. Most often, ecotones and the core of adjacent vegetation have unique species. So, ecotones should not only be assessed in term of species richness but also in term

of suitability for supporting particular species. This is despite the fact that there was no response of arthropods in general towards increasing distance from ecotone.

The species-specific response is probably driven by various factors. Kohler et al., (2008) found that hover fly species, *Episyrphus baltheatus* and *Phaerophoria scripta*, were strongly and positively related to flower abundance but not to distance, whereas *Melanostoma mellinum*, a species feeding mainly on grass pollen, was not related to flower abundance but negatively to distance. However, there were species-specific responses towards different ecotones. Moreover, more arthropod species responded similarly to the same type of ecotone in different locations, confirming the common assumption (Sisk and Haddad, 2002).

Habitat heterogeneity

An increase in habitat heterogeneity may lead to an increase in species diversity in a landscape (MacArthur and MacArthur, 1961), and has been suggested as a strategy of conserving species richness in habitats threatened by human activities (McGarigal and McComb, 1992, Greenberg et al., 1995). In agreement with Cramer and Willig (2005), there was no evidence here supporting the hypothesis that habitat heterogeneity, at least in terms of landscape elements, enhances arthropod diversity. Nonetheless, paired habitats were more similar in terms of species diversity. In contrast, habitat heterogeneity affects diversity by significantly altering the relative proportions of species in contrasting habitat fragments.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

These results confirmed established knowledge about species restricted to habitat fragments, and pointed to some recommendations for management and conservation of natural habitats within modified areas. Different ecotone types performed differently in supporting arthropods, depending on the type of neighbouring vegetation. The alien cleared/fynbos ecotone supported both greater species richness and abundance. In most cases, alien invaded vegetation was associated with the least species richness. Especially, when they were next to alien cleared vegetation or vineyards, the species richness was sharply reduced. A combination of vineyard and alien invaded vegetation supported the lowest arthropod diversity. Nevertheless, when these two are bordered by natural fynbos, species richness increased considerably. It is therefore crucial to

maintain natural fynbos patches adjacent to any highly disturbed areas, which will serve as refuge areas for arthropods during unfavourable conditions.

Species diversity was not necessarily higher in the ecotones than in adjacent habitat fragments. Nevertheless, individual species did not always have a continuous distribution along the ecotone. Indeed, some species had a preference for ecotones, while others were restricted to either one or the other side of the ecotone, indicating that ecotones are habitats that are suitable for certain species but not habitable to others. In short, different ecotone types are not suitable for all arthropod species but for those that are well adapted to the different adjacent habitats.

Species richness does not necessarily increase or decline with increased distance from ecotone. However, individual species abundance was highly influenced by the distance from the ecotone. This implies that there was no general response pattern for an overall arthropod species but rather species-specific responses. There were three possible classes of arthropod responses towards ecotone with respect to distance from the closest habitat fragment a) positive ecotone responses, where surface-active arthropod species abundance increase near the ecotone; b) neutral responses, where there was no pattern with respect to the ecotone; and c) negative responses, where surface-active arthropod species abundance decrease near the ecotone. Moreover, more arthropod species responded similarly to the same type of ecotone in different locations. As might be expected, some species were more frequently recorded along adjacent vegetation types than others. However, majority of sampled species increased significantly in abundance at the ecotone. Overall, spiders and beetles were more abundant and species rich in all ecotone types. For biodiversity management purposes, where preservation of intact native habitat is not possible, ecotones should be maintained wherever possible, despite their geographical location.

Thus, a conservation strategy for fynbos patches must also consider types of surrounding habitat fragments. Supporting findings by Gill et al. (1996), arthropods responded to fragmentation by avoiding highly disturbed areas or underutilizing them. Vineyard habitats are often perceived as less valuable than fynbos and, as a consequence, are subject to many anthropogenic disturbances such as fragmentation and afforestation. Protecting habitat fragments around vineyards not only helps conserve rich diversity, but also conserves the biota in the fynbos fragments and at the

edges, and would therefore be more meaningful in terms of the overall conservation of CFR biodiversity. If habitat loss and species extinction in the CFR are to be reduced, restoration actions are urgently required.

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APPENDICES

Appendix 4.a Arthropod species sampled along different ecotone types: alien cleared/fynbos (AC/FB), alien cleared/alien invaded (AC/AI), fynbos/alien invaded (FB/AI), alien cleared/vineyard (AC/VY), vineyard/alien invaded (VY/AI), and vineyard/fynbos (VY/FB) ecotone.

Order/Family	Species	ADJACENT VEGETATION TYPES					
		AC/FB	AC/AI	FB/AI	AC/VY	VY/AI	VY/FB
Araneae							
Amaurobiidae	<i>Chresiona</i> sp.1		8	5	1	1	2
Gallieniellidae	<i>Drassodella</i> sp.1	15	22	15	14	7	15
Gnaphosidae	<i>Camillina</i> sp.1	35	18	18	21	12	16
Gnaphosidae	<i>Camillina</i> sp.2		2			3	
Gnaphosidae	<i>Pterotricha varia</i>	15	2	5	34	18	22
Gnaphosidae	<i>Zelotes lightfooti</i>	21	21	17	21	8	15
Lycosidae	<i>Geolycosa</i> sp.1	3		4			5
Lycosidae	<i>Hogna</i> sp.1	2	7	3	7	1	9
Lycosidae	<i>Proevippa schreineri</i>	23	18	27	33	15	15
Lycosidae	<i>Proevippa</i> sp.1	35	16	9	30	10	27
Lycosidae	<i>Proevippa</i> sp.2	6	7	4	5	4	4
Lycosidae	Sp.1	8	3	12	6	7	28
Lycosidae	<i>Trabea purcelli</i>	17	17	15	22	25	17
Nemesiidae	<i>Pionothele straminae</i>	4	3		1		1
Oxyopidae	<i>Oxyopes</i> sp.	50	19	41	35	20	52
Palpimanidae	<i>Diaphorocellus</i> sp.1	19	6	17	16	9	12
Philodromidae	<i>Tibellus</i> sp.		1	4	1	12	
Pisauridae	<i>Rothus purpurissatus</i>	15	11	29	11	8	15
Salticidae	<i>Aelurillus</i> sp.1	38	11	16	10	6	10
Salticidae	<i>Evarcha</i> sp.1	26	26	21	39	16	60
Salticidae	<i>Habrocestrum</i> sp.1	24		7	12	5	8
Salticidae	<i>Langona</i> sp.1	30	25	21	27	10	9
Salticidae	undertermined genus	5	1	2	3		
Theridiidae	<i>Steatoda capensis</i>	12	18	10	6	7	8
Theridiidae	<i>Steatoda</i> sp.2	20	8	34	17	8	10
Theridiidae	<i>Theridion</i> sp.		3			63	8
Thomisidae	<i>Ozyptila</i> sp.		3				
Thomisidae	<i>Synema imitator</i>	12	5	8	12	3	125
Trochanteriidae	<i>Platyoides</i> sp.	3					
Zodariidae	<i>Diores capensis</i>	3	8	8	12	6	2
Zodariidae	<i>Diores youngai</i>	112	101	92	80	61	75
Blattodea							
Blaberidae	<i>Aptera fusca</i>	11	4	7	2	2	8
	<i>Temnopteryx phalerata</i>						
Blattidae	<i>phalerata</i>	31	11	16	13	4	22
Coleoptera							
Anobiidae	<i>Xestobium</i> sp.	42	36	78	66	121	89
Anthicidae	<i>Endomia elongatus</i>	39	13	48	74	11	20
	<i>Formicomus</i>						
Anthicidae	<i>coeruleus</i>	6	3	2	23	11	39
Buprestidae	<i>Achmaedera</i> sp.	40	4	36	72	7	42
Cantharidae	<i>Cantharis</i> sp.1	7		1	14	1	2
	<i>Acanthoscelis</i>						
Carabidae	<i>ruficornis</i>	11		6		2	1

Order/Family	Species	ADJACENT VEGETATION TYPES					
		AC/FB	AC/AI	FB/AI	AC/VY	VY/AI	VY/FB
Carabidae	<i>Anthia decemquittata</i>	3	17				1
Carabidae	<i>Caminara</i> sp.1	17	6		21	13	16
Carabidae	<i>Graphipterus trilineatus</i>					1	15
Carabidae	<i>Microlestia tabida</i>	89	38	25	9	1	60
Carabidae	<i>Passalidius fortipes</i>		1				
Carabidae	Sp.1	1	27	28			13
Chrysomelidae	<i>Leptinotarsa</i> sp.1	2					3
Cicindelinae	<i>Platydelia quadriguttata</i>						2
Cleridae	Sp.1	26	1	7	23	32	124
Coccinellidae	<i>Cheilomenes lunata</i>	20	8	2	35	30	23
Coccinellidae	<i>Harmonia axyridis</i>				18	11	6
Curculionidae	Sp.1	14	6	13	27	52	15
Curculionidae	Sp.2	5		2	2	2	
Curculionidae	Sp.3		7	1	15	1	6
Curculionidae	Sp.4	7	8		5	8	1
Dermestidae	<i>Dermestes maculatus</i>	10		1	9	6	
Dermestidae	<i>Dermestes</i> sp.	4	1	3	8	1	37
Elateridae	<i>Cardiotarsus acuminatus</i>	5	2	1	1	2	
Lampyridae	<i>Lampyrus disticollis</i>	1				2	
Lampyridae	<i>Luciola</i> sp.	1			3		
Lycidae	Sp.1	1					
Meloidae	<i>Decapotoma lunata</i>	2		1	2		2
Melyridae	<i>Apalochrus</i> sp.1	13	3	1			
Mordellidae	Sp.1	8					1
Mordellidae	Sp.2	17	6	9	11	5	12
Nitidulidae	Sp.1	32	12	10	8	8	8
Nitidulidae	Sp.2	17	9	2	42	24	12
Scarabaeidae	<i>Schizonycha</i> sp.	2		4	1		
Scarabaeidae	Sp.2	2	1	3	4	3	6
Scarabaeidae	<i>Aphodius</i> sp.	5	6		5	12	3
Scarabaeidae	Hopliini sp.	50	32	133	72	10	31
Scarabaeidae	<i>Scarabaeus rigosus</i>	22	11	28	3	1	3
Scarabidae	Sp.1		1	5	1	2	6
Silvanidae	<i>Oryzaephilus</i> sp.	3			3		5
Staphylinidae	<i>Paederinae</i> sp.	28	20	18	23	21	23
Tenebrionidae	<i>Psammodes striatus</i>						1
Tenebrionidae	Sp.1	31	6	38	10	1	25
Tenebrionidae	Sp.2	41		2	5	9	40
Tenebrionidae	Sp.3	1			2	1	1
Tenebrionidae	<i>Tenebrio</i> cf. <i>molitor</i>	9		9	4	1	9
Tenebrionidae	<i>Tenebrio</i> sp.1	26	3	28	9	46	38
Tenebrionidae	<i>Trigonopus</i> sp.1	31	7	14	37	28	19
Tenebrionidae	<i>Zophosis boei</i>	382	89	68	341	86	171
Tenebrionidae	<i>Zophosis</i> sp.1	2		6			
Collembola							
Sminthuridae	<i>Sminthurus viridis</i>		65	33	45	37	3
Crustacea							
Amphipoda	<i>Paramelita igroculus</i>	167	33	16	9	5	
Dermaptera							
Forficulidae	<i>Forficula senegalensis</i>	33	11	16	49	20	18
Diptera							
Anthomyiidae	<i>Anthomyia</i> sp.	25	26	14	8	17	16
Asilidae	<i>Daspletis</i> sp.	90	97	104	39	9	48
Bombyliidae	<i>Exoprosopa</i> sp.	1		2		1	

Order/Family	Species	ADJACENT VEGETATION TYPES					
		AC/FB	AC/AI	FB/AI	AC/VY	VY/AI	VY/FB
Calliphoridae	<i>Chrysomya chloropyga</i>	13	1	1	7	1	3
Calliphoridae	Sp.1	12	2		3	8	2
Calliphoridae	Sp.2	20	5	15	14	9	18
Cecidomyiidae	<i>Cecidomyia</i> sp.1	13	3	247		6	
Heleomyzidae	<i>Helomyza picta</i>	2	5	4	1	3	
Muscidae	Sp.1	8	4	2	5	3	12
Pipunculidae	Sp.1	20	66	81	36	59	9
Sarcophagidae	Sp.1		4	3	1	1	6
Sciaridae	Sp.1	2	156	30	22	56	
Syrphidae	Sp.1	8	1	5	4	1	
Tachinidae	<i>Dejeania</i> sp.	20	3	28	4	4	3
Tachinidae	<i>Gonia</i> sp.	1		1	2	4	3
Tachinidae	Sp.1	3	3		2	1	19
Tachinidae	Sp.2	3	3	1	2	4	9
Tachinidae	Sp.3	5	6	7	9	7	2
Tachinidae	Sp.4	1	2	2	1	3	4
Therividae	<i>Therevid</i> sp.			5	2		
Tipulidae	<i>Tipula jocosa</i>	2	1	4	1	2	
Hemiptera							
Anthocoridae	<i>Anthocoris</i> sp.1	15	6	3	16	11	2
Cercopidae	<i>Poophilus</i> sp.1	18	10	15	24	16	24
Cicadellidae	<i>Ciphalilus</i> sp.			2	4		
Cicindelidae	<i>Cicindela quadriguttata</i>		12		8	1	
Cydnidae	Sp.1	1	48			3	6
Miridae	<i>Deraeocoris</i> sp.	2	1		1	1	1
Pentatomidae	<i>Agonoscelis</i> sp.1	5			4	1	
Pentatomidae	<i>Agonoscelis</i> sp.2			1			
Pentatomidae	<i>Antestia lymphata</i>	2	2	3	3	1	3
Pentatomidae	<i>Bagrada hilaris</i>	2			19	4	
Pentatomidae	<i>Coenomorpha</i> sp.				1		2
Pyrrhocoridae	<i>Scantius forsteri</i>	13	7	3	3	7	9
Reduviidae	<i>Acanthaspis sanguinosa</i>		1				
Reduviidae	<i>Cleptria rufipes</i>	39	19	21	18	8	10
Reduviidae	<i>Ectrichodia crux</i>	1	2		1		2
Reduviidae	<i>Holoptilus</i> sp.1		1				
Reduviidae	?Sp.	1	18	2	2	7	
Piesmatidae	Sp.1				1		1
Hymenoptera							
Anthophoridae	<i>Thyreus delumbatus</i>	2		3	2	2	
Anthophoridae	<i>Xylocopa</i> sp.			2			
Apidae	<i>Apis mellifera</i>	7	8	3	23	5	9
Apidae	<i>Meliponula</i> sp.1	5	1		17	6	6
Braconidae	<i>Charops</i> sp.1	2	1			2	
Chalcididae	<i>Brachymeria kassalensis</i>			3	1		
Chrysididae	<i>Spintharina</i> sp.		3				
Formicidae	<i>Camponotus maculatus</i>	3	2	2	16	1	91
Formicidae	<i>Camponotus</i> sp.1	6	6	3	7	3	25
Formicidae	<i>Camponotus</i> sp.2	693	174	220	179	90	303
Formicidae	<i>Camponotus</i> sp.3	5	13	6		28	
Formicidae	<i>Crematogaster peringueyi</i>	238	74	170	77	55	37
Formicidae	<i>Dorylus helvolus</i>		1				
Formicidae	<i>Linepithema humile</i>	319	75	108	261	150	359

Order/Family	Species	ADJACENT VEGETATION TYPES					
		AC/FB	AC/AI	FB/AI	AC/VY	VY/AI	VY/FB
Formicidae	<i>Messor capensis</i>	19	50	11	4	20	11
Formicidae	Sp.1	8	9	11	6	12	9
Formicidae	Sp.2		3				1
Formicidae	Sp.3	22	37	21	14	17	76
Formicidae	<i>Tetramorium capense</i>	126	150	139	290	59	100
Halictidae	<i>Nomia amabilis</i>		26	42	1	1	
Halictidae	Sp.1	20	14	19	85	17	55
Halictidae	Sp.2		4	1	2		
Halictidae	Sp.3				4	2	
Masaridae	<i>Ceramius</i> sp.1				1		
Melittidae	Sp.1	26	4	5	2	1	2
Melittidae	Sp.2	11	4		8	2	7
Mutillidae	Sp.1	4	2		15	4	5
Mutillidae	Sp.2	3		1	6	1	6
Pompilidae	Sp.1	29	12	11	18	10	17
Pompilidae	Sp.2	4		4	12	3	7
Scoliidae	<i>Campsomeriella</i> sp.1	11	7	5	2	1	2
Sphecidae	Sp.1	3	5		2	3	4
Sphecidae	Sp.2	34	22	50	41	18	14
Sphecidae	Sp.3	4		2	3		6
Sphecidae	Sp.4	1					
Sphecidae	<i>Sphex bonaspei</i>	20	13	11	20	3	16
Sphecidae	<i>Sphex tydei</i> var <i>capensis</i>	7			4	1	6
Tiphiidae	Sp.1	2	8		3	4	1
Isopoda							
Porcellionidae	<i>Porcello</i> sp.	264	148	329	102	325	238
Isoptera							
Hodotermitidae	<i>Microhodotermes viator</i>	1025	283	491	195	48	179
Spirobolida							
Pachybolidae	<i>Centrobolus</i> sp.1	151	38	15	37	122	24
Pachybolidae	<i>Centrobolus</i> sp.2	534	490	297	305	461	83
Lepidoptera							
Arctiidae	<i>Rhodogastria amasis</i>	1	6	16	2	4	1
Geometridae	Sp.1	4		2	4	2	2
Lasiocampidae	<i>Eutricha capensis</i>	3	1				1
Lycaenidae	<i>Aloeides</i> sp.		2		6	4	6
Noctuidae	<i>Helicoverpa armigera</i>	3	5	26	9	54	7
Noctuidae	Sp.1	1	1		7	4	9
Noctuidae	Sp.2	4	1	9	16	14	28
Nymphalidae	<i>Cynthia cardui</i>	4	23	9	13	38	20
Psychidae	Sp.1	1			1		1
Pyalidae	Sp.1	3	2	4	4	4	2
Sphingidae	<i>Coelonia fulvinotata</i>				1	1	2
Tineidae	Sp.1	4	1	3	1		4
Tortricidae	Sp.1			3			
Lithobiida							
Lithobiidae	<i>Lithobius</i> sp.1	11	1	15	13	7	16
Lithobiidae	<i>Lithobius</i> sp.2	7	8	7	41	11	6
Mantodea							
Hymenopodidae	<i>Harpagomantis tricolor</i>	5			1		
Mantidae	<i>Polyspilota aeruginosa</i>					1	
Mantidae	Sp.1		5	4	12	1	1

Order/Family	Species	ADJACENT VEGETATION TYPES					
		AC/FB	AC/AI	FB/AI	AC/VY	VY/AI	VY/FB
Mantidae	Sp.2	6	2	1	8	1	5
Mantidae	Sp.3	8	4	12		1	3
Mecoptera							
Bittacidae	<i>Anomalobittacus gracilipes</i>	9		57		5	
Neuroptera							
Nemopteridae	<i>Laurhervasia setacea</i>	3			1		
Lithobiomorpha							
Peripatopsidae	<i>Peripatopsis</i> sp.		5	8			
Opiliones							
Phalangiidae	<i>Phalangium</i> sp.1	120	10	23	8	2	4
Orthoptera							
Gryllidae	<i>Cophogryllus</i> sp.	42	19	20	19	16	36
Gryllidae	<i>Gryllus bimaculatus</i>	117	78	67	53	70	108
Scolopendromorpha							
Scolopendromorphae	<i>Scolopendra cingulata</i>	6	3	2	13	4	14
Scorpiones							
Buthidae	<i>Uroplectes lineatus</i>	9		7			8
Scorpionidae	<i>Opisththalmus macer</i>	18	11	24	3		
Solifugae							
Eremobatidae	<i>Eremobates</i> sp.1	2	3		1		
Galeodidae	<i>Galeodes</i> sp.1	37	7	14	17	19	35
Galeodidae	<i>Galeodes</i> sp.2	35	8	6	34	14	49
Total sampled individuals		6299	3324	4095	3923	2999	3833
Total sampled arthropod species		133	104	99	128	118	108
%Total sampled arthropod species		68	53	50	65	60	55
Overall sampled species (abundance)				197 (24473)			

Appendix 4.b Mean arthropod species richness and abundance for the alien cleared/fynbos ecotone.

Variable	Adjacent vegetation types	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean			
						Lower Bound	Upper Bound	Min.	Max.
Species richness	Fynbos	36	22.57	6.879	1.061	20.43	24.72	7	37
	Alien cleared	36	18.90	5.665	.874	17.14	20.67	9	31
	<i>Total</i>	36	20.74	6.529	.712	19.32	22.16	7	37
Species abundance	Fynbos	36	72.38	41.259	6.366	59.52	85.24	22	191
	Alien cleared	36	77.67	38.537	5.946	65.66	89.68	12	175
	<i>Total</i>	36	75.02	39.769	4.339	66.39	83.65	12	191

Appendix 4.c Mean arthropod species richness and abundance for the alien cleared/alien invaded ecotone.

Variable	Adjacent vegetation types	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Min.	Max.
						Lower Bound	Upper Bound		
Species richness	Alien invaded	6	11.86	4.846	.819	10.19	13.52	3	25
	Alien cleared	6	15.63	5.094	.861	13.88	17.38	7	35
	<i>Total</i>	12	13.74	5.288	.632	12.48	15.00	3	35
Species abundance	Alien invaded	6	37.97	30.055	5.080	27.65	48.30	4	110
	Alien cleared	6	49.74	39.055	6.601	36.33	63.16	8	217
	<i>Total</i>	12	43.86	35.097	4.195	35.49	52.23	4	217

Appendix 4.d Mean arthropod species richness and abundance for the fynbos/alien invaded ecotone.

Variable	Adjacent vegetation types	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Min.	Max.
						Lower Bound	Upper Bound		
Species richness	Fynbos	6	17.29	5.802	.865	15.55	19.03	9	37
	Alien invaded	6	12.00	4.735	.676	10.64	13.36	2	25
	<i>Total</i>	12	14.53	5.878	.606	13.33	15.74	2	37
Species abundance	Fynbos	6	48.47	25.619	3.819	40.77	56.16	11	125
	Alien invaded	6	41.06	25.284	3.612	33.80	48.32	3	97
	<i>Total</i>	12	44.61	25.579	2.638	39.37	49.85	3	125

Appendix 4.e Mean arthropod species richness and abundance for the alien cleared/vineyard ecotone.

Variable	Adjacent vegetation types	N	Std.			95% Confidence Interval for Mean			
			Mean	Deviation	Std. Error	Lower Bound	Upper Bound	Min.	Max.
Species richness	Alien cleared	6	17.52	5.242	.809	15.89	19.16	8	30
	Vineyard	6	15.17	6.148	.949	13.25	17.08	4	32
	<i>Total</i>	12	16.35	5.801	.633	15.09	17.60	4	32
Species abundance	Alien cleared	6	46.57	23.245	3.587	39.33	53.82	17	127
	Vineyard	6	46.83	22.475	3.468	39.83	53.84	14	107
	<i>Total</i>	12	46.70	22.726	2.480	41.77	51.63	14	127

Appendix 4.f Mean arthropod species richness and abundance for the vineyard/alien invaded ecotone.

Variable	Adjacent vegetation types	N	Std.			95% Confidence Interval for Mean			
			Mean	Deviation	Std. Error	Lower Bound	Upper Bound	Min.	Max.
Species richness	Alien invaded	6	10.86	4.182	.645	9.55	12.16	3	21
	Vineyard	6	12.60	4.670	.721	11.14	14.05	6	26
	<i>Total</i>	12	11.73	4.492	.490	10.75	12.70	3	26
Species abundance	Alien invaded	6	34.86	26.219	4.046	26.69	43.03	3	117
	Vineyard	6	36.55	26.417	4.076	28.32	44.78	7	105
	<i>Total</i>	12	35.70	26.173	2.856	30.02	41.38	3	117

Appendix 4.g Mean arthropod species richness and abundance for the vineyard/fynbos ecotone.

Variable	Adjacent vegetations	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean			
						Lower Bound	Upper Bound	Min.	Max.
Species richness	Fynbos	6	16.90	5.098	.787	15.32	18.49	9	32
	Vineyard	6	14.98	6.479	1.000	12.96	17.00	6	38
	<i>Total</i>	12	15.94	5.875	.641	14.67	17.22	6	38
Species abundance	Fynbos	6	50.33	27.561	4.253	41.74	58.92	17	168
	Vineyard	6	36.74	23.938	3.694	29.28	44.20	6	107
	<i>Total</i>	12	43.54	26.553	2.897	37.77	49.30	6	168

Appendix 4.h Number of arthropod species sampled at an increasing distances from ecotone into the core of adjacent vegetation. ‘-’ and ‘+’ indicates different side of the ecotone (e.g. Alien cleared/fynbos: - indicates Alien cleared vegetation while + indicates fynbos vegetation).

Adjacent vegetation types	Component	-128m	-64m	-32m	-16m	-8m	-4m	-2m	+2m	+4m	+8m	+16m	+32m	+64m	+128m	Combined
Alien cleared/fynbos	Sampled individuals	336	539	509	453	416	426	575	502	424	369	446	297	608	392	6292
	Total sampled sp.	55	72	69	66	71	66	78	74	79	74	67	67	83	77	133
	%Total sampled sp.	41	54	52	50	53	50	59	56	59	56	50	50	62	58	
Alien cleared/ alien invaded	Sampled individuals	238	204	293	259	408	259	316	123	175	227	169	150	244	241	3306
	Total sampled sp.	56	57	61	59	66	49	58	38	56	42	47	33	42	43	104
	%Total sampled sp.	54	55	59	57	63	47	56	37	54	40	45	32	40	41	
Fynbos/alien invaded	Sampled individuals	207	288	175	328	321	281	332	270	295	314	354	346	357	227	4095
	Total sampled sp.	46	39	58	46	60	61	58	61	68	68	79	66	72	51	99
	%Total sampled sp.	46	39	59	46	61	62	59	62	69	69	80	67	73	52	
Alien cleared/Vineyard	Sampled individuals	313	190	283	249	308	335	289	237	283	338	226	269	232	371	3923
	Total sampled sp.	53	56	59	52	68	74	64	64	70	69	62	70	66	61	128
	%Total sampled sp.	41	44	46	41	53	58	50	50	55	54	48	55	52	48	
Vineyard/alien invaded	Sampled individuals	190	203	166	266	262	205	243	200	269	206	151	192	219	227	2999
	Total sampled sp.	50	48	50	55	52	49	62	47	58	46	41	49	41	50	118
	%Total sampled sp.	42	41	42	47	44	42	53	40	49	39	35	42	35	42	
Vineyard/fynbos	Sampled individuals	235	262	227	186	190	251	368	302	273	266	260	266	306	441	3833
	Total sampled sp.	51	53	60	55	48	65	69	70	60	56	58	54	61	75	108
	%Total sampled sp.	47	49	56	51	44	60	64	65	56	52	54	50	56	69	

Chapter 5 – Soil Compaction and Arthropod Species Richness Correlation in the Cape Floristic Region

INTRODUCTION

Soil compaction (dry bulk density) is one of many factors that threaten natural resources (Kirby, 2007), and its importance is often mentioned in popular materials for farmers and land practitioners (Davies et al., 1972). Soil compaction often occur when soil is subjected to mechanical stress, mainly through the use of heavy machinery (i.e. tractors in the vines) or overgrazing, especially during wet soil conditions. Compaction of the soil is directly caused by the passage of wheels or tracks. The indirect effects of soil compaction may be less clear (Kirby, 2007), and is often the result of highly variable soil characteristics (Boizard et al., 2000). Changing soil compaction is a form of physical degradation where soil productivity is reduced (Eudoxie and Springer, 2006), resulting also in environmental consequences (van den Akker and Soane, 2005). However, there is very limited research on its importance (Stone and Ekwue, 1993; Ekwue and Stone, 1995), and to my knowledge, no studies have ever considered the indirect consequences of soil structure on arthropod populations through its influence on vegetation or landscape of the Cape Floristic Region (CFR).

Soil warrants further attention because it may be more important in structuring the current vegetation (Mitchell, 1991). Many findings elsewhere have shown how the environment markedly influences vegetation composition and productivity, but there are few cases which show how vegetation influences the ecosystem (Roberts, 1987; Scholes and Nowicki, 1998), with the exception of alien (Payet et al., 2001).

Soil compaction influences infiltration and soil erosion (Chan et al., 2006). Soil erosion, in turn, is a serious environmental problem, and one obvious consequence of the reduction in surface vegetation cover in both plantations and vineyards. Reduction of soil porosity reduction causes increased runoff and erosion (Mitchell, 1991), and hence reduced surface cover. Moreover, farming operations using tractors often occur when the soils are moist and prone to soil compaction (Chan, 2006). However, the extent of soil compaction and its relative impact have not been quantified. Soil

compaction occurring during the harvesting period may well be reversed by soil-dwelling arthropods and plant root penetration. Some areas have soil with low infiltration and high runoff rate even on very low angle slopes because of high compaction and sheet erosion (Mitchell, 1991). The common understanding of soil compaction today is that it is 'problematic' by increasing runoff and erosion, and preventing vegetation establishment (Mitchell, 1991). Many insects spend part of or all their lives in the soil. Insects vary in their feeding habits, and many feed above ground and use the soil only as nesting site. Some insects such as ants, digger, wasps, and bees bring food into the soil through feeding their young (Borror et al., 1989).

Some soil fauna are important members of many ecosystems (Seastedt, 1984; Smit and Van Aarde, 2001; Hart and Klironomos, 2002) and their potential benefits to agro-ecosystems in terms of improving soil structure and nutrient cycling have been recognized (Lee, 1985; Edwards and Bohlen, 1996). Being soil inhabitants and obtaining their food supplies entirely from soils, their presence and abundance should be influenced by soil environmental conditions and therefore by the associated soil properties (Chan and Barchia, 2007). Plant litter decomposition is an important biological process driven by soil organisms (Dyer et al., 1990; Pausas et al., 2004). Better knowledge of the relationship between leaf litter arthropod populations and soil properties is thus useful for identifying appropriate management practices that will encourage the increase in abundance of desirable arthropod species (Chan, 2001). Moreover, such understanding will assist in the selection of the most suitable types of arthropod for introduction to areas that are totally lacking desirable species necessary in many agricultural fields. Lastly, the relationships between soil fauna abundance and soil properties could be used as indicators of the performance of landscape management in terms of soil quality (Clapperton et al., 2004).

For vineyards, compaction can be influenced either by superficial tillage and vehicular traffic during the life of the vineyard, or by deep tillage that is systematically undertaken before vine planting to loosen the soil and enhance vine rooting (Van Dijck and Van Asch, 2002; Lipiec and Hatano, 2003; Ferrero et al., 2005; Coulouma et al., 2006). Specific response of soils is significant in the case of deep cultivation, causing an increase of compacted zone fragmentation in relation to a decrease of clay content (Coulouma et al., 2006).

Invasion with pines often increases the above ground biomass and cause acidification (Armstrong and van Hensbergen, 1996; Payet et al., 2001), altering soil fertility and nutrient cycling. Morris (1984) reported changes in soil structure following plantation forestry in Swaziland. Soils under pine plantations were found to be more acidic, had lower cations (Ca^{2+} and K^+) and lower concentration of nitrogen and organic carbon in comparison with soils under native forest (Watts, 1951). A plant species in North America growing in soils that were experimentally invaded with *Alliaria petiolata* (garlic mustard) had lower ectomycorrhizal fungal root tip biomass compared to uninvaded soils (Wolfe et al., 2008), indicating unsuitability of soil to support species following invasion. Ectomycorrhizal fungi play important roles in the functioning of ecosystems such as forest (Read et al., 2004).

Disturbance of soil profile may directly or indirectly change the composition of arthropod diversity associated with it. In turn, biological parameters are important in sustaining critical soil functions and can serve as indicators which define soil quality (Block et al., 2002). However, assessing soil degradation is more challenging (Althoff and Thien, 2005) than evaluating its impacts.

Aims

In the Cape Floristic Region (CFR), the comparative influence of soil compaction under invasive alien trees (IATs) and vines on surface-active arthropod diversity has not yet been undertaken despite this area being a global biodiversity hotspot (Myers et al., 2000; Mittermeier et al., 2005). It is unclear how IATs and agricultural (specifically vineyard) practices influence soil compaction and thus the associated arthropod diversity. Consequently, the following specific questions were addressed:

- Does a change in structure type of vegetation alter soil compaction? The null hypothesis is that vegetation changes do not alter soil compaction.
- Is there any significant correlation between arthropod diversity, soil compaction, leaf litter, site location and soil moisture content under the different vegetation types.

STUDY SITES

The vegetation of the study area includes many species-rich communities, occurring on highly infertile soils derived from sandstone of the Table Mountain Group. These communities range from upland study sites where soils are well drained and rocky to sand-loamy soil on the foothill and in the valleys. Intrusion of shale in some vineyards (i.e. Rustenberg) was evident. Boucher and Moll (1981) recognize two main soil types in Stellenbosch area. Lithosols, which are generally shallow (<30 cm), grey sandy soils associated with the Table Mountain Group (TMG) sandstone, having a weak profile differentiation and contain coarse fragments and solid rock, and they are virtually the fynbos soil of the current study area. Ferralitic soils are sandy to loamy, poorly structured soils, and are derived from siltstone and coarse shale of the Malmesbury sediments. Ferralitic soils are often associated with the Renosterveld in Stellenbosch area (Buys et al., 1991). Each site consisted of pine or eucalypt plantations, invasive alien trees (IATs) stands, nearby disturbed fynbos, vineyards, and sites cleared of IATs (CIATs) with similar edaphic conditions derived from the same parent material, TMG sandstone. For each site, the topographical conditions of adjacent distinct vegetation were similar.

➤ *See Chapter 2 for details of study sites*

MATERIAL AND METHODS

From August 2006 to January 2008, three field visits at each sampling location were undertaken to sample arthropods and measure habitat characteristics (i.e. soil compaction, soil moisture and leaf litter). The following norms were used for sampling period determination: Sampling period 1 = September, October, November; sampling period 2 = December, January, February; and sampling period 3 = June, July, and August. Transects were used to sample arthropods, vegetation and soil. Transects have been widely used for studying arthropod assemblages and associated environmental variables (Whittaker, 1972; Jonhston et al., 1992; Procheş and Cowling, 2006). Soil compaction and % soil moisture measurements at each sampling station were undertaken using a Radioactive moisture-density gauge instrument (Troxler 3411-B). The 2 m² area quadrats for leaf-litter depth survey were created at each sampling

station around the trap set. Mean leaf-litter depth was estimated from three random measurements in each quadrat by inserting a steel rod, 4 mm in diameter, into the leaf-litter until the harder soil layer was reached (Lawes et al., 2005).

The distribution and diversity of surface-active arthropods within natural fynbos vegetation, IATs, CIATs and vineyard sites were studied here using pitfall traps. All sites were selected from an elevation ranging from about 304–390 m above sea level. Generally, surface-dwelling arthropod taxa are easily collected in pitfall traps, which provide a standardized sampling method for comparative analysis. Multiple sampling stations were chosen along transects running orthogonal to transitions between distinct vegetation fragments. Arthropod species assemblages were determined along 36 transects at sites experiencing similar environmental conditions but situated in areas with contrasting habitat context. A \log^2 scale was used to determine sampling stations along transects, extending 128 meters into both the adjoining vegetation fragments. The collected surface-active arthropods were sorted and placed into families (Appendix 5.1). Where possible, they were further identified to species level. Nevertheless, all specimens were recorded at least to family.

➤ *See Chapter 2 for details of Material and Methods*

DATA ANALYSES

One-way analyses of variances (ANOVA) were performed on the selected soil factors comparing the different vegetation sites (SPSS Inc., 2006). Classification Trees for all the vegetation types in terms of soil compaction, leaf litter depth, percentage soil moisture and species richness were produced separately using CHAID growth limits (SPSS Inc., 2006). Significance level for splitting nodes and merging categories was 0.05 and the significance values were adjusted using Bonferroni method. A variety of non-parametric species estimators were used to provide the best overall arthropod species estimates for all the vegetation types (Hortal et al., 2006). Incidence-based Coverage Estimator (ICE) was considered a robust and accurate estimator of species richness (Chazdon et al., 1998), whereas Chao2 and Jackknife estimators provide the least biased estimates for insufficient sampling (Colwell and Coddington, 1994). Therefore, these estimators were calculated using EstimateS (Colwell, 2006) for all the vegetation types separately and for a combination of these. One-way ANOVAs were

performed on the species and the log transformed abundance data comparing the different vegetation sites with multiple comparisons of the means using the Bonferroni methods (Legendre and Legendre, 1998). Overall correlations between species assemblages and soil factors were calculated using Spearman's Rank Order Coefficient, since the data were not normally distributed.

RESULTS

Soil factors in different vegetation types

Of all the sampled vegetation types, vineyard sites had the highest soil compaction (Table 5.1). Fynbos and CIATs supported higher and more similar soil compaction (Table 5.1). IAT sites had the lowest soil compaction measured (Table 5.1). Contrary to soil compaction, vineyard sites had the lowest percentage moisture content, while fynbos and IATs had the highest. Although CIATs moisture content was not the lowest, it was less than in fynbos and IATs (Table 5.1). IATs and natural fynbos had larger leaf litter depths, while vineyard and CIATs sites had the lowest (Table 5.1).

Table 5.1 Soil factor means, standard deviation (Std.Dev), variance and standard error (Std.Err.) for fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites.

Variable	Vegetation types	Means	N	Std.Dev.	Variance	Std.Err.
Soil compaction (Kg/m³)	Cleared IATs	1277.1	115	129.3159	16722.61	12.05877
	Fynbos	1270.7	145	142.4644	20296.09	11.83102
	IATs	1205.4	124	175.5982	30834.72	15.76918
	Vineyard	1341.6	126	109.4648	11982.55	9.7519
	<i>All groups</i>	<i>1273.8</i>	<i>510</i>	<i>148.8711</i>	<i>22162.61</i>	<i>6.59212</i>
Percentage soil moisture	Cleared IATs	8.7783	115	2.918416	8.51715	0.272144
	Fynbos	9.7614	145	3.322152	11.03669	0.27589
	IATs	9.4911	124	3.34421	11.18374	0.300319
	Vineyard	8.3619	126	2.502314	6.26158	0.222924
	<i>All groups</i>	<i>9.1282</i>	<i>510</i>	<i>3.096697</i>	<i>9.58954</i>	<i>0.137124</i>
Leaf litter depth (mm)	<i>All groups</i>	<i>346.31</i>	<i>360</i>	<i>206.551</i>	<i>42663.32</i>	<i>10.88619</i>
	Cleared IATs	13.652	115	5.534649	30.63234	0.516109
	Fynbos	15.572	145	4.640918	21.53812	0.385407
	IATs	20.798	124	6.436658	41.43057	0.578029
	Vineyard	2.7381	126	2.367036	5.60286	0.210872
Elevation	<i>All groups</i>	<i>13.239</i>	<i>510</i>	<i>8.194446</i>	<i>67.14895</i>	<i>0.362856</i>
	Cleared IATs	390.48	115	226.6297	51361.03	28.55266
	Fynbos	372.96	145	249.3724	62186.59	23.05449
	IATs	320.29	124	202.6135	41052.23	19.9641
	Vineyard	304.47	126	64.9157	4214.04	7.39782
<i>All groups</i>	<i>346.31</i>	<i>510</i>	<i>206.551</i>	<i>42663.32</i>	<i>10.88619</i>	

Classification and comparisons of soil factors in different vegetation types

Analysis of variance (ANOVA) among fynbos, IATs, CIATs, and vineyard sites showed that there were significant differences in soil compaction ($df = 3$, $f = 19.355$, $p = 0.00$). There were also significant differences among vegetation types in terms of leaf litter depth ($df = 3$, $f = 296.618$, $p = 0.00$) and percentage soil moisture ($df = 3$, $f = 15.800$, $p = 0.001$).

The classification tree significance values were adjusted using Bonferroni method, and indicated similarity between fynbos and CIATs (Figure 5.1). IATs and vineyards were significantly different from each other ($p > 0.05$). Moreover, both the IATs and vineyards were not comparable to either fynbos or CIATs. Classification of site locations based on soil compaction resulted in three nodes (Figure 5.2). All nature reserves (Helderberg, Hottentots Holland and Jonkershoek) had similar low soil compaction (Node 1) (Figure 5.1). However, some vineyard sites (Node 2) had significantly higher soil compaction

than any other location (Figure 5.2). Rustenberg, Bilton, and Dornier vineyards had more comparable soil compaction (Node3) (Figure 5.2). Classification of different vegetation types in terms of leaf litter depths resulted in four separate categories (Figure 5.3). Moreover, there were significant differences between the ten site locations ($df = 506$, $f = 35.357$, $p = 0.00$) in terms of leaf litter depths, with the highest leaf litter depths recorded from Jonkershoek and Hottentots Holland nature reserves respectively (Figure 5.4). However, Helderberg nature reserve was associated with relatively lower leaf litter depth, which was comparable to those in some vineyards. As expected, vineyards clustering (Node 2) resulted in lower leaf litter depths (Figure 5.4). Classification of different vegetation types using percentage soil moisture content resulted in two vegetation categories (Figure 5.5). CIATs were comparable to vineyards, whereas fynbos and IATs had relatively similar percentage soil moisture content (Figure 5.5).

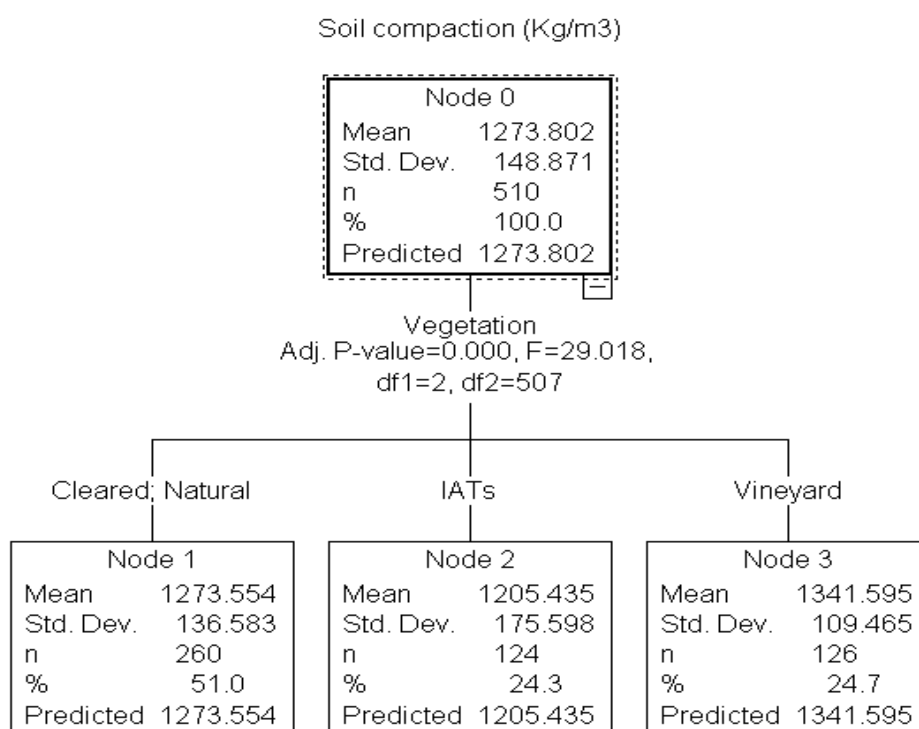


Figure 5.1 Classification tree for vegetation types in terms of soil compaction. Cleared = vegetation cleared of invasive alien trees, natural = natural fynbos, IATs = invasive alien trees.

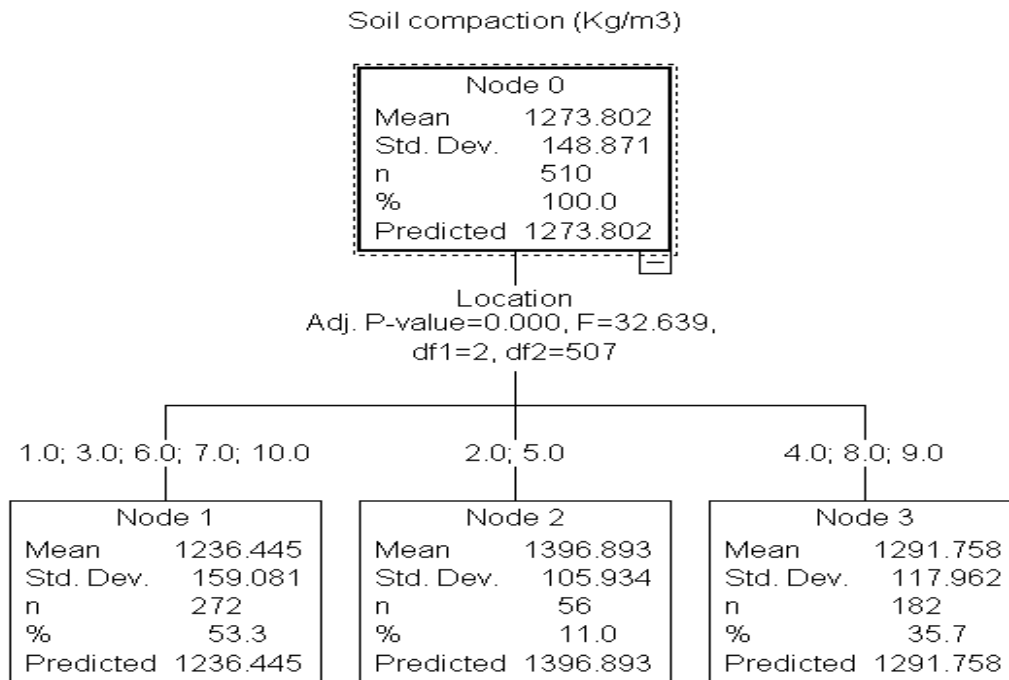


Figure 5.2 Classification tree for site locations in terms of soil compaction. 1.0 = Vergelegen, 2.0 = Stellenzicht-Driekoppen, 3.0 = Waterford, 4.0 = Rustenberg, 5.0 = Waterford-Driekoppen, 6.0 = Helderberg, 7.0 = Hottentots Holland, 8.0 = Bilton, 9.0 = Dornier, 10.0 = Jonkershoek.

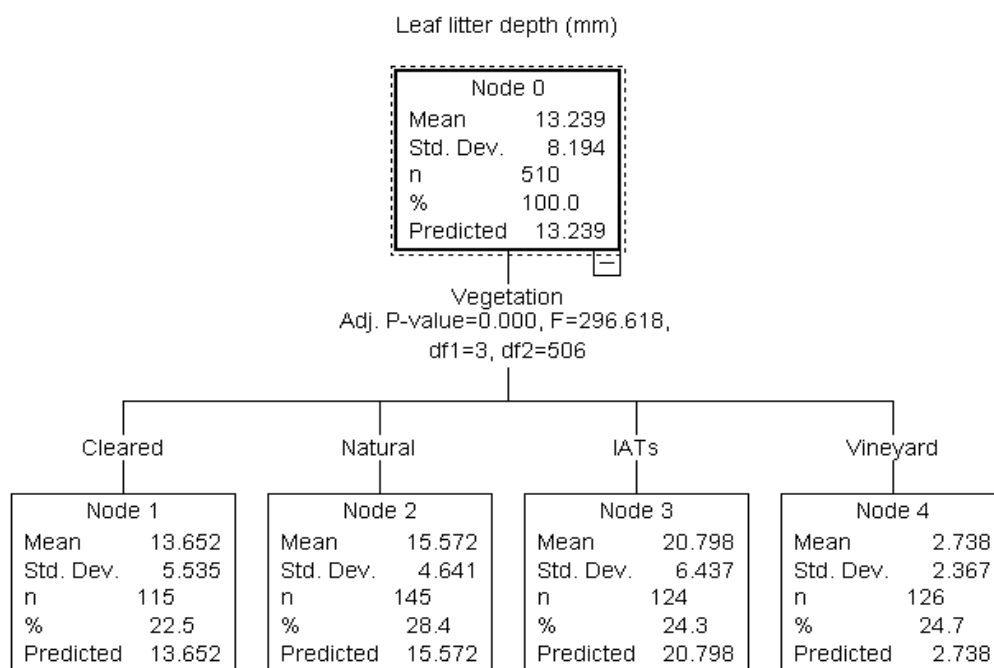


Figure 5.3 Classification tree for vegetation types in terms of leaf litter depth. Cleared = vegetation cleared of invasive alien trees, natural = natural fynbos, IATs = invasive alien trees.

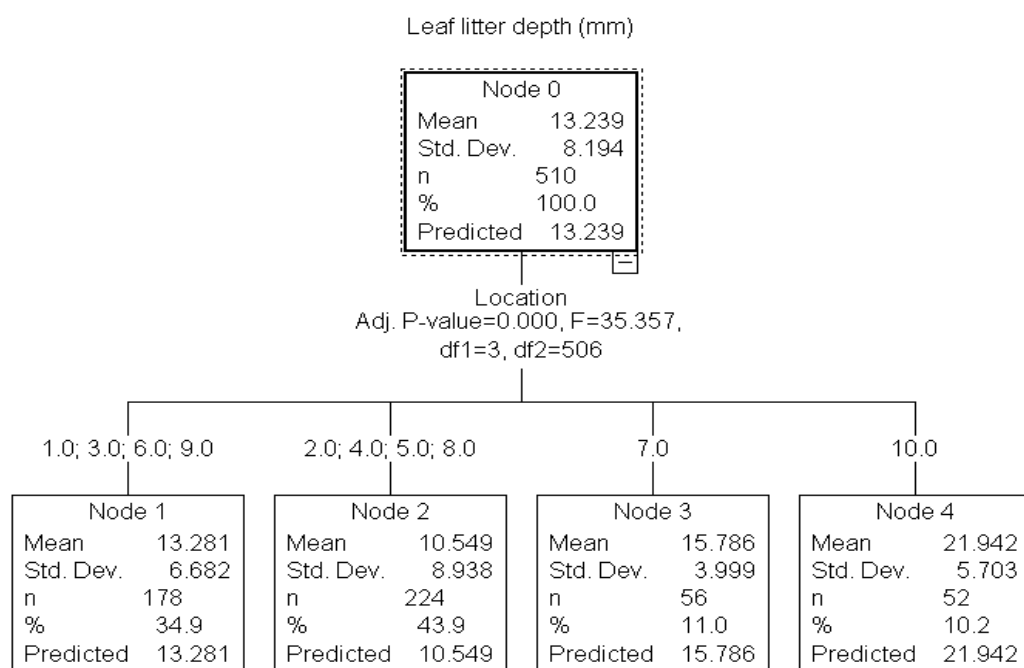


Figure 5.4 Classification tree for site locations in terms of leaf litter depth. 1.0 = Vergelegen, 2.0 = Stellenzicht-Driekoppen, 3.0 = Waterford, 4.0 = Rustenberg, 5.0 = Waterford-Driekoppen, 6.0 = Helderberg, 7.0 = Hottentots Holland, 8.0 = Bilton, 9.0 = Dornier, 10.0 = Jonkershoek.

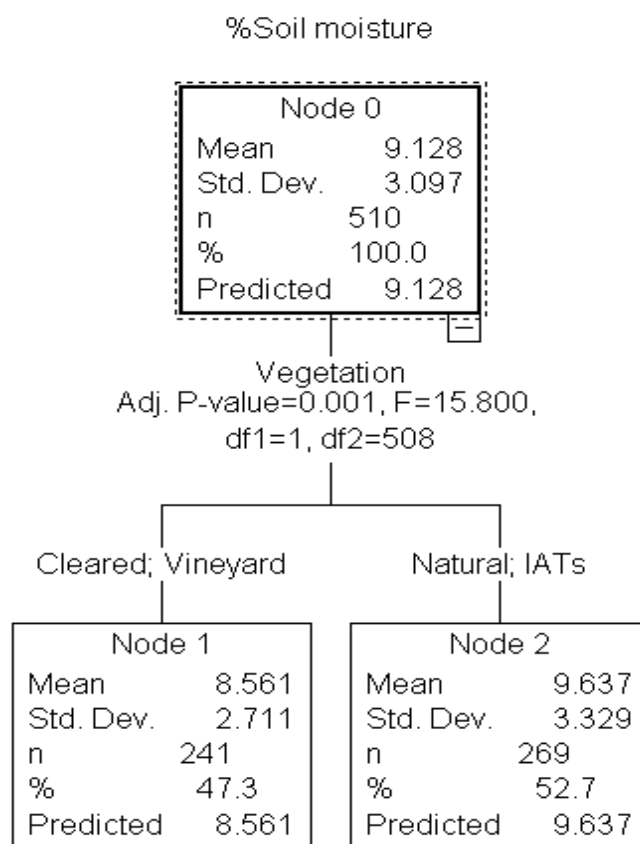


Figure 5.5 Classification tree for vegetation types in terms of percentage soil moisture. Cleared = vegetation cleared of invasive alien trees, natural = natural fynbos, IATs = invasive alien trees.

Arthropod species assemblage in different vegetation types

The species estimators for all the sites are given in **Chapter 3**. Fynbos and CIATs supported relatively higher mean species richness and abundance than IATs and vineyards (Table 5.2). ANOVA among fynbos, IATs, CIATs, and vineyard sites showed that there were significant differences among ($df = 3, f = 62.586, p = 0.00$), and within ($df = 509, f = 62.586, p = 0.00$) vegetation types in terms of species richness. Nested ANOVA's among fynbos, IATs, CIATs, and vineyard sites indicated significance differences in species richness between fynbos and IATs ($p = 0.00$); fynbos and vineyard ($p = 0.00$); CIATs and IATs ($p = 0.00$); and CIATs and vineyard ($P = 0.00$). However, there were no statistically significant differences between CIATs and fynbos vegetation types ($p = 1.00$) in terms of arthropod species richness

(Figure 5.6). Although vineyards had relatively higher mean species abundance than IATs (Table 5.2), the difference was not significant ($p = 1.00$). Detailed species data are given in Appendix 5.1.

Table 5.2 Species richness and abundance means, standard deviation (Std.Dev), variance and standard error (Std.Err.) for the fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites.

Variable	Vegetation type	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean	
						Lower Bound	Upper Bound
Species richness	Fynbos	145	18.06	6.284	0.522	17.03	19.09
	IATs	124	11.57	4.601	0.413	10.75	12.39
	Cleared IATs	115	18.13	5.736	0.535	17.07	19.19
	Vineyard	126	14.06	5.547	0.494	13.08	15.03
	<i>All vegetations</i>	510	15.51	6.239	0.276	14.97	16.05
Abundance	Fynbos	145	53.42	33.298	2.765	47.96	58.89
	IATs	124	38.10	26.496	2.379	33.39	42.81
	Cleared IATs	115	65.77	39.253	3.660	58.51	73.02
	Vineyard	126	41.44	25.885	2.306	36.87	46.00
	<i>All vegetations</i>	510	49.52	33.239	1.472	46.63	52.41

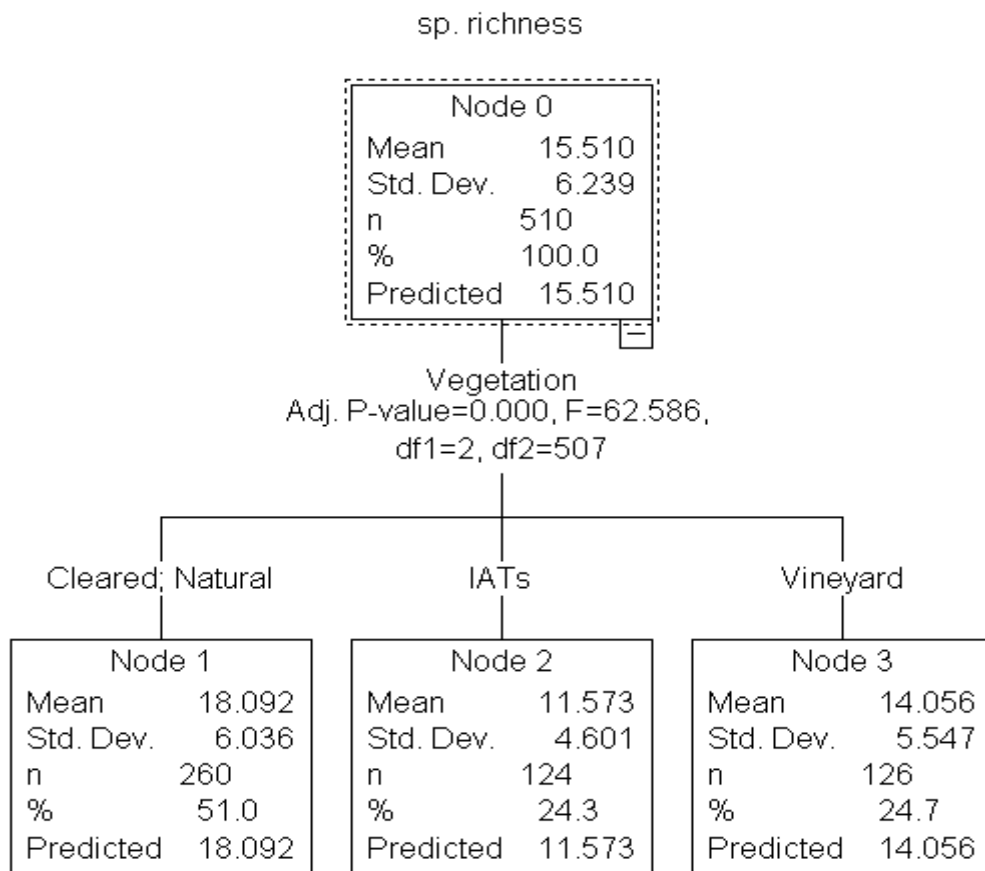


Figure 5.6 Classification Tree for vegetation types in terms of mean arthropod species richness. Cleared = vegetation cleared of invasive alien trees, natural = natural fynbos, IATs = invasive alien trees.

Correlation between soil factors and arthropod species richness and abundance in different vegetation types

Generally, there were significant correlations between soil factors and arthropod species richness and abundance (Table 5.3). Moreover, there was an overall significant correlation between vegetation type and soil compaction (Spearman's Coefficient = 0.154; $p = 0.000$, $n = 510$).

Table 5.3 Correlations between soil factors and overall arthropod assemblages in fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs) and vineyard sites.

Variable	Spearman's rho	Species richness	Abundance	Soil compaction (Kg/m ³)	Percentage soil moisture	Elevation	Leaf litter depth (mm)	Location
Species richness	Correlation Coefficient	1.000	0.696**	-0.177**	0.005	-0.124*	-0.020	-0.240**
	Sig. (2-tailed)	.	0.000	0.000	0.919	0.019	0.651	0.000
Abundance	Correlation Coefficient	0.696**	1.000	0.011	0.061	-0.264**	-0.033	-0.218**
	Sig. (2-tailed)	0.000	.	0.808	0.171	0.000	0.454	0.000
Soil compaction (Kg/m³)	Correlation Coefficient	-0.177**	0.011	1.000	-0.417**	0.061	-0.277**	0.031
	Sig. (2-tailed)	0.725	0.000	.	0.000	0.248	0.000	0.491
Percentage soil moisture	Correlation Coefficient	0.005	0.061	-0.417**	1.000	0.293**	0.188**	0.181**
	Sig. (2-tailed)	0.919	0.171	0.000	.	0.000	0.000	0.000
Elevation	Correlation Coefficient	-0.124*	-0.264**	0.061	0.293**	1.000	0.022	0.270**
	Sig. (2-tailed)	0.019	0.000	0.248	0.000	.	0.678	0.000
Leaf litter depth (mm)	Correlation Coefficient	-0.020	-0.033	-0.277**	0.188**	0.022	1.000	0.174**
	Sig. (2-tailed)	0.651	0.454	0.000	0.000	0.678	.	0.000
Site location	Correlation Coefficient	-0.240**	-0.218**	0.031	0.181**	0.270**	0.174**	1.000
	Sig. (2-tailed)	0.000	0.000	0.491	0.000	0.000	0.000	.
N		510	510	510	510	510	510	510

** Correlation is significant at the 0.01 level (2-tailed). * Correlation is significant at the 0.05 level (2-tailed).

DISCUSSION

Soil factors in different vegetation types

Higher soil compaction within the vineyards may be a result of many activities by farm workers and tractors during grape production. Surprisingly, IATs had significantly lower soil compaction compared to either fynbos or CIATs. Such lower compaction in IATs may be attributed to the effect of Eucalypts species that provided lower compaction in most study sites. More often, low compaction soil was evident in most of the IATs (i.e. under Black wattle *Acacia mearnsii* and Grey poplars *Populus canescens*). Generally, sites with pine species had little or no surface cover, exposing soil to surface to severe erosion during rainy times. However, some IAT species (i.e. pines) supported relatively higher soil compaction. Although with high soil compaction, vineyard sites had the lowest percentage moisture content. As expected, evaporation will be greater in areas such as vineyards than in dense fynbos or IAT vegetation. Soil moisture was more responsive to the type and structure of associated vegetation. For the same reason, fynbos and IATs had the highest soil moisture content respectively. Clearing of IATs resulted in slightly reduced soil moisture compared to that in IATs. In addition to relatively higher soil moisture content, IATs and fynbos supported larger leaf litter depths, while vineyards and CIATs had the lowest respectively. It is possible that there is a relationship between soil moisture content and the decomposition of dead material.

The importance of soil factors in determining the distribution of fynbos species has been emphasized (Richards et al., 1997). Indeed, abiotic soil factors are important in determining arthropod species assemblage (Wallwork, 1970; Manzer et al., 1984; Wall and Moore, 1999; Dunxiao et al., 1999; Holland and Luff, 2004; Jabin, 2008). Therefore, any soil disturbance may affect arthropod assemblages and other associated taxa.

Classification and comparison of soil factors in different vegetation types

Generally, there were significant differences between different vegetation types in soil compaction, leaf litter depth and percentage soil moisture. Fynbos and CIATs had similar soil compaction. It is unclear whether this is an indication of soil recovery following clearing of IATs. Soil compaction in IATs was significantly lower compared to that in fynbos and CIATs. In contrast to the IATs, vineyards had the highest soil

compaction. Interestingly, soil compaction was comparable in all the nature reserves, even though they were far from another. This clearly indicates the effect of land-use on soil compaction. This was shown by all vineyards having similar and higher soil compaction. This implies that soil compaction levels are not confined to a particular local area but is relative to land use. These results arrive from farming operations using tractors that often occur when the soils are moist and prone to soil compaction (Chan, 2006). Increased soil compaction can potentially reduce root penetration, water extraction and plant growth (Kirkegaard et al., 1992; Passioura, 2002), and evidenced from reductions in crop production as a result of soil compaction that have been reported across a wide range of soil types and environments (e.g. McGarry and Chan, 1984; McGarry, 1990; Radford et al., 2001; Hamza and Anderson, 2003).

Leaf litter depth was significantly different among different vegetation types. The higher leaf litter depth in IATs may be attributed to the lack essential detritivores. Nevertheless, fynbos sites had significantly lower leaf litter depth compared to IATs. This may be an indication of the availability of detritivores under fynbos vegetation. Highly decomposed materials were evident from fynbos as opposed to IATs and vineyards. Sites with a similar vegetation type also supported different depths of leaf litter depending on the structure of vegetation or vineyard management. Fynbos vegetation dominated by *Protea* species had relatively higher leaf litter depth than *Erica* dominated vegetation. In the case of vineyards, higher leaf litter depth was measured in areas where there was a cover crop. Sites cleared of IATs were comparable to the vineyard sites, whereas fynbos and invaded sites had significantly similar and higher percentage soil moisture content. Such higher moisture content in fynbos and IATs, may be the result of vegetation structural composition that arguably reduced evaporation from the understorey.

Arthropod diversity in the different vegetation types

Sites cleared of invasive alien trees (CIATs) were very similar to the fynbos sites in its arthropod species richness. Indeed, they supported about 90% and 85% of the total sampled arthropod species respectively. On the other hand, vineyard and IATs sites supported the least number of species respectively. Chemical application and grape harvesting activities might have resulted in a greater soil compaction in the vineyard (Ferrero et al., 2005). However, the intensity of this compaction can be enhanced by

relatively higher soil water content in lower parts of the slope where most of the vineyards are found. The effect of soil compaction may be more pronounced at high than at low soil water contents (Usowicz et al., 1992). Nevertheless, there was species overlap between fynbos, IATs, CIATs and vineyards. Interestingly, there was higher species richness in vineyard sites than in IATs. Does this mean that vineyard sites are better habitats than IATs? The answer to this appears to be 'yes'. Species diversity at different locations can vary considerably, with Vergelegen wine Estate for example, supporting more species than the Jonkershoek nature reserve. This implies that, in addition to the available fynbos habitats, Wine estates may be more important than IATs in conserving arthropod diversity.

IATs soil supported fewer arthropod species than did that of native vegetation. These results support the finding of Watts (1951), where native vegetation soil had more taxa of soil fauna compared to invasive alien vegetation. In addition, IATs sites were drier than native vegetation sites. IATs stands may have modified the vegetation and appear to be functionally different from stands of native trees, confirming findings in Michigan (Leege and Murphy, 2001). Arthropod species richness was relatively high in fynbos, with its wide range of soil compaction levels, compared to IATs, which had relatively uniform soil compaction. However, millipedes and woodlice were abundant in IATs. Indigenous ant species were found in IATs soil, but in lower richness and abundance in comparison with those of fynbos. Similar findings on ants were obtained previously from the same region (Donnelly and Giliomee, 1985).

The strong correlation between elevation and arthropod species richness as well as abundance may be an indication that they are responsive to different elevations. Nevertheless, it was unclear how elevation influenced surface-active arthropod diversity in general.

Correlation between soil factors and arthropod species richness and abundance in different vegetation types

Correlations between increased soil compaction and reduction in species diversity have been established (Heisler and Kaiser, 1995; Schrader and Lingnau, 1997; Dittmer and Schrader, 2000). Here, there were significant correlations between soil factors and arthropod species richness and abundance. Perhaps the most interesting result is the

significant correlations between soil compaction and arthropod species richness. There was sufficient evidence to reject the null hypothesis that all land-uses are equally associated with soil compaction. This implies that disturbance of soil profile had great impact on the associated surface-active arthropods. Soil has an effect on the amount of water held to enhance plant growth (Thomas and Squires, 1991), and thus provision of habitat for arthropods.

The density of soil fauna such as Collembola (Usher, 1975) in an agricultural area is closely linked to soil structure and function. Vineyard activities may change the soil compaction and alter the characteristics in soil. These activities often lead to a reduction in habitable pore space for the soil mesofauna (Larsen et al., 2004). With the reduction or loss of pore space, water infiltration and gas diffusion is reduced, soil oxygen concentration is decreased, and carbon dioxide concentration increases, possibly to toxic levels (Watson and Kelsey, 2006). This deterioration in quality of the soil conditions may make the soil less favourable for arthropods.

Both IATs and vineyards led to relatively homogenous soil compaction. Fynbos and CIATs sites had similar soil compaction, and hence comparable arthropod species richness. This implies that IATs clearing might have led to a reversion in soil compaction and arthropod assemblages to the condition which is similar to that in natural vegetation, at least after ten years of subsequent follow-up clearing.

There was a significant change in arthropod assemblage composition within the IATs when compared to vineyards. Studies on plant-animal relationship in fynbos vegetation of South Africa have been carried out previously (Bond and Slingsby, 1983 and 1984; Knight, 1988; French and Major, 2001) and in savanna (Boughey, 1963). In both vegetation types, soil profile significantly influenced fauna composition (McNaughton, 1988). Abiotic and biotic variables may interact, affecting populations through indirect effects (Farji–Brener et al., 2008). For instance, litter nutrients and changes in rainfall may influence invertebrate litter fauna directly, by regulating food availability and persistence, or indirectly through their interaction with parasitoids and predatory arthropods (Lensing and Wise, 2006; Classen et al., 2007; McGlynn et al., 2007). Consequently, certain arthropods (e.g. antlions) are often less abundant in areas with soils exposed to rainfall and sites with high abundance of leaf-litter (McClure, 1976; Lucas, 1982; Gotelli, 1993; Crowley and Linton, 1999; Arnett and Gotelli,

2001; Lomascolo and Farji-Brener, 2001; Farji-Brener, 2003). Moreover, these direct effects on soil structure may also affect species density through indirect effects mediated by their potential influence on plants (Van Dijck and Van Asch, 2002; Coulouma et al., 2006). The physical characteristics of soils affect seedling establishment, plant growth and density (Bazzaz, 1996). Plant abundance, in turn, may influence (1) the temperature of the soil surface directly by creating shade and (2) the amount of leaf litter (Farji–Brener et al., 2008), necessary to soil fauna.

Effects of pine cultivation on soils

Soils under IATs, which were largely pine trees here, were highly compacted. These results confirm other findings (Payet et al., 2001), where soils beneath stands of exotic pines were higher. Cultivation of pines led to an increase acidification and enhanced exchangeable aluminium in soils (Payet et al., 2001), which alters enzyme functioning and other metabolic roles. Moreover, the establishment of alien plantation forestry generally tends to enhance acidity and reduces nutrient availability in the soils (Ogden and Stewart, 1995). Yet here, there was no general impact of IATs on soil compaction, with the influence of IATs on arthropods, through soil compaction, also being species-specific. This implies that generalization of the impacts of IATs should be considered with care.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

The results of this study demonstrated the susceptibility of IATs and vineyard soils to compactive degradation. In contrast, efficiency of soil to support higher arthropod diversity within the vineyards can be increased if operations are scheduled at appropriate moisture contents (i.e. in dry soils). The similar classification of vegetation types based on arthropod species richness and soil compaction levels indicated that these two had responded to the impacts of IATs and its consequent removal. The arthropods were more numerous and evenly distributed in the soil under native vegetation than under IATs. During sampling, leaf litter under fynbos vegetation was highly decomposed, whereas those under IATs were less or not decomposed at all. These clearly indicated that IATs leaf litter was less palatable than that of fynbos vegetation to the arthropods associated with the decomposition process. High loads of non-decomposed leaf litter may be also attributed to the fewer detritivorous species in IATs. Vineyards increased soil compaction, making the soil more water repellent.

Water infiltration was altered, resulting in a much reduced water holding capacity in the vineyards. There was a significant change in soil compaction and the associated arthropod assemblages in response to invasive alien trees removal. Both soil compaction and arthropod assemblages in CIATs sites were more comparable to that of fynbos vegetation. The soil profile is highly compacted under vineyard areas compared to fynbos vegetation. However, soil moisture content was relatively higher in fynbos and IATs than that in vineyard and cleared of invasive alien tree vegetation. There was significant correlation between surface-active arthropod diversity, soil compaction, and site. However, there were no significant correlation between arthropod diversity, leaf litter and soil moisture. Nevertheless, where the decomposed organic matter was relatively high, the arthropod species richness was generally high as well.

Overall, these results indicate the extremely negative impact of alien invasive vegetation, much more so than vineyards. Yet, this negative situation is reversible, with removal of invasive aliens being associated with rapid recovery of soil conditions, and improved arthropod assemblages.

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APPENDICES

Appendix 5.1 Arthropod species and their trophic guilds sampled from different vegetations: fynbos, invasive alien trees (IATs), cleared of invasive alien trees (CIATs) and vineyard sites.

Trophic guild	Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
	Araneae						
Predator	Amaurobiidae	<i>Chresiona</i> sp.1	12	4	1	1	18
Predator	Gallieniellidae	<i>Drassodella</i> sp.1	30	24	19	15	88
Predator	Gnaphosidae	<i>Camillina</i> sp.1	28	32	35	26	121
Predator	Gnaphosidae	<i>Camillina</i> sp.2		3		2	5
Predator	Gnaphosidae	<i>Pterotricha varia</i>	24	15	19	39	97
Predator	Gnaphosidae	<i>Zelotes lightfooti</i>	31	19	35	18	103
Predator	Lycosidae	<i>Geolycosa</i> sp.1	12		5		17
Predator	Lycosidae	<i>Hogna</i> sp.1	8	5	2	14	29
Predator	Lycosidae	<i>Proevippa schreineri</i>	41	22	32	38	133
Predator	Lycosidae	<i>Proevippa</i> sp.1	45	12	33	37	127
Predator	Lycosidae	<i>Proevippa</i> sp.2	11	2	11	6	30
Predator	Lycosidae	Sp.1	29	5	5	25	64
Predator	Lycosidae	<i>Trabea purcelli</i>	34	16	22	45	117
Predator	Nemesiidae	<i>Pionothele straminea</i>	3		4	2	9
Predator	Oxyopidae	<i>Oxyopes</i> sp.	78	35	41	64	218
Predator	Palpimanidae	<i>Diaphorocellus</i> sp.1	41	12	20	7	80
Predator	Philodromidae	<i>Tibellus</i> sp.	3	5	3	9	20
Predator	Pisauridae	<i>Rothus purpurissatus</i>	31	23	21	14	89
Predator	Salticidae	<i>Aelurillus</i> sp.1	38	15	41	10	104
Predator	Salticidae	<i>Evarcha</i> sp.1	53	9	56	72	190
Predator	Salticidae	<i>Habrocestrum</i> sp.1	29	4	14	9	56
Predator	Salticidae	<i>Langona</i> sp.1	47	18	32	25	122
Predator	Salticidae	undetermined genus	6		4	1	11
Predator	Theridiidae	<i>Steatoda capensis</i>	13	21	20	8	62
Predator	Theridiidae	<i>Steatoda</i> sp.2	40	19	27	12	98
Predator	Theridiidae	<i>Theridion</i> sp.	11			63	74
Predator	Thomisidae	<i>Ozyptila</i> sp.		3			3
Predator	Thomisidae	<i>Synema imitator</i>	115	3	16	34	168
Predator	Trochanteriidae	<i>Platyoides</i> sp.	3				3
Predator	Zodariidae	<i>Diores capensis</i>	8	9	7	15	39
Predator	Zodariidae	<i>Diores youngai</i>	186	85	131	120	522
	Blattodea						
Omnivore	Blaberidae	<i>Aptera fusca</i>	27	3	3	2	35
Omnivore	Blattidae	<i>Temnopteryx phalerata</i>	43	15	29	11	98
	Coleoptera						
Wood borer	Anobiidae	<i>Xestobium</i> sp.	42	109	45	237	433
Omnivore	Anthicidae	<i>Endomia elongatus</i>	62	57	64	22	205
Omnivore	Anthicidae	<i>Formicomus coeruleus</i>	11	4	9	61	85
Wood borer	Buprestidae	<i>Achmaedera</i> sp.	85	21	66	30	202
Predator	Cantharidae	<i>Cantharis</i> sp.1	4		15	6	25
Predator	Carabidae	<i>Acanthoscelis ruficornis</i>	5	3	11	1	20
Predator	Carabidae	<i>Anthia decemquattata</i>	16	3			19
Predator	Carabidae	<i>Caminara</i> sp.1	15	6	16	36	73
Predator	Carabidae	<i>Graphipterus trilineatus</i>	15			1	16
Predator	Carabidae	<i>Microlestia tabida</i>	125	3	82	12	222
Predator	Carabidae	<i>Passalidius fortipes</i>	1				1
Predator	Carabidae	Sp.1	18	36	4	12	70

Trophic guild	Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Phytophage	Chrysomelidae	<i>Leptinotarsa</i> sp.1	1		2	2	5
Predator	Cicindelinae	<i>Platydelia quadriguttata</i>	2				2
Predator	Cleridae	Sp.1	68	28	14	103	213
Predator	Coccinellidae	<i>Cheilomenes lunata</i>	10	9	30	69	118
Predator	Coccinellidae	<i>Harmonia axyridis</i>		5	1	29	35
Phytophage	Curculionidae	Sp.1	26	13	16	72	127
Phytophage	Curculionidae	Sp.2	5	1	4	1	11
Phytophage	Curculionidae	Sp.3	3	1	6	20	30
Phytophage	Curculionidae	Sp.4	4	9	13	4	30
Nectarivore	Dermestidae	<i>Dermestes maculatus</i>	10	7	9		26
Nectarivore	Dermestidae	<i>Dermestes</i> sp.	4	2	6	42	54
Phytophage	Elateridae	<i>Cardiotarsus acuminatus</i>	4	3	4		11
Predator	Lampyridae	<i>Lampyris disticollis</i>		2	1		3
Predator	Lampyridae	<i>Luciola</i> sp.	1		3		4
Nectarivore	Lycidae	Sp.1	1				1
Predator	Meloidae	<i>Decapotoma lunata</i>	4	1		2	7
Predator	Melyridae	<i>Apalochrus</i> sp.1	8	1	17		26
Nectarivore	Mordellidae	Sp.1			8	1	9
Nectarivore	Mordellidae	Sp.2	22	7	16	15	60
Saprophage	Nitidulidae	Sp.1	32	18	27	1	78
Saprophage	Nitidulidae	Sp.2	11	22	21	52	106
Saprophage	Scarabaeidae	<i>Schizonycha</i> sp.	2	2	2	1	7
Saprophage	Scarabaeidae	Sp.2	4	4	4	7	19
Saprophage	Scarabaeidae	<i>Aphodius</i> sp.	9		5	17	31
Saprophage	Scarabaeidae	Hopliini sp.	116	83	104	25	328
Saprophage	Scarabaeidae	<i>Scarabaeus rigosus</i>	38	11	18	1	68
Phytophage	Scarabidae	Sp.1	5	1	1	8	15
Phytophage	Silvanidae	<i>Oryzaephilus</i> sp.	1		2	8	11
Predator	Staphylinidae	<i>Paederinae</i> sp.	46	23	35	31	135
Saprophage	Tenebrionidae	<i>Psammodes striatus</i>				1	1
Saprophage	Tenebrionidae	Sp.1	78	17	8	8	111
Saprophage	Tenebrionidae	Sp.2	69	5	10	13	97
Saprophage	Tenebrionidae	Sp.3	1		3	1	5
Saprophage	Tenebrionidae	<i>Tenebrio</i> cf. <i>molitor</i>	15	7	3	7	32
Saprophage	Tenebrionidae	<i>Tenebrio</i> sp.1	44	45	23	38	150
Saprophage	Tenebrionidae	<i>Trigonopus</i> sp.1	34	16	21	67	138
Saprophage	Tenebrionidae	<i>Zophosis boei</i>	317	25	449	365	1156
Saprophage	Tenebrionidae	<i>Zophosis</i> sp.1	7		1		8
Saprophage	Collembola						
Saprophage	Sminthuridae	<i>Sminthurus viridis</i>	23	39	63	58	183
Saprophage	Crustacea						
Saprophage	Amphipoda	<i>Paramelita nigroculus</i>	119	21	89	6	235
Scavenger	Dermaptera						
Scavenger	Forficulidae	<i>Forficula senegalensis</i>	39	17	32	60	148
Saprophage	Diptera						
Saprophage	Anthomyiidae	<i>Anthomyia</i> sp.	33	37	26	11	107
Predator	Asilidae	<i>Daspletis</i> sp.	155	41	163	35	394
Parasitoid	Bombyliidae	<i>Exoprosopa</i> sp.	2	2			4
Saprophage	Calliphoridae	<i>Chrysomya chloropyga</i>	11		13	2	26
Saprophage	Calliphoridae	Sp.1	13	9		5	27
Saprophage	Calliphoridae	Sp.2	35	10	22	21	88
Phytophage	Cecidomyiidae	<i>Cecidomyia</i> sp.1	36	220	13		269
Saprophage	Heleomyzidae	<i>Helomyza picta</i>	2	10	4		16
Saprophage	Muscidae	Sp.1	13	8	24	6	51
Parasitoid	Pipunculidae	Sp.1	34	127	60	54	275
Saprophage	Sarcophagidae	Sp.1	3	4	1	7	15
Saprophage	Sciaridae	Sp.1	15	148	63	40	266
Predator	Syrphidae	Sp.1	1	5	12	1	19

Trophic guild	Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Parasitoid	Tachinidae	<i>Dejeania</i> sp.	30	16	18	2	66
Parasitoid	Tachinidae	<i>Gonia</i> sp.	3	1	2	5	11
Parasitoid	Tachinidae	Sp.1	14	1	5	8	28
Parasitoid	Tachinidae	Sp.2	8	7	3	4	22
Parasitoid	Tachinidae	Sp.3	8	6	16	8	38
Parasitoid	Tachinidae	Sp.4	3	2	4	7	16
Parasitoid	Therividae	<i>Therevid</i> sp.	5	3		2	10
Parasitoid	Tipulidae	<i>Tipula jocosae</i>	2	4	2	2	10
Hemiptera							
Predator	Anthocoridae	<i>Anthocoris</i> sp.1	8	6	18	21	53
Phytophage	Cercopidae	<i>Poophilus</i> sp.1	30	14	26	39	109
Phytophage	Cicadellidae	<i>Ciphalilus</i> sp.	2		3	1	6
Phytophage	Cicindelidae	<i>Cicindela quadriguttata</i>	3	1	17		21
Phytophage	Cydnidae	Sp.1	7	49		2	58
Phytophage	Miridae	<i>Deraeocoris</i> sp.	2	1	2	1	6
Phytophage	Pentatomidae	<i>Agonoscelis</i> sp.1	2		3	5	10
Phytophage	Pentatomidae	<i>Agonoscelis</i> sp.2	1				1
Phytophage	Pentatomidae	<i>Antestia lymphata</i>	3	1	5	5	14
Phytophage	Pentatomidae	<i>Bagrada hilaris</i>	2			23	25
Phytophage	Pentatomidae	<i>Coenomorpha</i> sp.	2		1		3
Phytophage	Pyrhocoridae	<i>Scantius forsteri</i>	15	11	12	4	42
Predator	Reduviidae	<i>Acanthaspis sanguinosa</i>	1				1
Predator	Reduviidae	<i>Cleptria rufipes</i>	49	17	39	10	115
Predator	Reduviidae	<i>Ectrichodia crux</i>	2		3	1	6
Predator	Reduviidae	<i>Holoptilus</i> sp.1			1		1
Predator	Reduviidae	<i>Oncocephalinae</i> sp.	1	27		2	30
Phytophage	Piesmatidae	Sp.1				2	2
Hymenoptera							
Nectarivore	Anthophoridae	<i>Thyreus delumbatus</i>	2	2	4	1	9
Nectarivore	Anthophoridae	<i>Xylocopa</i> sp.	1	1			2
Nectarivore	Apidae	<i>Apis mellifera</i>	12	2	13	28	55
Nectarivore	Apidae	<i>Meliponula</i> sp.1	5		4	26	35
Parasitoid	Braconidae	<i>Charops</i> sp.1		2	3		5
Parasitoid	Chalcididae	<i>Brachymeria kassalensis</i>	1	2	1	1	5
Nectarivore	Chrysididae	<i>Spintharina</i> sp.		3			3
Omnivore	Formicidae	<i>Camponotus maculatus</i>	96		14	5	115
Omnivore	Formicidae	<i>Camponotus</i> sp.1	28	4	17	2	51
Omnivore	Formicidae	<i>Camponotus</i> sp.2	758	108	588	253	1707
Omnivore	Formicidae	<i>Camponotus</i> sp.3	21	30	9		60
Omnivore	Formicidae	<i>Crematogaster peringueyi</i>	300	132	228	18	678
Omnivore	Formicidae	<i>Dorylus helvolus</i>		1			1
Omnivore	Formicidae	<i>Linepithema humile</i>	359	99	366	467	1291
Omnivore	Formicidae	<i>Messor capensis</i>	62	27	34	14	137
Omnivore	Formicidae	Sp.1	16	15	8	19	58
Omnivore	Formicidae	Sp.2	4				4
Omnivore	Formicidae	Sp.3	135	12	22	18	187
Omnivore	Formicidae	<i>Tetramorium capense</i>	241	157	317	159	874
Nectarivore	Halictidae	<i>Nomia amabilis</i>	54	15	18		87
Nectarivore	Halictidae	Sp.1	36	21	40	118	215
Nectarivore	Halictidae	Sp.2	5			2	7
Nectarivore	Halictidae	Sp.3		2	2	2	6
Nectarivore	Masaridae	<i>Ceramius</i> sp.1			1		1
Nectarivore	Melittidae	Sp.1	29	1	12		42
Nectarivore	Melittidae	Sp.2	10	4	7	12	33
Nectarivore	Mutillidae	Sp.1	7	1	8	17	33
Nectarivore	Mutillidae	Sp.2	9	1	4	4	18

Trophic guild	Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Parasitoid	Pompilidae	Sp.1	43	15	29	14	101
Parasitoid	Pompilidae	Sp.2	8	3	14	5	30
Parasitoid	Scoliidae	<i>Campsomeriella</i> sp.1	14	3	11		28
Predator	Sphecidae	Sp.1	6	2	4	6	18
Predator	Sphecidae	Sp.2	56	39	44	42	181
Predator	Sphecidae	Sp.3	6		3	6	15
Predator	Sphecidae	Sp.4			1		1
Predator	Sphecidae	<i>Sphex bonaspei</i>	41	4	36	3	84
Predator	Sphecidae	<i>Sphex tydei</i>	12		5	1	18
Nectarivore	Tiphiidae	Sp.1	4	7	5	4	20
Isopoda							
Saprophage	Porcellionidae	<i>Porcello</i> sp.	309	569	336	214	1428
Isoptera							
Saprophage	Hodotermitidae	<i>Microhodotermes viator</i>	776	298	1366	118	2558
Lepidoptera							
Phytophage	Arctiidae	<i>Rhodogastria amasis</i>	6	19	2	4	31
Phytophage	Geometridae	Sp.1	6		4	4	14
Phytophage	Lasiocampidae	<i>Eutricha capensis</i>			4	1	5
Phytophage	Lycaenidae	<i>Aloeides</i> sp.	1	2	3	12	18
Phytophage	Noctuidae	<i>Helicoverpa armigera</i>	18	51	6	32	107
Phytophage	Noctuidae	Sp.1		2	64	17	83
Phytophage	Noctuidae	Sp.2	6	12	5	49	72
Phytophage	Nymphalidae	<i>Cynthia cardui</i>	21	32	13	52	118
Phytophage	Psychidae	Sp.1	2		1		3
Phytophage	Pyrilidae	Sp.1	4	8	3	4	19
Phytophage	Sphingidae	<i>Coelonia fulvinotata</i>				4	4
Phytophage	Tineidae	Sp.1	9	3		1	13
Phytophage	Tortricidae	Sp.1		3			3
Lithobiida							
Saprophage	Lithobiidae	<i>Lithobius</i> sp.1	26	4	20	16	66
Saprophage	Lithobiidae	<i>Lithobius</i> sp.2	9	10	12	49	80
Mantodea							
Predator	Hymenopodidae	<i>Harpagomantis tricolor</i>	5		1		6
Predator	Mantidae	<i>Polyspilota aeruginosa</i>				1	1
Predator	Mantidae	Sp.1	3	4	11	5	23
Predator	Mantidae	Sp.2	3		13	7	23
Predator	Mantidae	Sp.3	14	7	7		28
Mecoptera							
Predator	Bittacidae	<i>Anomalobittacus gracilipes</i>	17	44	9	4	74
Neuroptera							
Omnivore	Nemopteridae	<i>Laurhervasia setacea</i>	2		2	1	5
Lithobiomorpha							
Predator	Peripatopsidae	<i>Peripatopsis</i> sp.	7	6	3		16
Opiliones							
Predator	Phalangiiidae	<i>Phalangium</i> sp.1	31	14	116	6	167
Orthoptera							
Omnivore	Gryllidae	<i>Cophogryllus</i> sp.	63	24	52	22	161
Omnivore	Gryllidae	<i>Gryllus bimaculatus</i>	196	93	119	92	500
Scolopendromorpha							
Predator	Scolopendromorphae	<i>Scolopendra cingulata</i>	13	2	3	24	42
Scorpiones							
Predator	Buthidae	<i>Uroplectes lineatus</i>	21	1	1	1	24
Predator	Scorpionidae	<i>Opisthophthalmus macer</i>	12	22	22		56
Solifugae							
Predator	Eremobatidae	<i>Eremobates</i> sp.1	2		4		6
Predator	Galeodidae	<i>Galeodes</i> sp.1	47	16	26	40	129

Trophic guild	Order/Family	Species	Fynbos	IATs	CIATs	Vineyard	Combined
Predator	Galeodidae	<i>Galeodes</i> sp.2	45	4	38	59	146
	Spirobolida						
Detritivore	Pachybolidae	<i>Centrobolus</i> sp.1	67	113	122	85	387
Detritivore	Pachybolidae	<i>Centrobolus</i> sp.2	406	734	651	417	2208
Total sampled individuals			7746	4725	7563	5221	25255
Total sampled spp.			179	153	169	159	198
%Total sampled spp.			90	77	85	80	

Appendix 5.2 Nested analysis of variances (ANOVA's) among fynbos, invasive alien trees (IATs), cleared invasive alien trees (CIATs), and vineyard sites in terms of species richness, species abundance, soil compaction, percentage soil moisture, and leaf litter depth. * indicates that the mean difference is significant at the 0.05 level.

Dependent variable	(I) Vegetation	(J) Vegetation	(I-J) Mean difference	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Species richness	Fynbos	IATs	6.489*	0.685	0.000	4.67	8.3
		CIATs	-0.068	0.7	1.000	-1.92	1.78
		Vineyard	4.007*	0.682	0.000	2.2	5.81
	IATs	Fynbos	-6.489*	0.685	0.000	-8.3	-4.67
		CIATs	-6.558*	0.725	0.000	-8.48	-4.64
		Vineyard	-2.483*	0.709	0.003	-4.36	-0.61
	CIATs	Fynbos	0.068	0.7	1.000	-1.78	1.92
		IATs	6.558*	0.725	0.000	4.64	8.48
		Vineyard	4.075*	0.723	0.000	2.16	5.99
	Vineyard	Fynbos	-4.007*	0.682	0.000	-5.81	-2.2
		IATs	2.483*	0.709	0.003	0.61	4.36
		CIATs	-4.075*	0.723	0.000	-5.99	-2.16
Abundance	Fynbos	IATs	15.316*	3.866	0.001	5.08	25.55
		CIATs	-12.345*	3.946	0.011	-22.8	-1.89
		Vineyard	11.984*	3.849	0.012	1.79	22.18
	IATs	Fynbos	-15.316*	3.866	0.001	-25.55	-5.08
		CIATs	-27.660*	4.092	0.000	-38.5	-16.82
		Vineyard	-3.332	3.998	1.000	-13.92	7.26
	CIATs	Fynbos	12.345*	3.946	0.011	1.89	22.8
		IATs	27.660*	4.092	0.000	16.82	38.5
		Vineyard	24.329*	4.076	0.000	13.53	35.12
	Vineyard	Fynbos	-11.984*	3.849	0.012	-22.18	-1.79
		IATs	3.332	3.998	1.000	-7.26	13.92
		CIATs	-24.329*	4.076	0.000	-35.12	-13.53
Soil compaction (Kg/m ³)	Fynbos	IATs	65.309*	17.298	0.001	19.49	111.12
		CIATs	-6.351	17.659	1.000	-53.12	40.42
		Vineyard	-70.850*	17.223	0.000	-116.47	-25.23
	IATs	Fynbos	-65.309*	17.298	0.001	-111.12	-19.49
		CIATs	-71.660*	18.308	0.001	-120.15	-23.17
		Vineyard	-136.160*	17.889	0.000	-183.54	-88.78
	CIATs	Fynbos	6.351	17.659	1.000	-40.42	53.12
		IATs	71.660*	18.308	0.001	23.17	120.15
		Vineyard	-64.500*	18.238	0.003	-112.81	-16.19
	Vineyard	Fynbos	70.850*	17.223	0.000	25.23	116.47
		IATs	136.160*	17.889	0.000	88.78	183.54
		CIATs	64.500*	18.238	0.003	16.19	112.81
Percentage Soil moisture	Fynbos	IATs	0.2703	0.3735	1.000	-0.719	1.26
		CIATs	0.9831	0.3813	0.061	-0.027	1.993
		Vineyard	1.3995*	0.3719	0.001	0.414	2.385

Dependent variable	(I) Vegetation	(J) Vegetation	(I-J) Mean difference	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Percentage Soil moisture	IATs	Fynbos	-0.2703	0.3735	1.000	-1.26	0.719
		CIATs	0.7129	0.3953	0.432	-0.334	1.76
		Vineyard	1.1292*	0.3863	0.022	0.106	2.152
	CIATs	Fynbos	-0.9831	0.3813	0.061	-1.993	0.027
		IATs	-0.7129	0.3953	0.432	-1.76	0.334
		Vineyard	0.4164	0.3938	1.000	-0.627	1.459
	Vineyard	Fynbos	-1.3995*	0.3719	0.001	-2.385	-0.414
		IATs	-1.1292*	0.3863	0.022	-2.152	-0.106
		CIATs	-0.4164	0.3938	1.000	-1.459	0.627
Leaf litter depth (mm)	Fynbos	IATs	-5.226*	0.605	0.000	-6.83	-3.62
		CIATs	1.920*	0.618	0.012	0.28	3.56
		Vineyard	12.834*	0.603	0.000	11.24	14.43
	IATs	Fynbos	5.226*	0.605	0.000	3.62	6.83
		CIATs	7.146*	0.641	0.000	5.45	8.84
		Vineyard	18.060*	0.626	0.000	16.4	19.72
	CIATs	Fynbos	-1.920*	0.618	0.012	-3.56	-0.28
		IATs	-7.146*	0.641	0.000	-8.84	-5.45
		Vineyard	10.914*	0.638	0.000	9.22	12.6
	Vineyard	Fynbos	-12.834*	0.603	0.000	-14.43	-11.24
		IATs	-18.060*	0.626	0.000	-19.72	-16.4
		CIATs	-10.914*	0.638	0.000	-12.6	-9.22

* indicates that the mean difference is significant at the 0.05 level

Chapter 6 – Effects of Fire on Surface-Active Invertebrates

INTRODUCTION

Most Cape fynbos plant species survive fires either by regenerating from seed or by resprouting (Botha, 1990). Fire may influence plant growth and foliar chemistry by enhancing nutrient concentrations in the soil (Prieto-Fernandez et al., 1993). Fire is often excluded from agricultural areas and nature reserves, and species that are dependent on fire-induced regeneration from seed may become locally extinct if the interval between burns is longer than required (Bond, 1980). Generally, fynbos is reliant on regular burning for biodiversity maintenance. Therefore, disruption of such natural ecological processes (i.e. fire) can have a significant impact on biodiversity.

It is often assumed that fauna will naturally recolonize burned areas from the surrounding matrix, as successional development of regrowing vegetation proceeds (Nakamura et al., 2008). However, there is a dearth of studies investigating the recolonization of vegetation by surface-active invertebrates following fire (Scott et al., 2001).

It is likely that persistence of some invertebrate species in fynbos fragments may depend on successful recruitment from outside the fragment, especially after a major disturbance such as fire (Armstrong and Hensbergen, 1996). Importance of dead plant material as substrate for a wide range of organisms has been emphasized worldwide (Harmon et al., 1986; Siitonen, 2001; Grove, 2002), and are continuously created through natural disturbances, of which fire is one of the most important (Johnson et al., 1998; Niklasson and Granström, 2000). Therefore, an inappropriate fire regime can cause changes in species composition or cause local extinction of both fauna and flora.

Many invertebrates are influenced by fire (Wikars, 1997; Hyvärinen et al., 2005), and exposure (Lindhe et al., 2005). As a result, movement (Hanski, 1998) is the main invertebrate response pattern likely to occur, with individuals moving from unsuitable to suitable habitats. However, they often have to traverse patches of unsuitable habitat,

which may include burned areas. Indeed, it is important to understand the effect of fire on invertebrates, as they are crucial to the stability, regulation and functioning of many ecosystems (Gill, 1981; Flinn et al., 1983).

Fire is an important factor in the growth and maintenance of fynbos in the CFR. Moreover, surface-active invertebrates and other insect assemblages in the region are expected to have highly evolved adaptations to fire (Whelan and Main, 1979). Therefore, regular fire is unlikely to constitute major adverse factor in fire-adapted assemblages, but rather an essential feature for the long-term maintenance of these assemblages.

Still little is known on the surface-active invertebrate assemblages in the CFR, and no research has yet been done on the interaction between invertebrates, fire and boundaries between different biotopes in the CFR. This is particularly significant because the CFR, especially at lower elevations, has been extensively modified by human activities. To address this dearth of information, the aim here is to investigate the response of surface-active invertebrate populations to fire, especially at landscape fragment boundaries. Specifically, I ask whether there are changes in species richness composition and abundance after fire in both natural fynbos and invasive alien trees (IATs), with the null hypothesis that fire has no effect on the invertebrate assemblages. Specifically, I ask which taxa are affected most and to establish the initial recovery rates. This information assists with conservation management by establishing the impact of fire, as well as determining how invertebrate assemblages recover. This information will assist with conservation management by establishing the impact of fire as well as determining how invertebrates' assemblages recover.

STUDY SITES

The study was conducted in Hottentots Holland Nature Reserve (HHNR) (34.0824 E, 19.03658 S) (Figure 6.1), and on Waterford Wine Farm (WWF) (34.0066 E, 18.87529 S) (Figure 6.2) in the CFR, South Africa. The area has a mediterranean climate with warm, dry summers and cool, wet winters, and the soil is derived from granite and quartzite, moderately drained through varying percentages of cobbles and boulders in the profile. This study was opportunistic in that an array of pitfall traps were deployed for investigating ecotone effect per se, but the experimental layout was subject to

accidental fires in some IATs and fynbos areas. However, by chance, there was sufficient replication to determine the effects of fire on the invertebrates at the ecotone.

The dominant vegetation type at the HHNR study site was Proteoid fynbos, consisting of 1-2 m tall, open to closed overstorey (Table 6.1). Adjacent to the native fynbos was *Pinus radiata* plantation. Moreover, the remaining unburned fynbos site within the HHNR was selected to serve as control site. The dominant vegetation type at the WWF study site was pine plantation, consisting of 50-90 m tall and closed overstorey adjacent to a vineyard. However, the fire in WWF study site was of low to moderate intensity, burning in the forest understorey, consuming dead fuels and small trees, while leaving most fire-resistant adult trees alive and healthy.

Table 6.1 Dominant native fynbos species and their regeneration mode before fire in HHNR (adapted from Smith and Richardson, 1990).

Family	Species name	Growth form	Regeneration mode
Proteaceae	<i>Protea nitida</i>	Tall shrub	reseeder/sprouter
	<i>Protea repens</i>	Tall shrub	reseeder
	<i>Leucadendron salignum</i>	Low shrub	resprouter
Restionaceae	<i>Ischyrolepis gaudichaudianus</i>	Graminoid	resprouter
Iridaceae	<i>Watsonia borbonica</i>	Geophyte	resprouter

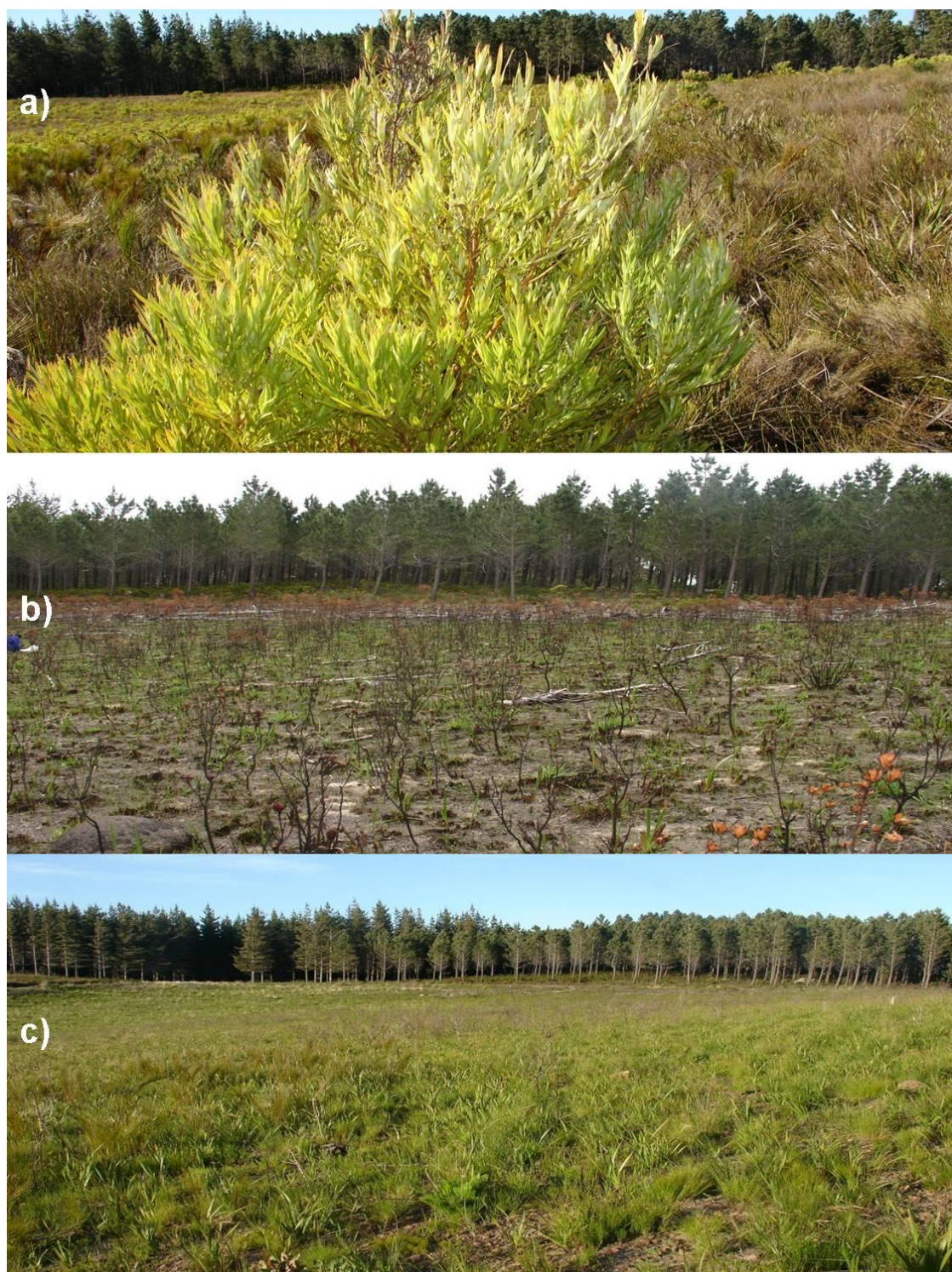


Figure 6.1 Hottentots Holland Nature Reserve study area: a) indigenous fynbos vegetation, 2 weeks before fire; b) indigenous fynbos vegetation, 3 months after fire; c) indigenous fynbos vegetation, 6 months after fire.



Figure 6.2 Waterford Wine farm study area: a) Pine plantation, 2 weeks before fire; b) Pine plantation, 3 months after fire; c) Pine plantation, 6 months after fire.

MATERIALS AND METHODS

Diversity of surface-active invertebrates in response to fire within fynbos and IATs were studied here using pitfall traps. Generally, surface-active invertebrate taxa are easily collected in pitfall traps and thus allow for standardized sampling method for comparative analyses (Luff, 1975; Margules and Usher, 1981; Woodcock, 2005).

Multiple sampling stations were set up along transects running orthogonal to transitions between vineyard-IATs (later burned) and IATs-native fynbos (later burned) vegetation fragments. A sampling station is defined as a specific pitfall trap location along a transect (at specific distances from the adjacent vegetation). A schematic representation of the experimental design showing pitfall trap placement along a transect is given in Figure 6.3. Transects have been widely used for studying insect assemblages and associated environmental variables (Whittaker, 1972; Jonhston et al., 1992; Procheş and Cowling, 2006). A \log^2 scale was used for sampling stations along three transects across each vegetation boundary/ecotone (Samways et al., 2010). For each transect, sampling stations were selected across the ecotone, which then extended 128 meters into both adjoining habitat fragments. Thus, an individual transect was 256 m long, covering many vegetation fragments, with multiple sampling stations within each vegetation fragment at different distances from the fragment boundary.

Each pitfall trap was a 500 ml plastic honey jar, containing a replaceable paper cup, 8 cm wide and 12 cm deep. Each trap was one third filled with 70% Ethanediol (antifreeze fluid). Specimens were washed in water and placed in 70% ethanol. Permanent pitfall traps were established at all sampling stations, which remained closed during non-sampling periods and opened during sampling period for five days to allow invertebrate trapping before collection days. Sampling in vineyards was done within vine rows to minimize impact of disturbance by tractors and farm workers. All pitfall traps were set up at least three months prior initial sampling to eliminate any 'digging-in effect'. Traps were operational only when there was no rain for at least five consecutive days during the two-year sampling period from August 2006 to January 2008, two weeks before an accidental fire, and then three months and six months after fire.

The collected surface-active invertebrates were sorted and placed into morphospecies and identified to species where possible (Appendix 6.1-Appendix 6.3). Voucher

specimens of each invertebrate species are held in the Entomology Museum of Stellenbosch University. Spider specimens were deposited at the National Collection of Arachnida in the National Museum in Pretoria. Identification was by Picker et al. (2004), Leroy and Leroy (2003), McGavin (2000), Preston-Mafham and Preston-Mafham (2005); and the specialists, Prof. Henk Geertsema and Mr. Patrick Reavel (Stellenbosch University) assisted with the identification of insects. Dr. Ansie Dippenaar-Schoeman of ARC-Plant Protection Research Institute assisted with spider identification.

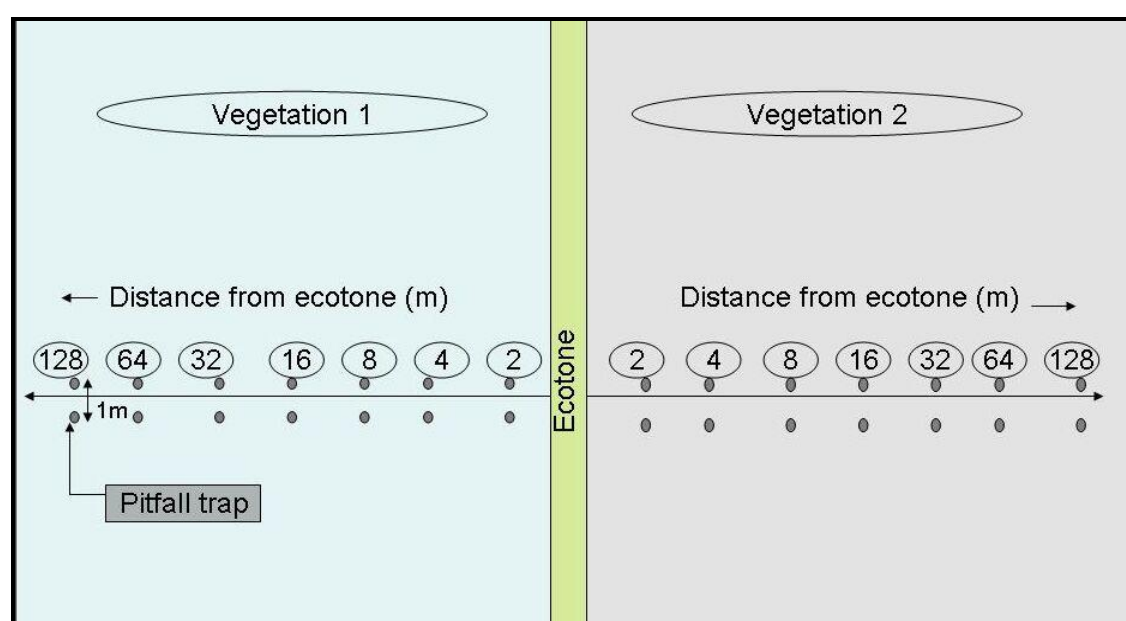


Figure 6.3 Schematic representation of the experimental design showing one pair of vegetation types, and pitfall trap distance placement within transects. Seven pitfall trap locations were placed on either side of the ecotone. (N.B: Drawing not to scale).

DATA ANALYSIS

Samples collected two weeks before fire, three and six months after fire were compared for their species diversity in IATs and fynbos vegetation separately. This was done using one-way analyses of variance (ANOVA) on the species, on log-transformed abundance data. Multiple comparison of the means was done using the Bonferroni methods (Legendre and Legendre, 1998). The SPSS 15.0 for Windows programme was used for ANOVA (SPSS Inc., 2006). Scatter plots with species data points were connected by

smoothed lines to reflect invertebrate variations along ecotone/vegetation boundaries. Species richness and abundance scatter plots were produced separately for vineyard/IATs and IATs/fynbos ecotones. This showed the responses of invertebrates towards ecotones and adjacent vegetation. Invertebrate trophic guild composition for fynbos and IATs, based on species richness, were also compared using histograms. Species accumulation curves were used to establish sufficiency of sampling using EstimateS version 8.0.0 with samples randomized 50 times (Colwell, 2006). The Abundance-based Coverage Estimator (ACE Mean) of species richness was calculated using EstimateS (Colwell, 2006), for the unburned fynbos. Curves were plotted for unburned and burned native fynbos and for the ACE estimated richness.

Cluster, Multi-Dimensional Scaling (MDS) and Analysis of Similarity (ANOSIM) analyses were also carried using PRIMER v5 (Clarke and Gorley, 2006) to detect trends in arthropod similarity. MDS was also performed to compare arthropod assemblage composition in burned fynbos vegetation and unburned (control) fynbos. In MDS, the dendrograms were comprised of group-average linking on Bray-Curtis species similarities from standardised abundance data. In the ordination, distances between samples attempt to match the corresponding dissimilarities in invertebrate assemblage composition: nearby samples have very similar assemblages, while samples that were far apart have few species in common, or the same species at very different level of abundance. In the case of ANOSIM, R-value is approximately zero if the null hypothesis is true, indicating that similarities between and within sampling period will be the same on average (Clarke and Warwick 2001).

In the case of clear grouping of samples, individual species contributions to the separation of the groups were examined using SIMPER, a component of PRIMER v5. As a result of large number of species sampled, a more automatic, analytical procedure (SIMPER procedure) for identifying influential species or discriminating species was more important. Average invertebrate dissimilarity between inter-sampling period samples (e.g. every sample collected two weeks before fire paired with every sample collected six months later, following fire) and then, this average was broken down into separate contributions from each species to average dissimilarity. If average dissimilarity of a particular species was large, and the ratio Diss/SD (dissimilarity/standard deviation) was also large, then a particular species not only

contributed much to the dissimilarity between samples, but it also did so consistently in inter-comparisons of all samples in the different sampling periods, thus being considered a good discriminating species. The null hypothesis for ANOSIM test statistics was that there were no significant difference in invertebrate assemblage composition between the samples collected two weeks before fire, three months after fire, and six months after fire.

RESULTS

Total number of recorded invertebrates

For the whole study, a total of 2876 individuals, in 135 species representing 84 families were recorded from IATs-fynbos vegetation whereas 2247 individuals, in 129 species representing 82 families were recorded from vineyard-IATs. The taxonomic group breakdown of these is given in Appendices 6.1 and 6.2.

Invertebrate responses to burned invasive alien trees

IATs sites showed no significant differences in overall mean invertebrate abundance ($df = 2$, $f = 0.525$, $p = 0.666$) and species richness ($df = 503$, $f = 2.298$, $p = 0.133$) before and after the accidental fire (Table 6.2). Nevertheless, certain species were recorded before fire, which decreased in abundance immediately after fire, and then increased several months later (Figure 6.4). Generally, all invertebrate trophic guilds were not significantly different during sampling periods; 2 weeks before fire ($df = 6$, $f = 1.483$, $p = 0.209$), 3 months after fire ($df = 7$, $f = 0.408$, $p = 0.864$) and 6 months after fire ($df = 6$, $f = 2.660$, $p = 0.45$) (Figure 6.5).

Table 6.2 Species abundance means, standard deviation and standard error (Std. Error.) for invasive alien trees sites. N = Number of samples.

Sampling period	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Min.	Max.
					Lower Bound	Upper Bound		
2 Weeks before fire	46	3.87	6.622	0.976	1.90	5.84	1	30
3 months after fire	46	2.42	2.448	0.500	1.38	3.45	0	12
6 months after fire	46	4.28	5.926	1.047	2.14	6.42	0	28
<i>Total</i>	<i>138</i>	<i>3.67</i>	<i>5.690</i>	<i>0.517</i>	<i>2.65</i>	<i>4.69</i>	<i>1</i>	<i>30</i>

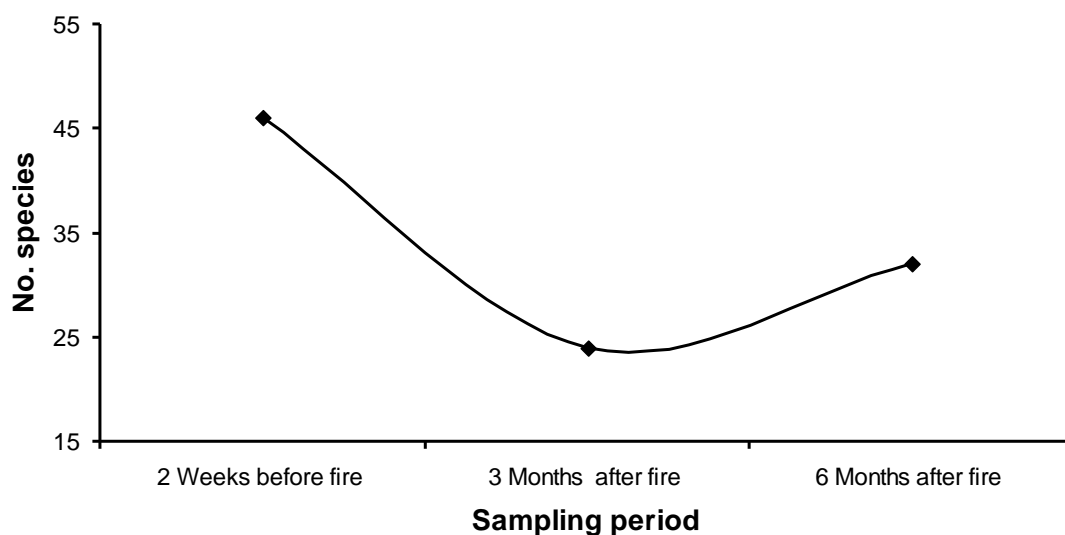


Figure 6.4 Number of species recorded from IATs during different sampling periods.

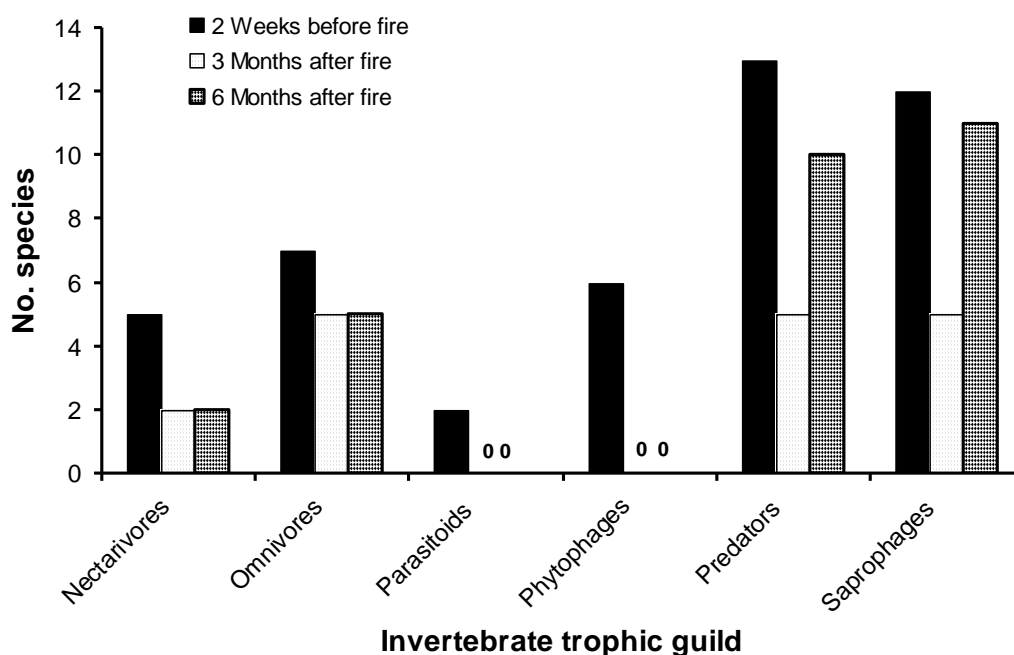


Figure 6.5 Total number of species recorded from the IATs, two weeks before fire; b) Pine plantation, 3 months after fire; c) Pine plantation, 6 months after fire.

Significance of vineyard adjacent to burned invasive alien trees

The vineyard had higher invertebrate species richness than the adjacent IATs (Figure 6.6). However, more species were also recorded near the boundary between the two adjoining habitat fragments and decreased with increasing distance from the ecotone into IATs (Figure 6.6). Dominant taxa were spiders, beetles, and hymenopterans on either side of the ecotone (see Chapter 4). Species richness was lower in the vineyard site before fire than that recorded three months after fire (Figure 6.6). However, there was a sharp decrease in species richness within vineyards, six months later. Moreover, species richness continued to decrease in the IATs, six months after the fire (Figure 6.6). Invertebrate abundance was also low in either side of the ecotone before the fire (Figure 6.7), but gradually increased following it. Surprisingly, abundance was more comparable on either side of the ecotone before fire but shifted toward the burned site six months later.

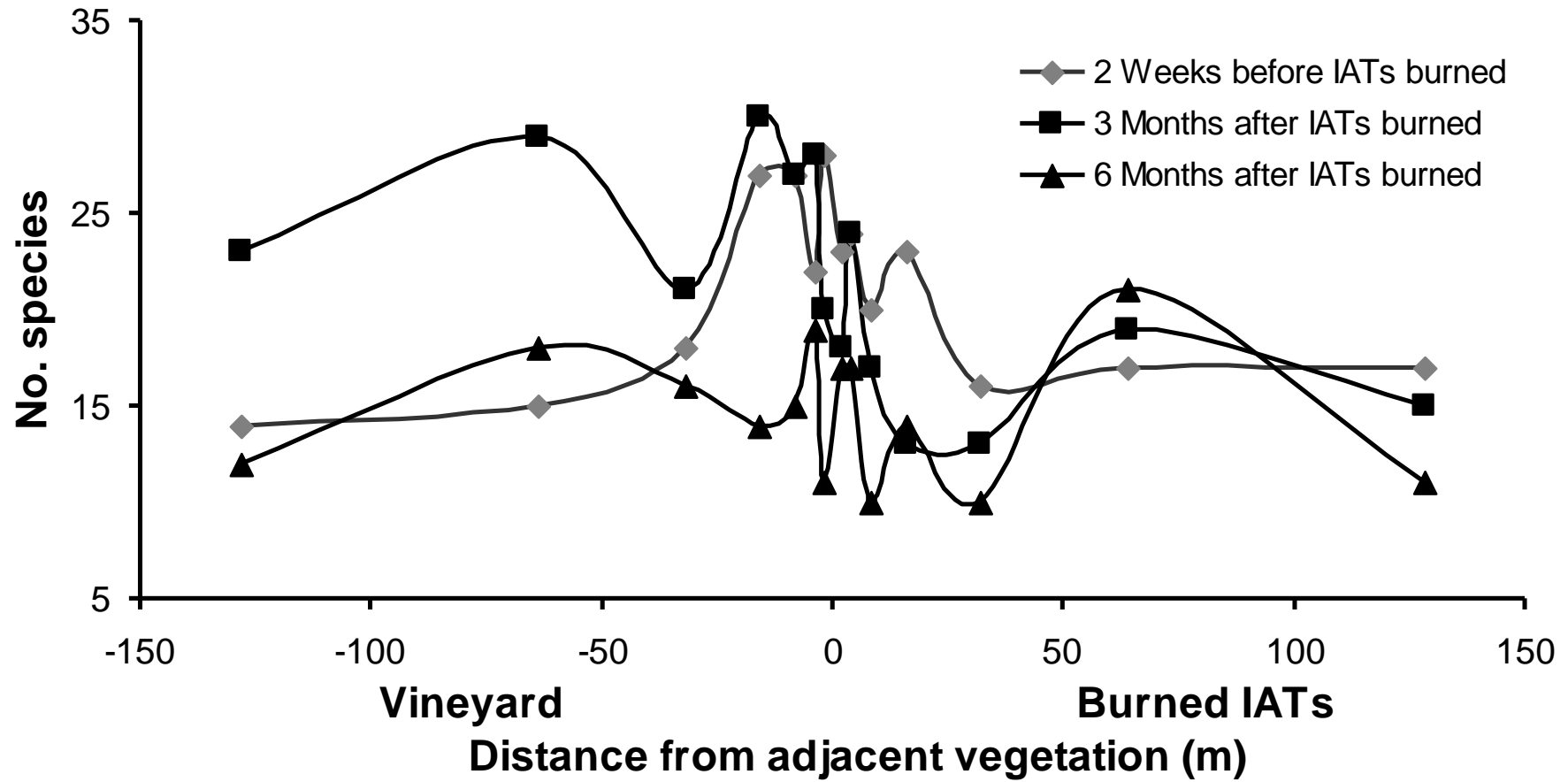


Figure 6.6 Species richness variations along vineyard/burned invasive alien trees (IATs) ecotone. 0 m = ecotone.

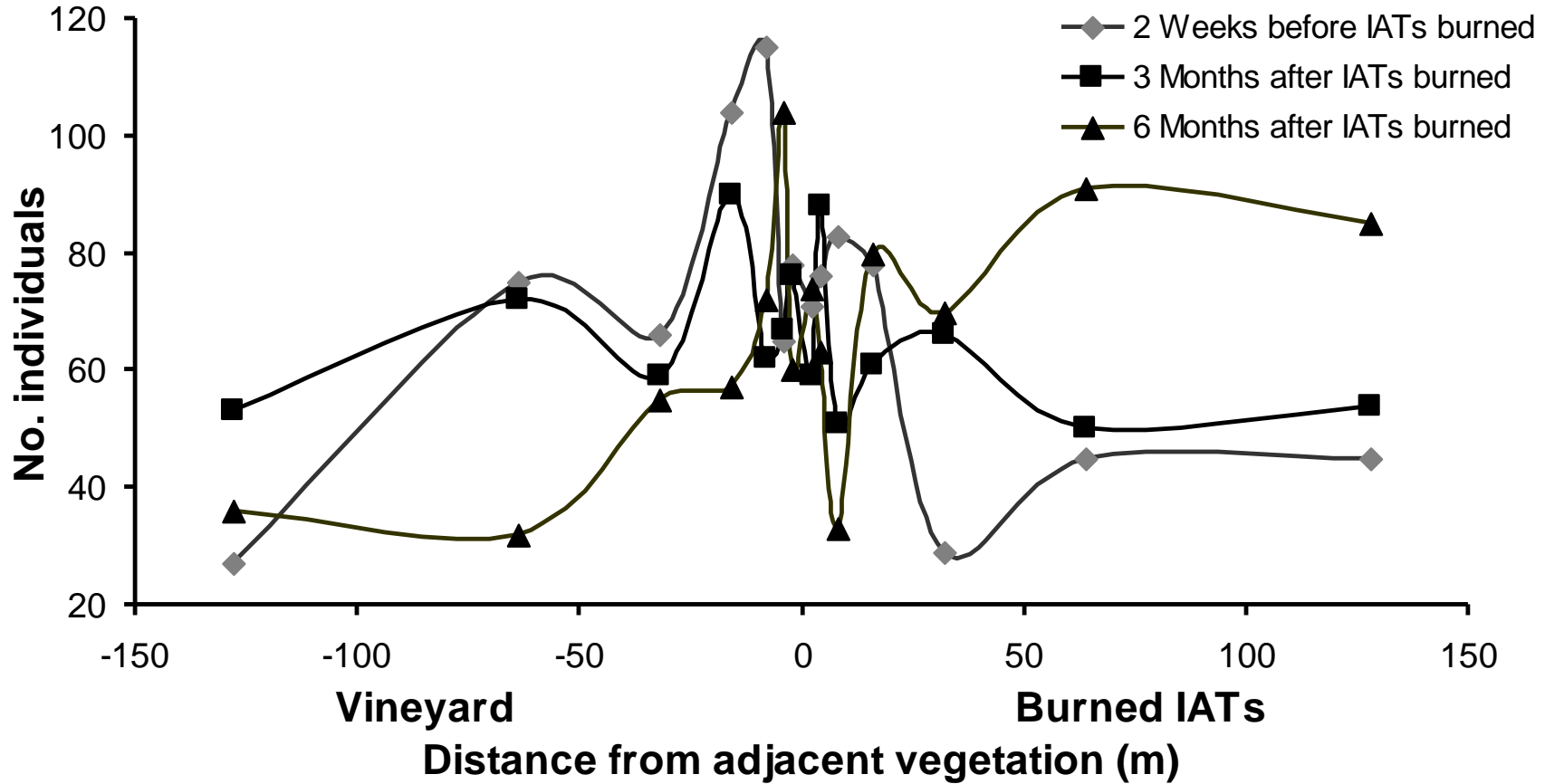


Figure 6.7 Species abundance variations along vineyard/burned invasive alien trees (IATs) ecotone. 0 m = ecotone.

Invertebrate assemblage composition in IATs

The Cluster (Figure 6.8) and MDS (Figure 6.9) analyses highlighted the potential for similarities in invertebrate assemblages before and after fire in IATs. Figure 6.8 shows a cluster analysis on samples based on invertebrate assemblage composition. Figure 6.9 shows the 2-dimensional MDS plot of the same species composition similarities as indicated in Figure 6.8. Samples grouped according to invertebrate assemblage composition with high percentage similarities. Only one sample (3MAF8) was distinct from the rest (i.e. collected from IATs before and after fire) (Figure 6.9). This sample was located at the core of IATs and had only one individual of Mutillidae species, whereas others had multiple number of species and abundance which was higher than one.

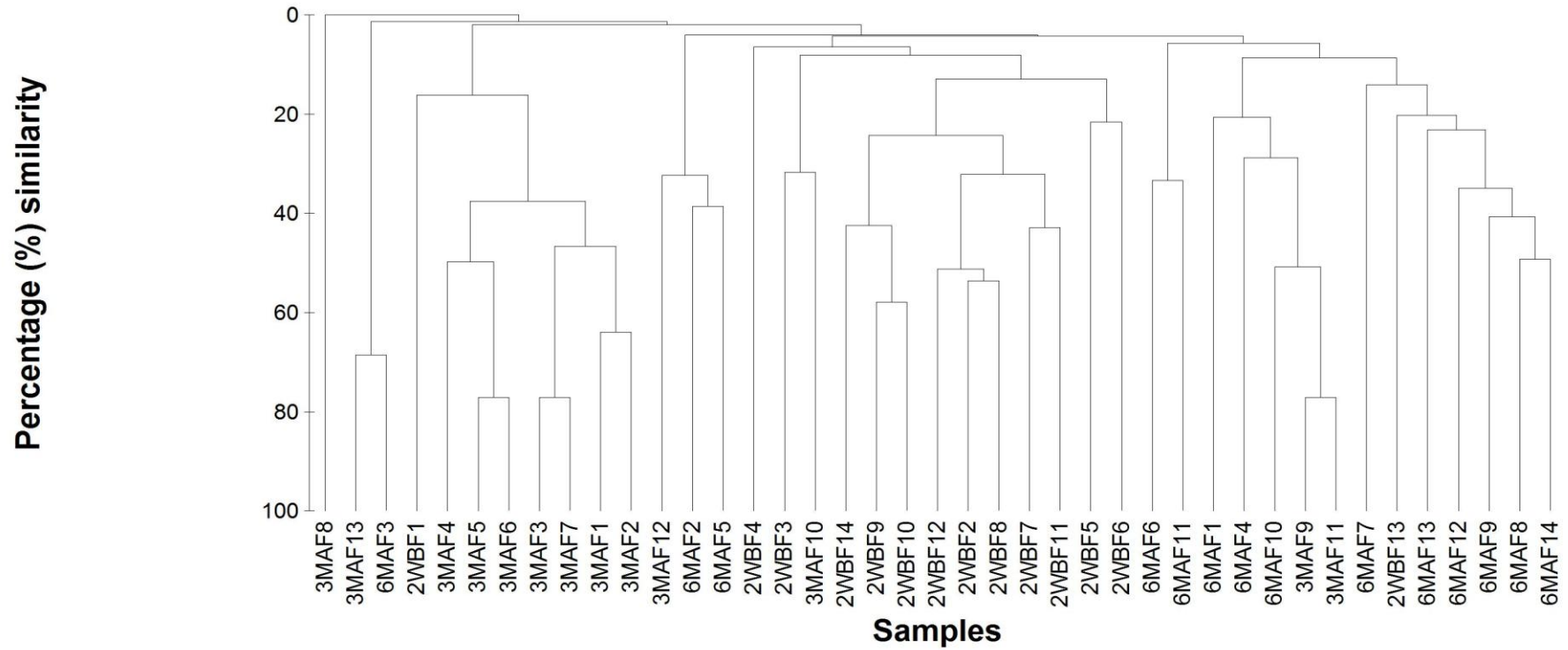


Figure 6.8 Classification tree of samples in terms of invertebrate assemblages using group-average linking on Bray-Curtis species similarities in IATs. 2WBF = Two weeks before fire, 3MAF = Three months after fire, 6MAF = Six months after fire.

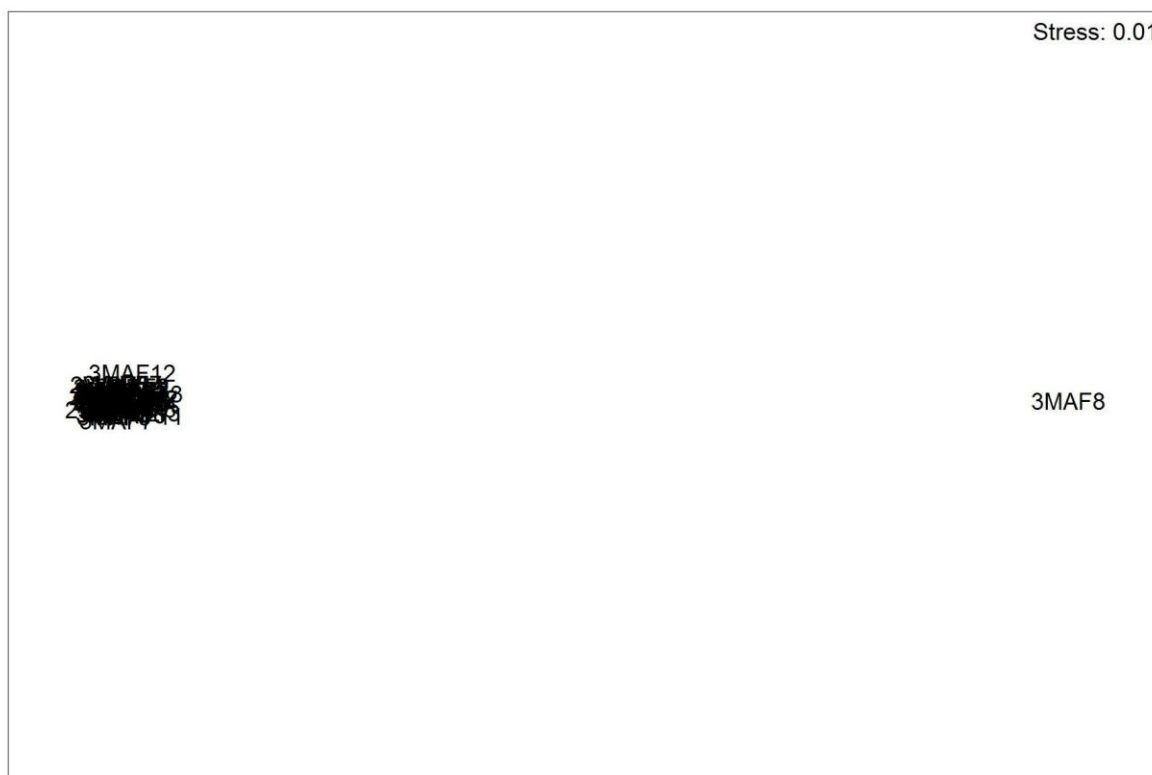


Figure 6.9 Multi-dimensional scaling (MDS) analysis of Bray-Curtis similarity between invertebrate samples from different sampling periods (2MBF: two months before fire; 3MAF: three months after fire and 6MAF: six months after fire) in the IATs.

Testing the significance of invertebrate assemblage composition differences before and after fire in IATs

The null hypothesis for these ANOSIM test statistics was that there were no significant differences in invertebrate assemblage composition between the samples collected two weeks before fire, three months after fire and six months after fire. The global test of the null hypothesis based on invertebrates sampled during different times relative to the fire was accepted ($R = 0.02$, $P < 56\%$). R is approximately zero when the null hypothesis is true, indicating that similarities between and within samples will be the same on average (Clarke and Warwick 2001).

Invertebrate responses to burned fynbos

Fynbos sites supported significantly higher species richness ($df = 42$, $f = 8.316$, $p = 0.006$) and abundance ($df = 42$, $f = 10.331$, $p = 0.002$) before fire than after it (Figure 6.10). There were consistent species richness and abundance variations between different individual samples, and hence similar/equal standard error bars (Figure 6.10).

Moreover, invertebrate groups were significantly different at different times relative to the fire ($df = 42$, $f = 11.614$, $p = 0.000$). Nevertheless, some invertebrate groups were recorded consistently throughout the study period, while others were not sampled until at least six months after the fire (Figure 6.11). Species accumulation curves indicated higher species richness after fire than in the unburned fynbos (Figure 6.12).

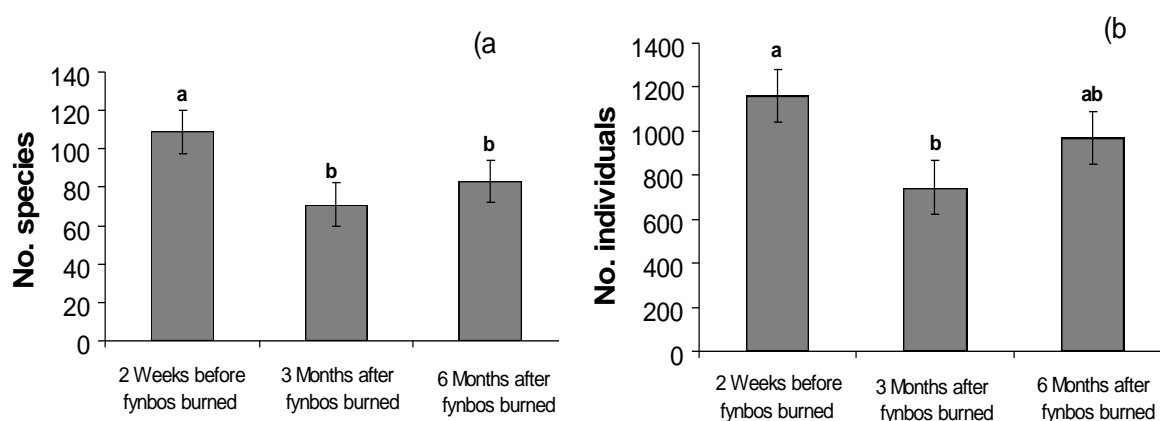


Figure 6.10 Species richness (a) and abundance (b) recorded from fynbos sites at different times in relation to a fire event. Vertical bars denote \pm standard errors. Different letters above vertical bars indicates a significant difference, whereas similar letters indicates non-significant difference.

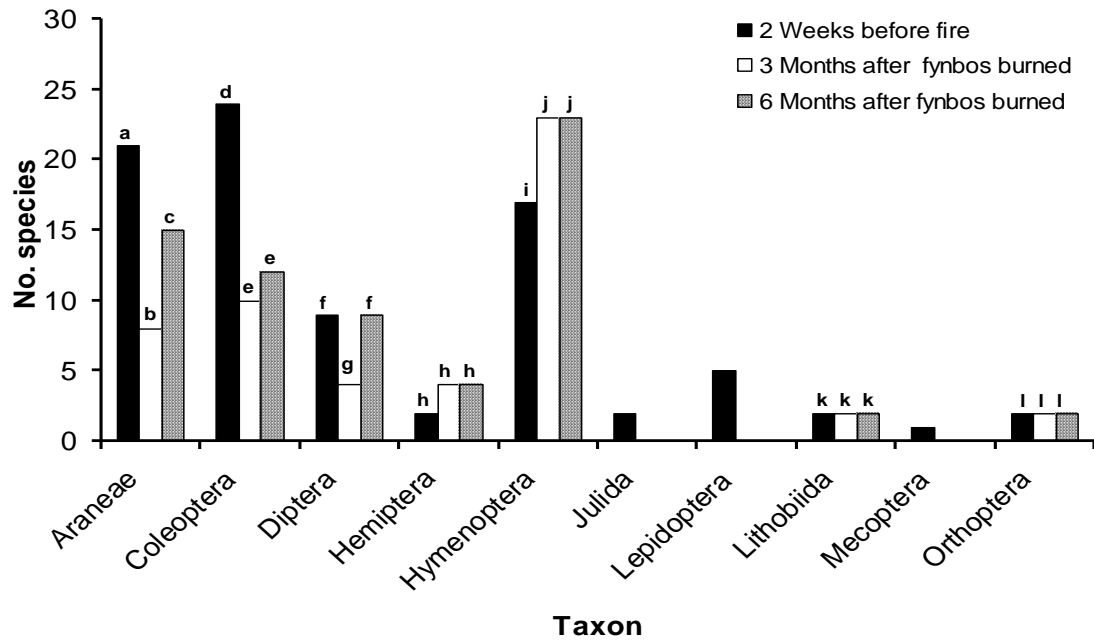


Figure 6.11 Invertebrate groups represented before and after fire within the fynbos. Different letters above columns indicates a significant difference, whereas similar letters indicates non-significant difference.

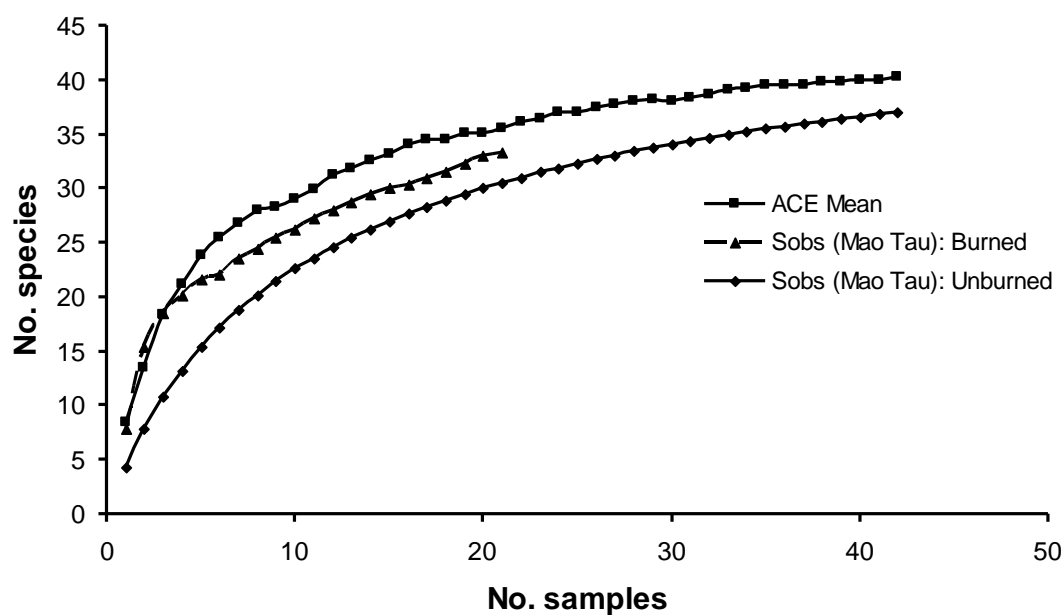


Figure 6.12 Invertebrate species accumulation curves for fynbos sites: estimated species richness (upper curve), six months after fire (middle curve) and unburned or before fire (lower curve). Sobs (Mao Tau): Total number of species observed in all samples pooled; ACE Mean: Abundance-based Coverage Estimator of species richness (mean among runs).

Significance of invasive alien trees adjacent to burned fynbos

Fynbos vegetation had higher invertebrate species richness than the adjacent IATs (Figure 6.13). However, more species were recorded near the boundary between the two adjoining habitat fragments, while decreasing into the cores (Figure 6.13). Species richness was lower in IATs than in adjacent fynbos for most invertebrate taxa (see Chapter 4). Abundance within IATs was reduced following fire. However, abundance recovered in fynbos three months later, with lower abundance in the fynbos core (i.e. 128 m from the IATs) (Figure 6.14).

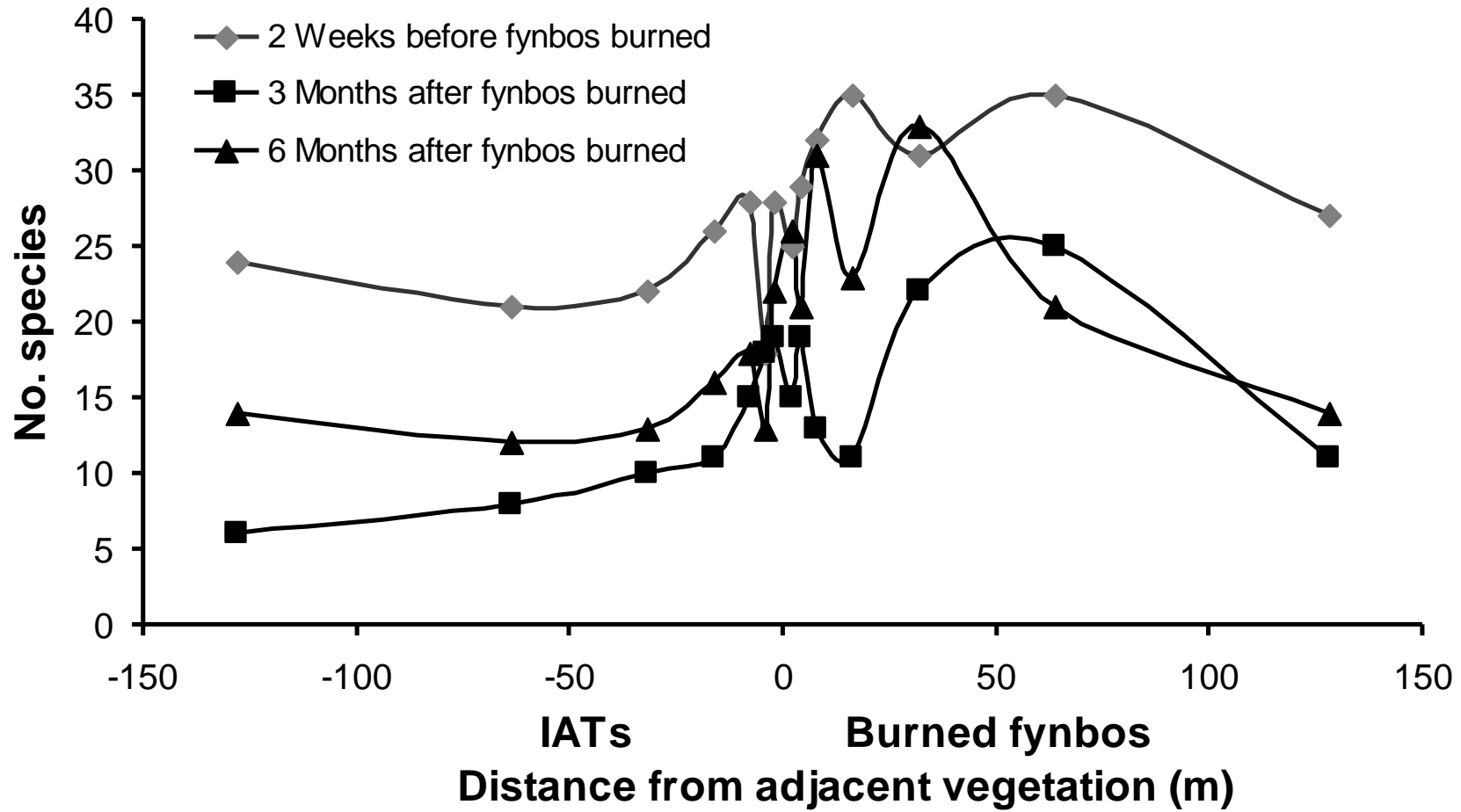


Figure 6.13 Species richness variations along invasive alien trees (IATs)/fynbos ecotone. 0 m = ecotone.

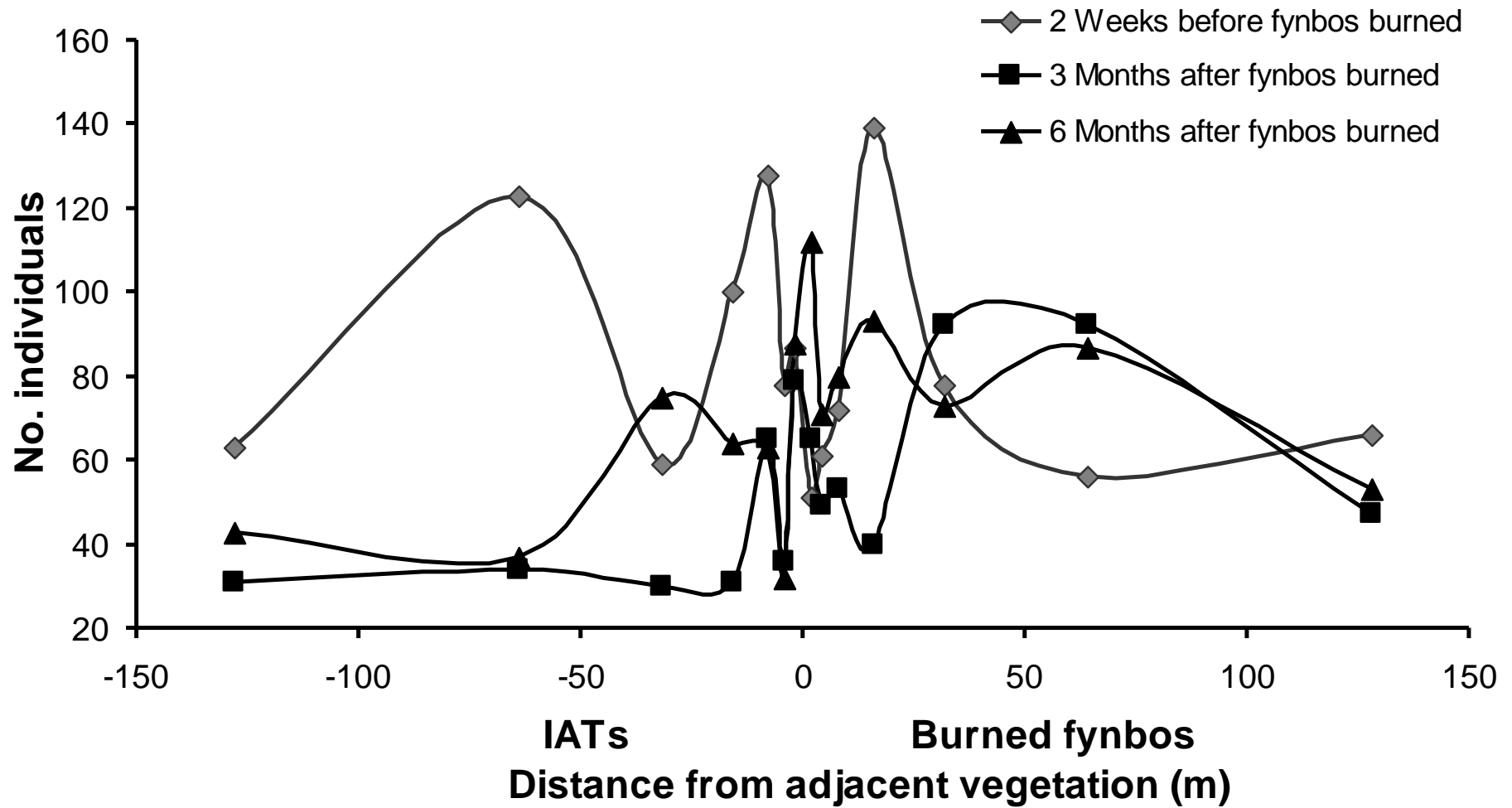


Figure 6.14 Species abundance variations along invasive alien trees (IATs)/fynbos vegetation ecotone. 0 m = ecotone.

Invertebrate assemblage composition in burnt fynbos vegetation

The Cluster (Figure 6.15) and MDS (Figure 6.16) analyses highlighted the potential for differences in invertebrate assemblages before and immediately after fire within the fynbos. Figure 6.15 displays the results of a cluster analysis on invertebrate assemblage composition at an arbitrary similarity level of around 20%.

Figure 6.16 shows the 2-dimensional MDS plot of the same species composition similarities. Samples grouped according to invertebrate assemblage composition with high percentage similarities. There was a good separation of samples collected at different times relative to the fire event based upon their invertebrate assemblage composition, as shown by MDS analyses (Figure 6.16).

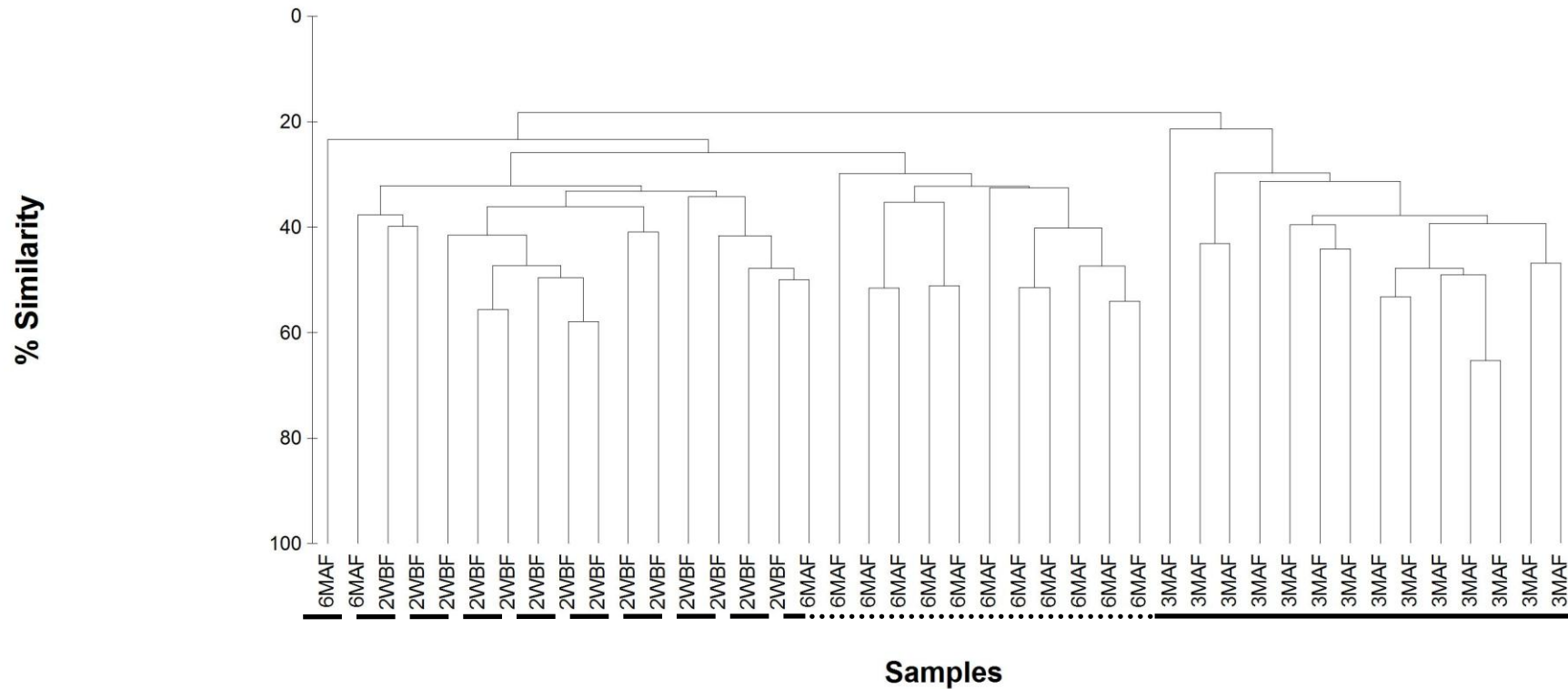


Figure 6.15 Classification tree of samples in terms of invertebrate assemblages using group-average linking on Bray-Curtis species similarities in fynbos. 2WBF = Two weeks before fire, 3MAF = Three months after fire, and 6MAF = Six months after fire.

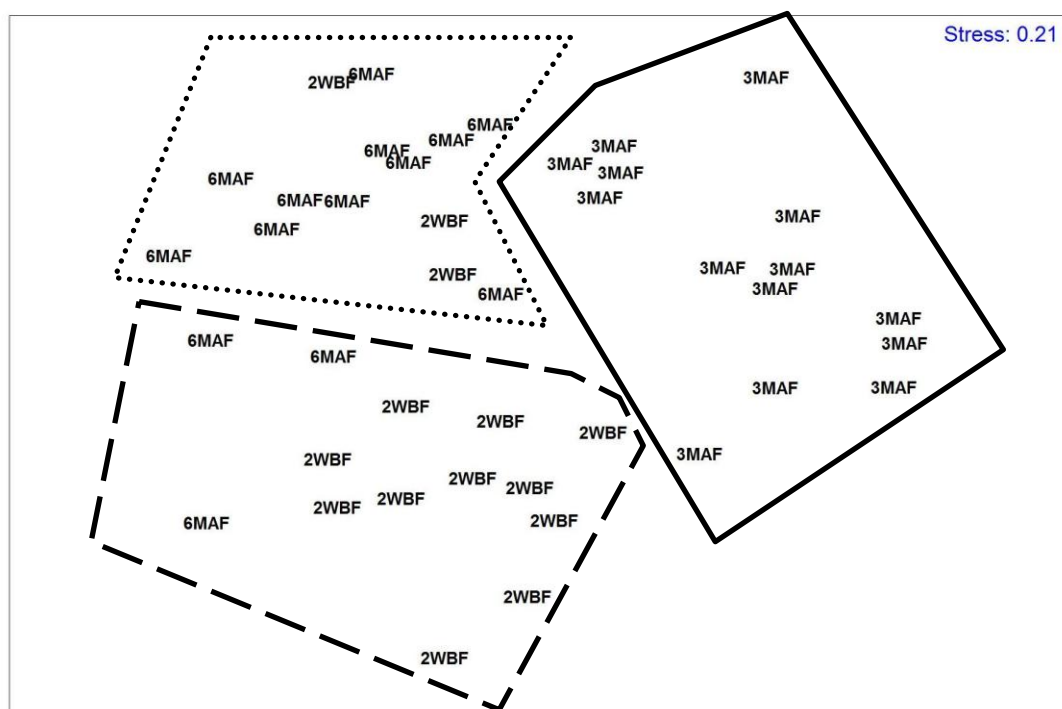


Figure 6.16 Multi-dimensional scaling (MDS) analysis of Bray-Curtis similarity between invertebrate samples from different sampling periods (2WBF: two weeks before fire; 3MAF: three months after fire and 6MAF: six months after fire) in the fynbos.

Testing the significance of invertebrate assemblage composition differences before and after fire in the fynbos

The global test of the null hypothesis based on invertebrates sampled during different times relative to the fire was rejected ($R = 0.704$, $P = 0.001$). This means that the invertebrate assemblage composition before and after fire was not similar, and that assemblage groups could be distinguished based on sampling period relative to the fire event. Therefore, there were sampling period-by-sampling period similarities worth investigating. This was done by completing the pair-wise test for the sampling period comparisons (Table 6.3).

The spread of R -values possible from three sampling periods can be seen in Figure 6.17. An observed value of $R = 0.704$ was seen to be a most unlikely event, with a probability of less than 1 in 1000 if null hypothesis (H_0) is true. Therefore, H_0 was

rejected at a significance level of $P < 0.001$. Nevertheless, $R = 0.704$ may still have been the most extreme outcome observed if larger number of simulations was chosen.

Table 6.3 Statistically significant comparisons based upon invertebrate assemblage composition in fynbos, two weeks before fire (2WBF), three months after fire (3MAF) and six months after fire (6MAF) event.

Sampling period	R Statistic	Significance level	Possible Permutations	Actual Permutations	Number \geq Observed
2WBF Vs 3MAF	0.777	0.001	20058300	999	0
2WBF Vs 6MAF	0.611	0.001	20058300	999	0
3MAF Vs 6MAF	0.745	0.001	20058300	999	0

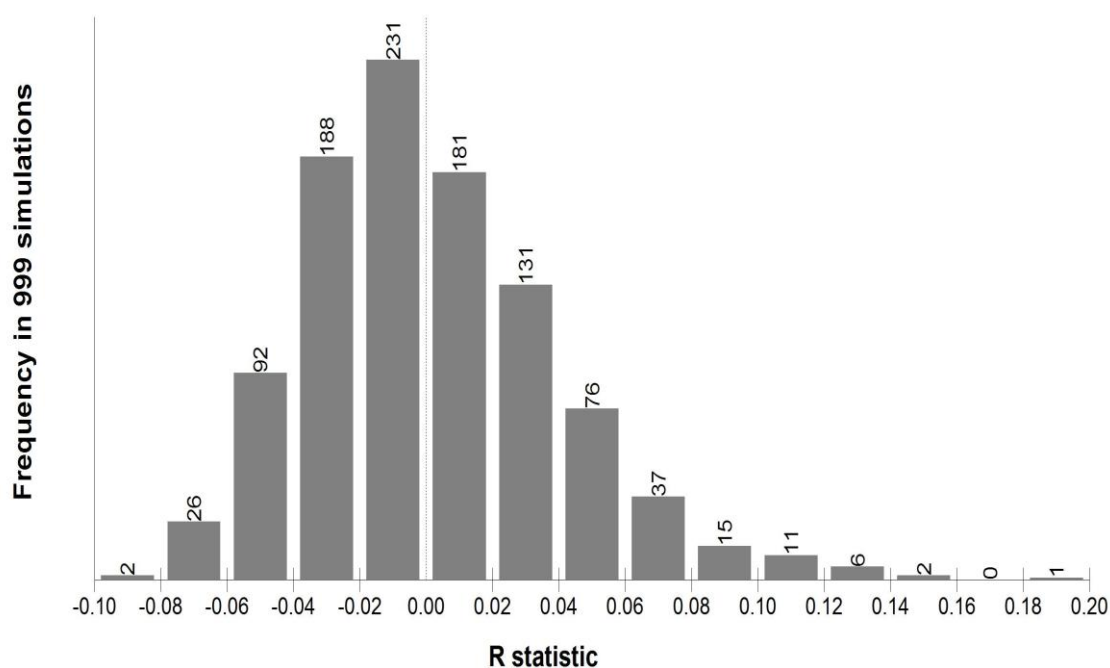


Figure 6.17 Simulated distribution of the test statistic R under the null hypothesis of ‘no sample differences relative to fire event’; this contrast with an observed value for R of 0.704.

Species typifying a fynbos vegetation type

Table 6.4 shows the contributions of each species to the Bray-Curtis similarity within a fynbos vegetation type before fire. The Bray-Curtis similarity between all pairs of samples in the fynbos vegetation type was 28.04, made up mainly of the contributions from eight species: *Diores youngai* (4.28, i.e. 15.28%), *Centrobolus* sp.2 (3.44, i.e. 12.28%), etc, with a cumulative contribution of over 60% of the total similarity of 28.04. However, the list only considers top species until 60% cumulative percentage was reached (see Table 6.4).

Table 6.4 Average similarities between invertebrate samples at fynbos vegetation, two weeks before fire. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 28.04.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Diores youngai</i>	5.43	4.28	1.67	15.28	15.28
<i>Centrobolus</i> sp.2	3.71	3.44	2.21	12.28	27.56
<i>Camponotus</i> sp.2	3.14	2.76	1.65	9.85	37.40
<i>Langona</i> sp.1	1.86	1.56	1.04	5.56	42.97
<i>Tetramorium capense</i>	2.71	1.55	1.15	5.54	48.51
<i>Cecidomyia</i> sp.1	2.29	1.23	0.83	4.39	52.90
<i>Sminthurus viridis</i>	2.86	1.23	0.60	4.37	57.28
<i>Porcello</i> sp.	1.86	1.13	0.59	4.04	61.31

Species typifying a burned fynbos vegetation

Table 6.5 shows the contributions of each species to the Bray-Curtis similarity within a burned fynbos vegetation type, at least three months after fire. The Bray-Curtis similarity between all pairs of samples in the burned fynbos vegetation type after three months was 40.34, made up mainly of the contributions from four species: *Daspletis* sp. (8.67, i.e. 21.50%), *Camponotus* sp.2, *Hopliini* sp. and *Zophosis boei*, with a cumulative contribution of over 63% of the total similarity of 40.34. However, the list only considers the top species until 60% cumulative percentage was reached (see Table 6.5).

Table 6.5 Average similarities between invertebrate samples at fynbos vegetation type, three months after fire. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 40.34.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Daspletis</i> sp.	6.71	8.67	2.48	21.50	21.50
<i>Camponotus</i> sp.2	6.71	7.08	1.98	17.55	39.06
Hopliini sp.	7.29	5.51	0.61	13.67	52.72
<i>Zophosis boei</i>	5.43	4.48	1.21	11.12	63.84

Table 6.6 shows the contributions of each species to the Bray-Curtis similarity within a burned fynbos vegetation type, at least six months after fire. The Bray-Curtis similarity between all pairs of samples in the burned fynbos vegetation type after six months was 37.95, made up mainly of the contributions from three species (Table 6.6). However, the list only considered the top species until 60% cumulative percentage was reached.

Table 6.6 Average similarities between invertebrate samples at fynbos vegetation type, six months after fire. Av.Abund = average abundance, Av.Sim = average similarity, Sim/SD = average similarity/ standard deviation of the average, Contribut% = % contributed towards total average similarity per taxon, and Cum% = cumulative percentage total of all taxa towards average similarity. Average similarity: 40.34.

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
<i>Microhodotermes viator</i>	19.00	14.41	2.18	37.98	37.98
<i>Centrobolus</i> sp.2	12.71	5.37	0.80	14.14	52.12
<i>Gryllus bimaculatus</i>	4.71	3.43	1.38	9.05	61.17

Determining discriminating species

Table 6.7 shows the results of the analysis of the invertebrate assemblage dissimilarities before and after fire. The average of the Bray-Curtis dissimilarities between all pairs of samples, before and after fire was 80.24, which was made up mainly *Microhodotermes viator* and *Centrobolus* sp.2. The *M. viator* contribution was 16.12% of the total 80.24,

Centrobolus sp.2 contributed 8.96 to this total, etc. (column 5) and these percentages were cumulated in column 6, until the cut-off of >60% was reached. Column 4 is the ratio of the average contribution (column 3) divided by the standard deviation (SD) of all contributions making up this average.

Columns 1 and 2 provide the average abundance of species in CIATs and fynbos vegetation types respectively. 4th-root transformation was used, and therefore the means in columns 1 and 2 were calculated on these transformed abundances. The higher ratio of the contribution to the standard deviation (SD) for species such as *M. viator*, *D. youngai*, *G. bimaculatus*, etc (marked with a star in column 4 of Table 6.7), was an indication that they were also consistently present in large numbers.

Table 6.7 Discriminating species between fynbos vegetation before fire and six months after fire, ordered in decreasing contribution. Average dissimilarity = 80.24.

Species	Fynbos	Vineyard	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Microhodotermes viator</i>	7.43	19.00	12.93	2.15*	16.12	16.12
<i>Centrobolus</i> sp.2	3.71	12.71	7.19	0.79	8.96	25.08
<i>Diores youngai</i>	5.43	0.14	3.42	1.42*	4.27	29.35
<i>Gryllus bimaculatus</i>	0.29	4.71	3.13	1.15*	3.89	33.24
<i>Crematogaster peringueyi</i>	2.57	3.14	2.71	0.75	3.38	36.62
<i>Linepithema humile</i>	1.71	4.43	2.60	0.89	3.24	39.86
Sphecidae sp.2	0.14	3.14	1.96	0.91	2.45	42.31
<i>Sminthurus viridis</i>	2.86	0.00	1.86	0.85	2.31	44.63
<i>Tetramorium capense</i>	2.71	3.00	1.68	1.57*	2.10	46.72
<i>Scarabaeus rigosus</i>	2.71	0.86	1.60	0.86	1.99	48.72
<i>Cecidomyia</i> sp.1	2.29	0.00	1.57	0.88	1.96	50.68
<i>Daspletis</i> sp.	0.43	2.57	1.54	1.19*	1.92	52.60
<i>Camponotus</i> sp.2	3.14	1.43	1.50	1.12*	1.87	54.47
<i>Oxyopes</i> sp.	0.00	2.29	1.48	1.81*	1.85	56.31
<i>Zophosis boei</i>	1.71	1.43	1.30	1.10*	1.62	57.93
<i>Sphex bonaspei</i>	1.57	0.86	1.27	0.63	1.58	59.51
<i>Porcello</i> sp.	1.86	0.43	1.26	1.05*	1.57	61.08

Linepithema humile

ANOVA among fynbos sites, before and after fire, showed that there was no significant difference (df = 13, f = 2.007, p = 0.177) in terms of invasive alien Argentine ant, *Linepithema humile*. This was also the case in IATs (df = 13, f = 0.175, p = 0.842).

Species assemblage changes in burned fynbos vegetation

When arthropod assemblage composition from burned fynbos site were compared with those from unburned fynbos (control) site to determine any deviation from the natural state, there was a good separation of samples collected at different times relative to the fire event based upon their invertebrate assemblage composition, as shown by MDS analyses (Figure 6.18).

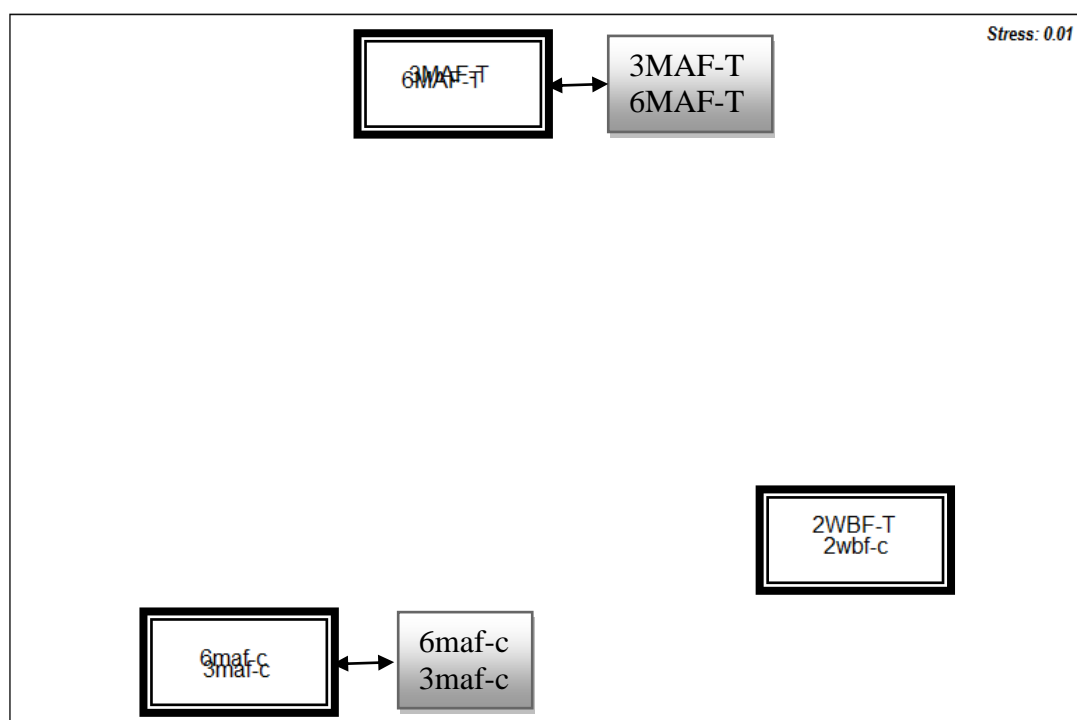


Figure 6.18 Multi-dimensional scaling (MDS) plot showing similarity between invertebrate samples from different sampling periods (2WBF-T: two weeks before fire at treatment site; 3MAF-T: three months after fire at treatment site and 6MAF-T: six months after fire at treatment site) in the fynbos. Control /unburned fynbos site samples: 2wbf-c = two months before fire at control site; 3maf-c = three months after fire at control site; 6maf-c = six months after fire at control site.

DISCUSSION

Invertebrate responses to burned invasive alien trees

It is possible that the risk of fire was increased by the presence of stands of IATs (Macdonald and Richardson, 1986; Richardson and van Wilgen, 2004), especially with the build-up of high fuel loads (natural, commercial and invasive). IATs sites supported lower species abundance than adjacent fynbos. This suggests that degradation of habitats through IAT invasion better explains the decline of surface-active invertebrates than fire alone.

Significance of vineyards adjacent to burned invasive alien trees

Vineyards had higher invertebrate species richness than IATs, although more species were recorded at the ecotone between the two. Reduction of species richness into the IATs core might be an indication of less preferred habitat than vineyards. More individuals were present in the ecotone than in the burned IATs. This implies that IATs habitat conditions may have worsened following fire. Abundance was more similar on either side of the vineyard/IATs ecotone before fire. However, this was not the case after fire, where abundance was higher at the 6-month post-burn sites. Burning of understorey vegetation in the IATs might have opened the way for some plant species that were excluded before fire.

Invertebrate assemblage composition before and after fire within IATs

There was no clear distinction between invertebrate assemblage composition before and after fire within IATs. Moreover, ANOSIM suggested that the invertebrate assemblage composition before and after fire was similar, indicating no effects of fire on invertebrate assemblages within the IATs.

Invertebrate responses to burned fynbos

Fire affects foliar nutritional suitability, with susceptibility to invertebrate infestation often drastically altered (Price, 1991; Rieske et al., 2002). Fire may also have a direct impact on phytophagous invertebrates by altering habitat heterogeneity, and species composition (Siemann et al., 1997), or indirectly through cascading effects caused by alteration in food quality and availability (Rieske et al., 2002).

Above ground habitat in the fynbos sites was reduced by fire, greatly decreasing the area available for surface-active invertebrates. However, increased species richness and

abundance six months after fire indicated that certain invertebrates benefited from fire. Similar results were also reported in Australia (Whelan et al., 1980). In contrast, decreased abundance of invertebrates immediately after fire has been recorded in most studies (e.g. Rice, 1932; Pearse, 1943; Springett, 1976; Bornemissza, 1969; Buffington, 1967; O'Dowd and Gill, 1984). After high intensity-fire, an initial decrease in the abundance of some invertebrates may be associated with depleted resources. Nevertheless, burned vegetation is important for many invertebrate species (Wikars, 2002), and several invertebrate species breed almost exclusively in burned forest (Wikars, 1997). Although, the substrate requirements of some species are known, there is a lack of basic knowledge on the majority of saproxylic species (Johansson et al., 2007). Several insect species respond to smoke and heat generated by wildfires, and they use recent burns to reproduce in high numbers (Saint-Germain et al., 2008). Some of these species are rare or uncommon in undisturbed areas, and the contribution of recently-burned habitats is crucial to their long-term persistence.

Species richness declined substantially immediately after the fire, through lack of resources and suitable habitats. Nevertheless, recovery was rapid, with common species recolonizing or re-establishing within three months. Bornemissza (1969) highlighted the speed of reinvasion by soil invertebrates following fire, being associated with the accumulation of leaf litter under trees and with the regeneration of herbs and shrubs in exposed areas. Survival of invertebrates subsequent to fire is affected by a variety of biotic and edaphic factors. Important biotic components include food source (plant or prey), competition, predation (including parasites) and the relationship between species. Plant regrowth and accumulation of dead organic materials following fire represents an increase in resources with time after fire, and may differ from the pre-fire resources in terms of quality as well as quantity. Surprisingly, rarer species both of presumed more mobile (flies) and relatively sedentary (spiders) taxa reappeared at similar rates. Nevertheless, spiders are known to have the ability to rapidly colonize disturbed areas (Bishop and Riechert, 1990), partly through ballooning by juveniles and adults of small species (Bell et al., 2005; Gavish-Regev et al., 2008).

Significance of invasive alien trees adjacent to burned fynbos

Fynbos had higher invertebrate species richness than adjacent IATs. However, as a result of fire, more species were at the ecotone than in the cores. There was a

significant reduction in species richness and abundance following fire in both vegetation types. Species diversity within the IATs was indirectly affected by the fire within the adjacent fynbos. Moreover, both species richness and abundance were higher in the fynbos than in IATs following fire. However, some invertebrate taxa were no longer represented six months after the fire. This suggests that certain invertebrate taxa are fire-tolerant, while others not. It also implies that some invertebrates had difficulty surviving in the IATs core, and remained within the ecotone, probably only recolonizing or re-establishing in the fynbos fragment once it recovered. This suggests that native habitat specialists prefer to stay within the ecotone rather than inhabit the adjacent IATs core during unsuitable conditions. However, some may have been in egg or pupal form in the ground where they were able to survive the fire.

Comparison of the suites of species recorded before fire with those after fire showed that the set of colonizing species was not the same. The relatively rapid colonization of fynbos by a suite of certain species, and not necessarily those that were on a particular vegetation type before fire, indicated that rarity was due neither to inability of the species to colonize the vegetation nor to the suitability of the fynbos vegetation conditions for these species. Whatever the factors that maintain suites of species in each vegetation type after fire, it is more likely that the ability of the species to reach and colonize make it suitable only for a particular subset of species. The recruitment of some invertebrates during the post-fire period of the study may be attributed to the prevailing conditions that may have been particularly unfavourable for soil invertebrate survival and habitat simplification.

Some invertebrates may survive better in highly altered landscapes because they are good at dispersing between suitable habitats and reproducing in newly colonized habitats (Kotze and O'Hara, 2003). Nevertheless, some invertebrates, such as the invasive alien ant *Linepithema humile*, survived the fire and maintained its relative abundance. It is likely that this ant escaped fire by remaining underground.

Invertebrate assemblage composition

Some samples collected two weeks before fire (2WBF) were mixed with those collected six months after fire (6MAF) in the fynbos. This suggests that there were strong similarities among samples based on their invertebrate assemblage composition.

Indeed, fire appeared to have a minimal impact on certain invertebrates that may be tolerant to fire (Uys et al., 2006). However, samples in the months immediately following the fire event within the fynbos had a distinct invertebrate assemblage composition. Nevertheless, such an assemblage was only temporary, as some samples had a different assemblage six months after fire, which was more comparable to the original assemblage. Therefore, original assemblage composition only begins to recover six months later after fire. A previous study found similar invertebrate community structure at least three months after fire (Uys et al., 2006). Invertebrate assemblage composition before fire was different from that after fire, indicating that fire had a major influence on invertebrate assemblage composition. The null hypothesis that there was no difference between invertebrate assemblage before and after fire was rejected.

The higher ratio of the contribution to the standard deviation (SD) for species such as the millipede *Centrobolus* sp.2, the spider *Diores youngai* and the ant *Camponotus* sp.2 was an indication that they were also consistently present in large numbers in the fynbos vegetation. These invertebrate species can be identified as typical of unburned fynbos vegetation type.

The fly *Daspletis* sp., the ant *Camponotus* sp.2, the beetles *Zophosis boei* and *Hopliini* sp. can be identified as typical species of burned fynbos vegetation immediately after fire, while the termite *Microhodotermes viator*, *Centrobolus* sp.2 and the cricket *Gryllus bimaculatus* can be seen as typical species of burned fynbos vegetation, at least six months following fire.

M. viator, the spider *Oxyopes* sp., the ant *Tetramorium capense*, the spider *D. youngai*, *Daspletis* sp., *G. bimaculatus*, *Camponotus* sp.2, *Z. boei* and the slaters *Porcello* sp. were identified as good discriminating species between burned and unburned fynbos vegetation.

Species assemblage changes relative to unburned fynbos vegetation

Both the burned site and control/unburned site had comparable arthropod assemblages before the fire (i.e. 2 weeks before fire). However, three months after fire, the arthropod assemblages at the two sites differed. After fire, the burned site had an arthropod assemblage composition that was not comparable to that in the control site. Then, three and six months after fire, assemblage composition at both sites were similar again. However, seasonality may also be associated with the different in assemblage composition in the unburned fynbos vegetation, although the seasonality effect per se would have been the same across all the sites.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

Different taxa of surface-active invertebrates contribute to post-fire populations according to their respective abilities to survive fire and the subsequent environmental conditions. Well-managed plantation forestry may serve as a refuge area for some species in the region, especially where the surrounding land-use practices have been destructive to their most preferred habitats. Certain invertebrate species may benefit from these plantations in terms of food and reproduction, which increases their area of occupancy. It is likely that mobile species escaped fire by sheltering in adjacent unburned vegetation. Nevertheless, density of some alien species such as the Argentine ant was not significantly reduced by fire. Instead, this ant continued to increase in abundance following fire.

Fire may have a detrimental effect on an ecosystem, especially when occurring at inappropriate times (i.e. periods of species active growth). Nevertheless, controlled burning of vegetation may be beneficial to some aspects of compositional biodiversity, especially for those species that are dependent on fire for reproduction. It is possible that such reduced invertebrate species richness in alien vegetation may result in major reduction in insectivorous species mainly because of food shortages. Wildfires often lead to the impoverishment of diversity by continuously favouring some species (e.g. Argentine ant) and not others.

These findings have important implications for conservation strategies because burned areas are influenced by the effects of adjacent fragment boundary, losing some species and being invaded by other species. Studies that only assess species richness between adjacent habitat fragments cannot detect such patterns in species composition

and distribution. This study has implications for vegetation and habitat management in highly fragmented landscapes, especially in areas where disturbances such as fire occur. Habitat fragments that escape fire, or are only partially burned, may continue to support suitable habitats and be a source of certain invertebrates, thereby facilitating overall assemblage recovery. Fire had a significant effect on invertebrate assemblage composition in the fynbos vegetation. However, invertebrate assemblage composition within alien vegetation was not significantly influenced by fire. The disruption of remaining or adjacent unburned fynbos habitats can modify refugial habitats and, in turn, impair invertebrate diversity recovery and species persistence in fragmented areas.

The general change in invertebrate species richness and abundance and assemblage composition sampled after the fire should be interpreted with caution. Data from successive years before fire, or from comparable unburned sites, is required to separate fire impact from arthropod annual variation without fire. However, the response of arthropod assemblage compositions in fynbos vegetation following fire event, suggests fire has a greater impact exceeding annual variation. Some groups, particularly the phytophages and parasitoids were greatly influenced by fire.

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APPENDICES

Appendix 6.1 Invertebrates recorded two weeks before fire, three and six months later after fire and their trophic guilds for invasive alien trees (IATs) adjacent to the vineyards. * represents known alien species.

Trophic guild	Order/Family	Species	Before fire		After fire			
			2 weeks		3 months		6 months	
			Vineyard	IATs	Vineyard	IATs	Vineyard	IATs
Araneae								
Predator	Gallieniellidae	<i>Drassodella</i> sp.1		1	1	2		2
Predator	Gnaphosidae	<i>Camillina</i> sp.1	2	5				2
Predator	Gnaphosidae	<i>Pterotricha varia</i>	1	5	6	5		
Predator	Gnaphosidae	<i>Zelotes lightfooti</i>		1	5	2		3
Predator	Lycosidae	<i>Hogna</i> sp.1						1
Predator	Lycosidae	<i>Proevippa schreineri</i>	1		8	1	1	3
Predator	Lycosidae	<i>Proevippa</i> sp.1	3	2	4			
Predator	Lycosidae	<i>Proevippa</i> sp.2	2	1		2		
Predator	Lycosidae	Sp.1			1			1
Predator	Lycosidae	<i>Trabea purcelli</i>						20
Predator	Oxyopidae	<i>Oxyopes</i> sp.	3	3	2	1		3
Predator	Palpimanidae	<i>Diaphorocellus</i> sp.1		2		2	1	2
Predator	Philodromidae	<i>Tibellus</i> sp.				3		
Predator	Pisauridae	<i>Rothus purpurissatus</i>	1	2	2			
Predator	Salticidae	<i>Aelurillus</i> sp.1	1					3
Predator	Salticidae	<i>Evarcha</i> sp.1	6		2		3	1
Predator	Salticidae	<i>Habrocestrum</i> sp.1			5	3		1
Predator	Salticidae	<i>Langona</i> sp.1	6	2			1	1
Predator	Theridiidae	<i>Steatoda capensis</i>				1		
Predator	Theridiidae	<i>Steatoda</i> sp.2	1				1	5
Predator	Thomisidae	<i>Synema imitator</i>						1
Predator	Zodariidae	<i>Diores capensis</i>	2	1	4	1		
Predator	Zodariidae	<i>Diores youngai</i>	34	10	8	6	2	2
Blattodea								
Omnivore	Blaberidae	<i>Aptera fusca</i>				1		
Omnivore	Blattidae	<i>Temnopteryx phalerata</i>			1			2
Coleoptera								
Wood borer	Anobiidae	<i>Xestobium</i> sp.	112	6		2		
Omnivore	Anthicidae	<i>Endomia elongatus</i>	2	6		2		
Omnivore	Anthicidae	<i>Formicomus coeruleus</i>	4	3	1		3	
Wood borer	Buprestidae	<i>Achmaedera</i> sp.	2	2	4	2		
Predator	Cantharidae	<i>Cantharis</i> sp.1			1			
Predator	Carabidae	<i>Acanthoscelis ruficornis</i>				2		
Predator	Carabidae	<i>Caminara</i> sp.1	1		11		1	
Predator	Carabidae	<i>Graphipterus trilineatus</i>	1					
Predator	Carabidae	<i>Microlestia tabida</i>	1					
Predator	Cleridae	Sp.1	6	19	1	1		1
Predator	Coccinellidae	<i>Cheilomenes lunata</i> *	11		16	3		
Predator	Coccinellidae	<i>Harmonia axyridis</i> *	3	5	4	1		
Phytophage	Curculionidae	Sp.4	1	5	1			1
Phytophage	Curculionidae	Sp.1	14	3	18	2	12	1
Nectarivore	Dermestidae	<i>Dermestes maculatus</i>		1		1		5
Nectarivore	Dermestidae	<i>Dermestes</i> sp.		1				
Phytophage	Elateridae	<i>Cardiotarsus acuminatus</i>		1				

Trophic guild	Order/Family	Species	Before fire		After fire			
			2 weeks		3 months		6 months	
			Vineyard	IATs	Vineyard	IATs	Vineyard	IATs
Predator	Lampyridae	<i>Lampyris disticollis</i>		1		1		
Nectarivore	Mordellidae	Sp.2	1		1			1
Saprophage	Nitidulidae	Sp.1		1	1	6		
Saprophage	Nitidulidae	Sp.2	5	19	3			
Saprophage	Scarabaeidae	Sp.2						1
Saprophage	Scarabaeidae	<i>Aphodius</i> sp.	1		2		5	
Saprophage	Scarabaeidae	Hopliini sp.	2	1	7	1		
Saprophage	Scarabaeidae	<i>Scarabaeus rigosus</i>		1				
Phytophage	Scarabidae	Sp.1			2			
Phytophage	Silvanidae	<i>Oryzaephilus</i> sp.				2		
Predator	Staphylinidae	<i>Paederinae</i> sp.		6	9	5	1	5
Saprophage	Tenebrionidae	Sp.1		1	2			
Saprophage	Tenebrionidae	Sp.2	4	4		3		
Saprophage	Tenebrionidae	Sp.3	1					
Saprophage	Tenebrionidae	<i>Tenebrio</i> sp.1				1	4	26
Saprophage	Tenebrionidae	<i>Trigonopus</i> sp.1	8	5	13			3
Saprophage	Tenebrionidae	<i>Zophosis boei</i>	42		47	20	4	
Collembola								
Saprophage	Sminthuridae	<i>Sminthurus viridis</i> *	10				26	
Crustacea								
Saprophage	Amphipoda	<i>Paramelita nigroculus</i>						8
Dermaptera								
Scavenger	Forficulidae	<i>Forficula senegalensis</i>	2	3	7		7	
Diptera								
Saprophage	Anthomyiidae	<i>Anthomyia</i> sp.		3			3	6
Predator	Asilidae	<i>Daspletis</i> sp.	3		17	4	5	
Parasitoid	Bombyliidae	<i>Exoprosopa</i> sp.		1	1			
Saprophage	Calliphoridae	<i>Chrysomya chloropyga</i>					1	
Saprophage	Calliphoridae	Sp.1		4		1		3
Saprophage	Calliphoridae	Sp.2	1	1		1	2	4
Phytophage	Cecidomyiidae	<i>Cecidomyia</i> sp.1						6
Saprophage	Heleomyzidae	<i>Helomyza picta</i>		2				1
Saprophage	Muscidae	Sp.1		2			1	
Parasitoid	Pipunculidae	Sp.1	21	4		1		11
Saprophage	Sciaridae	Sp.1	20	36				
Predator	Syrphidae	Sp.1		1		1		
Parasitoid	Tachinidae	Sp.1			1			
Parasitoid	Tachinidae	Sp.2		1		2		
Parasitoid	Tachinidae	Sp.4	1					1
Parasitoid	Tipulidae	<i>Tipula jocosae</i>	2					
Hemiptera								
Predator	Anthocoridae	<i>Anthocoris</i> sp.1	7	1			2	
Phytophage	Cercopidae	<i>Poophilus</i> sp.1	3	2	4		3	
Phytophage	Cicindelidae	<i>Cicindela quadriguttata</i> .		1				
Phytophage	Cydnidae	Sp.1			1	1	1	
Phytophage	Miridae	<i>Deraeocoris</i> sp.		1				
Phytophage	Pentatomidae	<i>Agonoscelis</i> sp.1	1			1		
Phytophage	Pentatomidae	<i>Antestia lymphata</i>					1	
Phytophage	Pentatomidae	<i>Bagrada hilaris</i>					4	
Phytophage	Pyrrhocoridae	<i>Scantius forsteri</i>			1	3		2
Predator	Reduviidae	<i>Cleptria rufipes</i>		1	4	2		
Predator	Reduviidae	?Sp.		1				6

Trophic guild	Order/Family	Species	Before fire		After fire				
			2 weeks		3 months		6 months		
			Vineyard	IATs	Vineyard	IATs	Vineyard	IATs	
	Hymenoptera								
Nectarivore	Anthophoridae	<i>Thyreus delumbatus</i>				1			
Nectarivore	Apidae	<i>Apis mellifera</i>		2		2			
Nectarivore	Apidae	<i>Meliponula</i> sp.1		1		5			
Parasitoid	Braconidae	<i>Charops</i> sp. 1					2		
Omnivore	Formicidae	<i>Camponotus maculatus</i>						1	
Omnivore	Formicidae	<i>Camponotus</i> sp.1		1		1	2		
Omnivore	Formicidae	<i>Camponotus</i> sp.2		8	4	61	38	22	2
Omnivore	Formicidae	<i>Camponotus</i> sp.3				3			
Omnivore	Formicidae	<i>Crematogaster peringueyi</i>			16	5	25	4	8
Omnivore	Formicidae	<i>Linepithema humile</i> *		12	12	17	16	65	7
Omnivore	Formicidae	<i>Messor capensis</i>					1		
Omnivore	Formicidae	Sp.1						6	2
Omnivore	Formicidae	Sp.2		15		3			
Omnivore	Formicidae	<i>Tetramorium capense</i>		1	11	9	3	5	13
Nectarivore	Halictidae	Sp.1		4		2	2	1	
Nectarivore	Melittidae	Sp.1							1
Nectarivore	Melittidae	Sp.2		1		1			
Nectarivore	Mutillidae	Sp.1							1
Nectarivore	Mutillidae	Sp.2					1		
Parasitoid	Pompilidae	Sp.1			3	1	5	3	3
Parasitoid	Pompilidae	Sp.2		1	1				
Parasitoid	Scoliidae	<i>Campsomeriella</i> sp.1				1			
Predator	Sphecidae	Sp.1		2					
Predator	Sphecidae	Sp.2				1		6	11
Predator	Sphecidae	<i>Sphex bonaspei</i>				2	3		
Predator	Sphecidae	<i>Sphex tydei</i>						1	
Nectarivore	Tiphiidae	Sp.1		1	2			1	
	Isopoda								
Saprophage	Porcellionidae	<i>Porcello</i> sp.			41	43	140	2	81
	Isoptera								
Saprophage	Hodotermitidae	<i>Microhodotermes viator</i>				20	29	4	
	Spirobolida								
Saprophage	Pachybolidae	<i>Centrobolus</i> sp.1		50	61	1	31		3
Saprophage	Pachybolidae	<i>Centrobolus</i> sp.2		25	61	4	10	161	222
	Lepidoptera								
Phytophage	Arctiidae	<i>Rhodogastria amasis</i>			1				
Phytophage	Geometridae	Sp.1		1					
Phytophage	Lycaenidae	<i>Aloeides</i> sp.		1	2				
Phytophage	Noctuidae	<i>Helicoverpa armigera</i>		10	3	12	3	1	
Phytophage	Noctuidae	Sp.1				1	1	1	
Phytophage	Noctuidae	Sp.2		4	3	5	3	1	
Phytophage	Nymphalidae	<i>Cynthia cardui</i>		6		14			
Phytophage	Pyalidae	Sp.1		2	2				
Phytophage	Sphingidae	<i>Coelonia fulvinitata</i>						1	
	Lithobiida								
Saprophage	Lithobiidae	<i>Lithobius</i> sp.1		4			1		
Saprophage	Lithobiidae	<i>Lithobius</i> sp.2		6	2		1	1	
	Mantodea								
Predator	Mantidae	Sp.1						1	
	Orthoptera								
Omnivore	Gryllidae	<i>Cophogryllus</i> sp.		1	1	2	5		1
Omnivore	Gryllidae	<i>Gryllus bimaculatus</i>		2	9	15	4	10	18

Trophic guild	Order/Family	Species	Before fire		After fire			
			2 weeks		3 months		6 months	
			Vineyard	IATs	Vineyard	IATs	Vineyard	IATs
	Scolopendromorpha							
Predator	Scolopendromorphae	<i>Scolopendra cingulata</i>	2	1	1			
	Scorpiones							
Predator	Buthidae	<i>Uroplectes lineatus</i>					1	
Predator	Scorpionidae	<i>Opisththalmus macer</i>			2			
	Solifugae							
Predator	Galeodidae	<i>Galeodes</i> sp.1	6		10			
Predator	Galeodidae	<i>Galeodes</i> sp.2	5		5			1
Total individuals			530	427	479	429	416	496

Appendix 6.2 Invertebrates recorded two weeks before fire, three and six months later after fire within treatment/burned fynbos sites (2WBF-T, 3MAF-T and 6MAF-T) and control/unburned fynbos sites (2wbf-c, 3maf-c and 6maf-c). ‘+’ represents species present, ‘-’ represents species absent.

Family/species name	Treatment/burned			Control/unburned		
	2WBF-T	3MAF-T	6MAF-T	2wbf-c	3maf-c	6maf-c
<i>Aptera fusca</i>	+	+	+	-	+	-
<i>Temnopteryx phalerata</i>	-	+	-	-	-	-
<i>Endomia elongatus</i>	-	-	-	+	-	-
<i>Achmaedera</i> sp.	-	-	-	+	+	+
<i>Xestobium</i> sp.	+	+	+	+	-	+
<i>Oryzaephilus</i> sp.	-	-	-	-	-	+
Carabidae sp.1	+	+	+	-	-	+
<i>Microlestia tabida</i>	+	-	-	-	-	+
<i>Anthia decemquttata</i>	+	+	+	+	-	-
<i>Caminara</i> sp.1	-	-	-	+	-	-
<i>Dermestes maculatus</i>	-	-	+	-	-	-
Cleridae sp.1	-	-	-	+	-	-
<i>Cheilonas lunata</i>	+	-	-	+	+	-
Curculionidae sp.1	-	-	+	-	-	+
Curculionidae sp.3	-	-	-	-	-	+
Curculionidae sp.3	+	-	-	+	-	-
Curculionidae sp.4	+	-	-	-	+	-
Mordellidae sp.2	-	-	-	+	-	+
Nitidullidae sp.1	+	+	-	-	+	+
Nitidullidae sp.2	-	-	-	-	+	+
<i>Apalochrus</i> sp.1	-	-	-	-	-	+
Hopliini sp.	+	+	-	+	-	-
<i>Aphodius</i> sp.	-	-	-	-	-	+
<i>Scarabaeus rigosus</i>	-	-	+	+	+	+
<i>Schizonycha</i> sp.	-	-	+	-	-	-
Scarabidae sp. 1	+	-	-	-	-	-
Paederinae sp.	-	+	-	-	-	+
<i>Trigonopus</i> sp.1	+	+	-	-	+	-
<i>Zophosis boei</i>	-	-	-	+	+	+
<i>Tenebrio</i> sp.1	-	-	-	-	-	+
Tenebrionidae sp.1	+	-	-	-	-	-
Tenebrionidae sp.2	-	-	-	-	+	-
<i>Cardiotarsus acuminatus</i>	-	+	-	-	-	+
<i>Cantharis</i> sp.1	-	-	-	-	+	+
<i>Anthomyia</i> sp.	-	-	+	+	+	+
<i>Daspletis</i> sp.	+	+	+	-	+	+
Calliphoridae sp.1	-	+	-	-	-	+
Calliphoridae sp.2	-	-	-	-	-	+

Family/species name	Treatment/burned			Control/unburned		
	2WBF-T	3MAF-T	6MAF-T	2wbf-c	3maf-c	6maf-c
<i>Helomyza picta</i>	-	+	-	-	-	+
Muscidae sp.	-	-	+	+	+	-
Pipunculidae sp.1	-	+	-	-	-	+
Sciaridae sp.1	+	-	-	+	+	+
<i>Gonia</i> sp.	-	-	-	+	-	-
Tachinidae sp.2	-	-	-	-	-	+
Tachinidae sp.3	-	+	+	-	-	-
Tachinidae sp.4	+	-	-	-	-	+
<i>Forticula senegalensis</i>	-	-	-	+	+	+
<i>Anthocoris</i> sp.1	-	-	-	-	+	+
<i>Cicindela quadricuttata</i>	+	-	-	-	-	-
<i>Poophilus</i> sp.1	+	-	-	+	+	+
<i>Deraeocoris</i> sp.	-	-	-	-	+	-
<i>Agonoscelis</i> sp.1	-	-	-	-	-	+
<i>Bagrada hilaris</i>	-	-	-	-	+	+
<i>Scantius forsteri</i>	-	-	-	+	-	-
<i>Cleptria rufipes</i>	+	-	-	+	+	+
<i>Acanthaspis sanguinosa</i>	-	-	-	-	+	-
Reduviidae ?sp.	+	-	-	-	-	+
<i>Apis mellifera</i>	-	-	+	-	-	-
<i>Crematogaster peringueyi</i>	+	+	+	-	+	+
<i>Linepithema humile</i>	+	+	+	+	+	+
<i>Tetramorium capense</i>	+	+	+	+	+	+
<i>Camponotus maculatus</i>	+	-	-	-	-	-
<i>Camponotus</i> sp.2	+	+	+	+	+	+
<i>Camponotus</i> sp.3	+	-	-	-	+	-
<i>Messor capensis</i>	+	+	+	+	+	+
Formicidae sp.1	-	-	-	-	-	+
Formicidae sp.3	+	-	+	+	-	-
Halictidae sp.1	-	-	-	+	-	+
Melittidae sp.1	-	-	-	-	+	+
Melittidae sp.2	-	-	-	+	-	+
Pompilidae sp.1	+	-	+	+	+	+
<i>Campsomeriella</i> sp.1	+	-	-	+	-	-
<i>Sphex bonaspei</i>	+	-	-	+	-	+
Sphecidae sp.1	-	+	-	-	-	-
Sphecidae sp.2	+	-	-	-	+	+
Sphecidae sp.3	-	-	-	-	-	+
Tiphiidae sp.1	-	+	+	-	+	+
<i>Spintharina</i> sp.	-	+	-	-	-	-
<i>Porcello</i> sp.	-	-	+	+	-	+
<i>Microhodotermes viator</i>	-	-	-	-	+	+
<i>Centrobolus</i> sp.1	+	-	-	+	+	+
<i>Centrobolus</i> sp.2	+	+	+	+	+	+

Family/species name	Treatment/burned			Control/unburned		
	2WBF-T	3MAF-T	6MAF-T	2wbf-c	3maf-c	6maf-c
<i>Aloeides</i> sp.	-	-	-	+	-	-
Pyralidae sp.1	+	-	-	+	-	-
Tineidae sp.1	-	-	-	-	-	+
<i>Lithobius</i> sp.2	-	-	-	-	+	+
Mantidae sp.1	-	-	-	-	-	+
<i>Anomalobittacus gracilipes</i>	-	-	-	-	+	-
<i>Phalangium</i> sp.1	-	+	+	+	+	-
<i>Grylus bimaculatus</i>	-	+	+	+	+	+
<i>Cophogryllus</i> sp.	+	-	-	+	-	-
<i>Scolopendra cingulata</i>	-	-	+	-	-	-
<i>Uroplectes lineatus</i>	-	-	-	-	+	-
<i>Opisthophthalmus macer</i>	-	-	-	-	-	+
<i>Peripatopsis</i> sp.	-	-	-	+	-	-
<i>Langona</i> sp. 1	+	-	+	+	+	+
<i>Evarcha</i> sp.1	-	-	+	+	-	+
<i>Aelurillus</i> sp.1	-	+	+	-	-	+
<i>Habrocestrum</i> sp.1	-	-	-	+	-	-
Salticidae sp	-	-	-	-	-	+
<i>Steatoda</i> sp.2	-	+	+	-	+	+
<i>Steatoda capensis</i>	-	-	+	-	-	+
<i>Trabea purcelli</i>	-	-	-	-	-	+
<i>Hogna</i> sp. 1	-	-	+	-	+	+
<i>Geolycosa</i> sp. 1	-	-	+	-	+	+
<i>Proevippa schreineri</i>	-	-	+	+	+	+
<i>Proevippa</i> sp. 1	-	+	+	-	+	+
<i>Proevippa</i> sp. 2	-	-	+	-	-	-
Lycosidae sp.1	-	-	-	-	-	+
<i>Chresiona</i> sp.1	-	-	+	-	-	-
<i>Camillina</i> sp.1	+	+	-	-	-	+
<i>Camillina</i> sp.2	-	-	+	-	-	-
<i>Zelotes lightfooti</i>	+	+	-	+	-	-
<i>Pterotricha varia</i>	-	-	-	-	+	-
<i>Drassodella</i> sp. 1	-	-	+	+	+	+
<i>Platyoides</i> sp.	-	-	-	-	-	+
<i>Diaphorocellus</i> sp.1	-	-	-	-	-	+
<i>Oxyopes</i> sp.	-	-	-	+	+	+
<i>Diores youngai</i>	+	-	+	-	+	+
<i>Diores capensis</i>	+	+	-	+	-	-
<i>Synema imitator</i>	-	+	+	-	-	+
<i>Pionothele straminae</i>	+	-	-	+	-	-
<i>Galeodes</i> sp.1	-	-	-	+	-	+
<i>Galeodes</i> sp.2	-	-	-	+	-	+
<i>Rhodogastria amasis</i>	-	-	-	-	+	-
<i>Sminthurus viridis</i>	-	-	-	+	+	+

Family/species name	Treatment/burned			Control/unburned		
	2WBF-T	3MAF-T	6MAF-T	2wbf-c	3maf-c	6maf-c
<i>Paramelita nigroculus</i>	-	-	-	-	+	-

Appendix 6.3 Invertebrates recorded two weeks before fire, three and six months later after fire for the fynbos (F) vegetation adjacent to invasive alien trees (IATs). * represents known alien species.

Order/Family	Species	Before fire		After fire			
		2 weeks		3 months		6 months	
		IATs	F	IATs	F	IATs	F
Araneae							
Gallieniellidae	<i>Drassodella</i> sp.1	2	11			1	2
Gnaphosidae	<i>Camillina</i> sp.1	7	2			1	
Gnaphosidae	<i>Pterotricha varia</i>	1				3	
Gnaphosidae	<i>Zelotes lightfooti</i>	3	3	1		2	1
Lycosidae	<i>Hogna</i> sp.1	1	2	1	1		1
Lycosidae	<i>Proevippa schreineri</i>	1	3	2	2	1	5
Lycosidae	<i>Proevippa</i> sp.1	3	1			1	2
Lycosidae	<i>Proevippa</i> sp.2		1		1		1
Lycosidae	Sp.1		3				
Lycosidae	<i>Trabea purcelli</i>	6	4			1	2
Nemesiidae	<i>Pionothele straminae</i>		1				
Oxyopidae	<i>Oxyopes</i> sp.	2				3	16
Palpimanidae	<i>Diaphorocellus</i> sp.1	1	1		2	2	3
Pisauridae	<i>Rothus purpurissatus</i>	1	2		1	4	2
Salticidae	<i>Aelurillus</i> sp.1	1	3	2		4	
Salticidae	<i>Evarcha</i> sp.1	2	12				3
Salticidae	<i>Habrocestrum</i> sp.1		1				
Salticidae	<i>Langona</i> sp.1	3	13			2	3
Theridiidae	<i>Steatoda capensis</i>	6	1		1		
Theridiidae	<i>Steatoda</i> sp.2	3	6		1	6	2
Thomisidae	<i>Synema imitator</i>	2	3				2
Zodariidae	<i>Diores capensis</i>	1	2	1			
Zodariidae	<i>Diores youngai</i>	17	38	5	4	3	1
Blattodea							
Blaberidae	<i>Aptera fusca</i>		1		1		1
Blattidae	<i>Temnopteryx phalerata</i>			1	2	2	3
Coleoptera							
Anobiidae	<i>Xestobium</i> sp.	32	4			1	
Anthicidae	<i>Endomia elongates</i>	3	3				
Anthicidae	<i>Formicomus coeruleus</i>		1				
Buprestidae	<i>Achmaedera</i> sp.	12	8	3			
Carabidae	<i>Acanthoscelis ruficornis</i>		5	1			
Carabidae	<i>Anthia decemquattata</i>		7				
Carabidae	<i>Caminara</i> sp.1	2					
Carabidae	<i>Microlestia tabida</i>		10	3	12		
Carabidae	Sp.1	1	1			11	13
Cleridae	Sp.1	3	3				
Coccinellidae	<i>Cheilomenes lunata</i> *	3	1	1			
Curculionidae	Sp.1		3			1	2
Curculionidae	Sp.3		1		1		
Curculionidae	Sp.4				2		
Dermestidae	<i>Dermestes maculatus</i>	1					
Dermestidae	<i>Dermestes</i> sp.	1					1
Elateridae	<i>Cardiotarsus acuminatus</i>			1			
Meloidae	<i>Decapotoma lunata</i>			1			

Order/Family	Species	Before fire		After fire			
		2 weeks		3 months		6 months	
		IATs	F	IATs	F	IATs	F
Melyridae	<i>Apalochrus</i> sp.1						2
Mordellidae	Sp.2	1		5	4		
Nitidulidae	Sp.1	2	1	1			
Nitidulidae	Sp.2					2	1
Scarabaeidae	<i>Schizonycha</i> sp.	1	1	1			
Scarabaeidae	Sp.2					1	
Scarabaeidae	<i>Aphodius</i> sp.						6
Scarabaeidae	Hopliini sp.		1	73	51	2	1
Scarabaeidae	<i>Scarabaeus rigosus</i>	3	19		1	1	6
Scarabidae	Sp.1		4			1	
Staphylinidae	<i>Paederinae</i> sp.	3	2				7
Tenebrionidae	Sp.1	2	2	2	3		
Tenebrionidae	Sp.2		1				
Tenebrionidae	<i>Tenebrio</i> cf. <i>molitor</i>	3	1	1			
Tenebrionidae	<i>Tenebrio</i> sp.1		2		3	4	5
Tenebrionidae	<i>Trigonopus</i> sp.1		1		1	3	4
Tenebrionidae	<i>Zophosis boei</i>	2	12	5	38		10
Collembola							
Sminthuridae	<i>Sminthurus viridis</i> *	15	20	2		4	
Crustacea							
Amphipoda	<i>Paramelita nigroculus</i>	7				4	5
Dermaptera							
Forficulidae	<i>Forficula senegalensis</i>	7	3	1			5
Diptera							
Anthomyiidae	<i>Anthomyia</i> sp.	2	5			1	5
Asilidae	<i>Daspletis</i> sp.	3	3	30	47		18
Bombyliidae	<i>Exoprosopa</i> sp.			1	1		
Calliphoridae	<i>Chrysomya chloropyga</i>						1
Calliphoridae	Sp.2	1	3	2		1	4
Cecidomyiidae	<i>Cecidomyia</i> sp.1	68	16				
Heleomyzidae	<i>Helomyza picta</i>					2	
Muscidae	Sp.1	1		3		1	
Pipunculidae	Sp.1	65	10			4	2
Sarcophagidae	Sp.1		1			1	1
Sciaridae	Sp.1	14	15	3		10	
Syrphidae	Sp.1				1		
Tachinidae	<i>Dejeania</i> sp.	1	1	11	8		7
Tachinidae	<i>Gonia</i> sp.					1	
Tachinidae	Sp.2			1			
Tachinidae	Sp.3		1			5	1
Tachinidae	Sp.4					2	
Therividae	<i>Therevid</i> sp.	3					2
Hemiptera							
Anthocoridae	<i>Anthocoris</i> sp.1	1					
Cercopidae	<i>Poophilus</i> sp.1	4	2	2	6	1	1
Cicadellidae	<i>Ciphalilus</i> sp.						2
Miridae	<i>Deraeocoris</i> sp.				1		
Pentatomidae	<i>Antestia lymphata</i>					1	1
Reduviidae	<i>Acanthaspis sanguinosa</i>				1		
Reduviidae	<i>Cleptria rufipes</i>	2	3	2	4	2	2

Order/Family	Species	Before fire		After fire			
		2 weeks		3 months		6 months	
		IATs	F	IATs	F	IATs	F
Hymenoptera							
Anthophoridae	<i>Thyreus delumbatus</i>		2				
Anthophoridae	<i>Xylocopa</i> sp.						1
Apidae	<i>Apis mellifera</i>		2				1
Chalcididae	<i>Brachymeria kassalensis</i>	1	1	1			
Formicidae	<i>Camponotus</i> sp.2	24	22	9	47	2	10
Formicidae	<i>Camponotus</i> sp.3				2		
Formicidae	<i>Crematogaster peringueyi</i>	27	18	23	30	17	22
Formicidae	<i>Linepithema humile</i> *	2	12	3	11	27	31
Formicidae	<i>Messor capensis</i>		3	1	1	5	4
Formicidae	Sp.1					1	3
Formicidae	Sp.3	3	8	6	5		3
Formicidae	<i>Tetramorium capense</i>	52	19	7	3	22	21
Halictidae	<i>Nomia amabilis</i>						1
Halictidae	Sp.1	4	4	1	2	4	3
Halictidae	Sp.2				1		
Melittidae	Sp.1		1		2		1
Mutillidae	Sp.2		1				
Pompilidae	Sp.1		3	2	2		4
Pompilidae	Sp.2		1			1	1
Scoliidae	<i>Campsomeriella</i> sp.1		1		3		
Sphecidae	Sp.2	7	1			7	22
Sphecidae	Sp.3				2		
Sphecidae	<i>Sphex bonaspei</i>	1	11	1	2		6
Tiphiidae	Sp.1					1	
Isopoda							
Porcellionidae	<i>Porcello</i> sp.	5	13	44	35	6	3
Isoptera							
Hodotermitidae	<i>Microhodotermes viator</i>	90	52	30	73	98	133
Spirobolida							
Pachybolidae	<i>Centrobolus</i> sp.1	7	3	1		3	
Pachybolidae	<i>Centrobolus</i> sp.2	40	26			92	89
Lepidoptera							
Arctiidae	<i>Rhodogastria amasis</i>	1		4			
Lycaenidae	<i>Aloeides</i> sp.		1				
Noctuidae	<i>Helicoverpa armigera</i>		1				
Noctuidae	Sp.2		1				
Nymphalidae	<i>Cynthia cardui</i>		1				
Pyalidae	Sp.1	4					
Tineidae	Sp.1	2	1				
Tortricidae	Sp.1	3					
Lithobiida							
Lithobiidae	<i>Lithobius</i> sp.1		2		1		4
Lithobiidae	<i>Lithobius</i> sp.2	3	1		1		3
Mantodea							
Mantidae	Sp.3	5	3		1		
Mecoptera							
Bittacidae	<i>Anomalobittacus gracilipes</i>	12	4				
Lithobiomorpha							
Peripatopsidae	<i>Peripatopsis</i> sp.	2					

Order/Family	Species	Before fire		After fire			
		2 weeks		3 months		6 months	
		IATs	F	IATs	F	IATs	F
Opiliones							
Phalangidae	<i>Phalangium</i> sp.1	2	4	1	5		1
Orthoptera							
Gryllidae	<i>Cophogryllus</i> sp.	5	5		1	1	2
Gryllidae	<i>Gryllus bimaculatus</i>	2	2	1	3	12	33
Scolopendromorpha							
Scolopendromorphae	<i>Scolopendra cingulata</i>		1				
Scorpiones							
Buthidae	<i>Uroplectes lineatus</i>		1				2
Solifugae							
Galeodidae	<i>Galeodes</i> sp.1	2		2	4		
Total Individuals		638	523	306	438	402	569

Chapter 7 – General Conclusions and Management Recommendations

CONCLUSIONS

The results indicate that invasive alien trees (IATs) have by far greater impact on arthropod diversity than vineyards. Not surprisingly, both these vegetation types supported lower arthropod diversity compared to natural fynbos. Specific conclusions for each chapter are as follows:

Chapter 3:

- Vegetation type/land-use is an important determinant of arthropod diversity. Different vegetation types supported different arthropod diversity. In particular, both IATs and vineyards had very different arthropod assemblage composition in comparison with natural fynbos.
- Soil moisture content was not correlated with species richness despite varying significantly between different vegetation types.
- There was a significant correlation between arthropod diversity and leaf litter, with a higher diversity associated with more leaf litter.
- There was significantly lower arthropod species richness in areas of IAT infestations. Yet, arthropod species richness was significantly higher in CIATs than in IATs. Species richness in CIATs was more similar to that in fynbos. These results indicate the importance of IAT clearing. However, different invertebrate species responded differently to IAT invasion, with some species adapted to conditions in IAT patches.

Chapter 4:

- The two distinct but adjacent land-use types that appear to be important for sustaining high arthropod species richness were CIATs/fynbos, followed by CIATs/vineyard and vineyard/fynbos. Fynbos/IATs, CIATs/IATs and vineyard/IATs had the lowest arthropod diversity respectively. This suggests that IATs are associated with lower diversity, even when they were adjacent to fynbos.
- Species diversity was not necessarily higher in the ecotones compared to adjacent habitat fragments. However, some species preferred ecotones, while others were restricted to either one or the other side of the ecotone, indicating that ecotones are habitats that are suitable for certain species but not others.
- Species richness does not necessarily increase or decline with increased distance from the ecotone. However, abundance of certain species was highly influenced by the distance from the ecotone.

Chapter 5:

- IATs reduced soil compaction, while vineyards increased soil compaction. Higher soil compaction within the vineyards may be a result of many activities by farm workers and tractors. IATs had significantly lower soil compaction and arthropod diversity compared to either fynbos or CIATs vegetation types, indicating an improvement of soil compaction following IAT removal.
- There was a significant correlation between arthropod species richness and soil compaction. This implies that disturbance of soil profile had an indirect great impact on the associated surface-active arthropods.

Chapter 6:

- Surface-active invertebrate species richness declined substantially immediately after the fire, possibly through lack of resources and suitable habitats. Recovery was rapid, with some species recolonizing/reestablishing within three months.
- Invertebrate diversity within IATs was indirectly affected by the fire, which occurred within the adjacent fynbos. Moreover, both species richness and abundance were higher in the fynbos than in IATs following fire, with some invertebrate taxa no longer represented six months after fire. This suggests that certain invertebrate taxa are fire-tolerant, while others are not. Nevertheless, there was similar arthropod assemblage composition before and after the fire in IATs, whereas there was a different assemblage composition before after the fire in the fynbos. Fire had a significantly higher impact on the arthropod assemblage within the fynbos than in IATs.
- Habitat fragments that escape fire, or are only partially burned, may continue to support suitable habitats and be a source habitat for certain invertebrates, thereby facilitating overall assemblage recovery.

Overall, the results indicated that high arthropod biodiversity was linked with a structured, stable ecosystem, and that low biodiversity was an indication of a disturbed ecosystem. Moreover, the results confirmed the findings from the savanna biome (Druce et al., 2007) that the more heterogeneous habitats (here, fynbos) supported the greatest arthropod diversity and abundance. These results supported the fact that habitat type (natural vs disturbed) is a key determinant of species diversity in the fragmented landscape (Ewers et al., 2007). This leads into better understanding of ecological interactions to develop agricultural policies and strategic plans for sustainable conservation management. The findings here provide some principles on surface-active arthropod responses towards IATs and agricultural activities.

MANAGEMENT RECOMMENDATIONS

IATs coupled with agricultural activities and other anthropogenic disturbances are rapidly increasing the prevalence of habitat edges in CFR landscapes. Therefore, sustainable biodiversity conservation will require co-operation among conservation biologists, farmers, and landowners to actively plan the future of CFR landscapes. In such hugely transformed landscapes, conservation strategies become of major concern. It is necessary to select the type of optimal landscape and prioritize it for conservation. It is also crucial for biodiversity conservation, that policy makers consider developing a monitoring programme suitable for terrestrial arthropods, to complement the aquatic monitoring programme such as the River Health Programme and Working for Wetlands. I strongly agree with Thuiller (2007), that large-scale, long-term experiments and observations are required to efficiently monitor biodiversity. Where feasible, the South African government must provide incentives to farmers who benefit the environment by maintaining eco-friendly agricultural landscapes. Such practice is reversing biodiversity decline in Europe and North America, where nearly 4 billion Euros are paid annually through agri-environmental schemes to farmers for making environmental improvements to their land (Donald and Evans, 2006).

The protection and conservation of the fynbos should be considered as a priority. It is also important to be careful with the introduction of IATs into natural areas, or into areas where the species is not yet present, because of the potential of the species to become invasive. Ramula et al. (2008) developed general guidelines for invasive plant management, which can be applied to rapidly growing plant invasions in the fynbos vegetation of the Western Cape Province.

With the current rapid vegetation fragmentation in the CFR, critical examination of ecological patterns and processes near habitat edges, or ecotones, should be considered for biodiversity conservation planning.

Comprehensive check-lists of biota should be compiled before any conversion of native vegetation for agricultural activities, and follow-up surveys done to monitor changes in populations. With such adequate background information, specific requirements of rare and threatened species may be significantly catered for, and hence loss of species can be minimized. Within agricultural patches, it is also important to use methods of pest

suppression that have minimal impacts on adjacent natural areas. Unspoiled natural areas within agricultural land mosaics can serve as biodiversity banks and thus provide higher opportunities for conserving biodiversity than forms of land use that are more intensive, such as wall-to-wall forestry and urbanisation.

It is critical that conservation strategies enforce biodiversity stewardship not only on public land but on private property as well. Integration of environmental and economic policy is another crucial component of designing a more effective biodiversity conservation strategy, since the current economic system is not efficiently accounting for biodiversity. Carefully designed policies can improve economy, while meeting local needs and concerns, and therefore resolve conflict with economic development. Considering the current biodiversity crisis worldwide, it is time to act appropriately and effectively. However, lack of information on invertebrates in most of South Africa's vineyard areas is of major concern from a conservation perspective. It is therefore recommended that further surveys continue, although focusing on designated transects. For minimal resources, the surface-dwelling invertebrate surveys should take place in spring (late August–October) or summer (November–January), rather than winter (May–July). Numbers of insects change markedly throughout the year, with species numbers often continuing to decrease from spring into summer, after which, there is a large drop in the winter months. Surveying those invertebrates in winter would potentially lead to an underestimate of maximum invertebrate species richness, and would be significantly affected by rainfall in the CFR.

Biodiversity conservation within vineyard areas may be improved through an integrated management system. Integrated conservation management in vineyards ensures that representative and sensitive habitats, species and ecosystem process are maintained within the existing vineyard areas. Management and design of vineyards significantly affect the contribution that these areas make to the conservation of biodiversity. Poor planning can cause excessive fragmentation of habitats, which influences invertebrate diversity. It is therefore important to incorporate native habitat patches to reduce the effect of vineyards on biodiversity. Here, some invertebrates showed preference for a particular habitat vegetation type, while some invertebrates used various types of vegetation to varying degrees.

South Africa has taken preliminary steps to conserve biodiversity, primarily through the creation of protected areas and the enforcement of laws that protect threatened species and certain specific habitats. These are important and necessary steps for a new biodiversity conservation strategy, but they are not enough to address the current biodiversity crisis. South Africa has substantial Environmental Acts for biodiversity conservation. However, laws are as good as the government's ability to enforce them. This ability is to some extent lacking, with much of the fynbos being recklessly exploited, and requiring more effective management. The results here suggest that ecotones should also be considered as part of that effective management.

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