

**Effects of vineyard management and landscape  
context on taxonomic diversity and interaction networks  
of flower-visiting insects in the Cape Floristic Region  
biodiversity hotspot**

**by**

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## General Abstract

Both taxonomic diversity and diversity of species interaction networks are experiencing declines as a result of agricultural intensification at habitat and landscape scales. Reversing this trend is a key conservation issue, particularly for important functional groups such as flower-visiting insects and the networks within which they interact. This is of great concern in regions of high conservation priority such as the Cape Floristic Region (CFR), known for its high level of floral and faunal endemism and exceptional species turnover. Holistic approach to conservation in agricultural landscapes involves both preservation of natural land and wildlife friendly management of the farm land to achieve conservation targets. The value of these extensive management approaches is yet to be fully assessed, especially in perennial systems such as vineyards.

I examined here the effects of vineyard management and landscape context on species richness and abundance of flower-visiting insects and their species interaction networks. Possible taxon specific effects were verified. I also investigated whether vineyards under organic and conventional management homogenized insect-flower interaction networks and whether vineyards with different management practices vary in patterns of species turnover. I sampled flower-visiting insects and their interactions in organic and conventional vineyards, and in natural reference sites. Inclusion of natural reference sites enabled me to make management recommendations for patches of natural vegetation in CFR agricultural landscape.

Statistical models showed taxon-specific benefit of organic farm management, and of landscape (distance to natural habitat). There was benefit to monkey beetles (Scarabaeidae) but not to bees (Apidae). Organic vineyards had a higher number of insect-flower interactions than conventional ones, but vineyards under the two types of management were similar in terms of other important network indices. However, networks of the vineyards were more

nested than the natural sites, indicating that they may be potentially more stable to perturbation and random extinctions. Multivariate dispersion tests revealed insect-flower interaction networks were not homogenized by both organic and conventional vineyards across the landscapes. I also found, through additive partitioning, that organic and conventional vineyards were similar in terms of species turnover of bees and flowering plants.

The findings of this study provide heuristic value to current debates on the value of vineyard habitats for insect conservation. Both organic and conventional vineyards that promote sustainable management of the non-crop floral vegetation between vineyard rows are potential solutions for conservation of flower-visiting insects and their interactions. Also, attention has to be paid to the quality and connectivity of the natural habitat patches that are within CFR agricultural landscape. Site specific management and assessment of the value of these landscape elements is important. Management approaches such as carefully controlled burning may be beneficial, as the CFR natural vegetation is a fire-driven community.

## Algemene Opsomming

Taksonomiese diversiteit en diversiteit van spesies-interaksie netwerke ondervind beide afnames as gevolg van landboukundige intensifikasie op habitat en landskap skaal. Om die neiging terug te swaai, is 'n sleutel bewaringsaangeleentheid, veral vir belangrike funksionele groepe soos blom-besoekende insekte en die netwerke waarbinne hulle op mekaar inwerk. Dit is van groot kommer in streke met hoë bewaringsprioriteite soos in die Kaapse Floristiese Streek (KFS), bekend vir sy hoë vlak van plant- en dierendemisme en buitengewone spesies kentering. 'n Holistiese benadering tot bewaring van landboukundige landskappe behels beide die bewaring van natuurlike land en natuurlewe-vriendelike bestuur van die plaasgrond om bewaringsdoelwitte te bereik. Die waarde van hierdie ekstensiewe bestuursbenaderings moet nog volledig bepaal word, veral in meerjarige sisteme soos wingerde.

Ek het die uitwerkings van wingerdbestuur en landskapsamehang op spesiesrykheid en volopheid van blombesoekende insekte en hulle spesies interaksie netwerke ondersoek. Moontlike takson-spesifieke uitwerkings is nagegaan. Ek het ook ondersoek ingestel of wingerde onder organiese en gebruiklike bestuur ooreenstemmende insek-blom interaksie netwerke met wingerde met verskillende bestuurspraktyke in patroon van spesies kentering gewissel het. Ek het blom-besoekende insekte en hulle interaksies in organiese en konvensionele wingerde, asook in natuurlike verwysingsgebiede gemonster. Insluiting van natuurlike verwysingsgebiede het my in staat gestel om bestuursvoorstelle vir gebiede van natuurlike plantegroei in KFS landboulandskappe voor te stel.

Statistiese modelle toon takson-spesifieke voordeel van organiese plaasbestuur en van die landskap (afstand van natuurlike habitat) self. Daar was voordeel vir bobbejaankewers (Scarabaeidae), maar nie vir bye (Apidae) nie. Organiese wingerde het 'n groter getal insek-

blom interaksies as konvensionele wingerde gehad, maar wingerde onder beide tipes van bestuur was soortgelyk in terme van ander belangrike netwerk aanduiders. Netwerke van wingerde was egter meer geklomp dan natuurlike gebiede wat aandui dat hulle potensieel meer stabiel betreffende versteuring en lukrake uitsterwings is. Multivariate verspreidingstoetse het aangetoon dat insek-blom interaksie netwerke by beide organiese en konvensionele wingerde oor landskappe nie eenvormig was nie. Ek het ook bevind deur aanvullende verdeling dat organiese en konvensionele wingerde gelykwaardig was in terme van spesies kentering van bye en blomplante.

Die bevindings van hierdie studie verskaf heuristiese waarde tot huidige debatte oor die waarde van wingerdhabitats vir insekbewaring. Beide organiese en konvensionele wingerde wat volhoubare bestuur van die nie-gewas plantegroei binne wingerdrye bevorder, is moontlike oplossings vir die bewaring van blom-besoekende insekte en hulle wisselwerkings. Bykomend moet aandag gegee word aan die kwaliteit en verbindings van en tussen natuurlike habitat gebiede binne die KFS landboulandskap. Plekspesifieke (plaaslike) bestuur en bepaling van die waarde van hierdie landskapelemente is belangrik. Bestuursbenaderings, soos noukeurig beheerde brand, mag voordelig wees aangesien die KFS natuurlike plantegroei 'n vuurgedrewe gemeenskap is.

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## Chapter 1 – General Introduction

### **Biodiversity in agro-ecosystems**

Modern agriculture involves large scale and mechanized management of plant and animal growth for human use (Encarta 2008). Agriculture based on this definition and the attendant practices, provides the basic support for human livelihood, even in the modern world of the 21<sup>st</sup> century. Furthermore, the global human population is predicted to increase to around nine billion people (United Nations 2003) an increase of 50% in the next 50 years, and is anticipated to increase pressure on agricultural production, that might lead to conversion of about another one billion hectares of natural habitat to agricultural production (Tilman *et al.* 2001) to meet the demand for fuel, food and fibre. Agricultural land already dominates the landscape in Europe, with Britain having about 77% of the land area i.e 18.5 million hectares, under agricultural production in 2001(DEFRA 2002). There is also a forecast that the synergistic effects of agricultural intensification and climate change will result in unparalleled transformation of agricultural landscapes (Jackson *et al.* 2007), and consequent simplification of the farmed environment, as well as creation of semi-artificial ecosystems that require constant human intervention to stabilize dynamics of their activities (Altieri 1999). These modern systems rely mostly on high chemical inputs and mechanical disturbance to achieve mass production of food. However, their ability to balance productivity and support for biodiversity in the long term has been questioned (Krebs *et al.* 1999; Gurr *et al.* 2003; Hole *et al.* 2005).

There have been concerns for loss of biodiversity, and increased water pollution, soil erosion, as well as deteriorating landscapes and food safety (DEFRA 2002). This has made sustainable land use and conservation of biodiversity, ecosystems and natural resources, especially in agriculturally-dominated regions a pressing global agenda (United Nations 2000). Biodiversity conservation is particularly important, because of its significant

contribution to the overall well being of ecosystems by playing a key role in stability, structure and productivity of ecosystems (Larsen 1995; Tilman *et al.* 1996; Tscharntke *et al.* 2005). Agricultural intensification in recent years has brought about expanded cultivation of annual and perennial crops, leading to highly fragmented landscapes that have lost important refuge habitats which are known to support biodiversity in agricultural landscapes (Stoate *et al.* 2001; Tilman *et al.* 2001; Benton *et al.* 2003; Ockinger & Smith 2007).

The United Nations Earth Summit at Rio de Janeiro in 1992 recognised biological diversity as the variability among living organisms from all sources of life i.e. terrestrial and aquatic ecosystems and the ecological complexes that generate their interactions, this includes diversity within species (genetic diversity), between species (organismal diversity) and of ecosystem (ecological diversity). Genetic, organismal and ecological diversity are seen as indicators for environmental quality, especially in areas heavily influenced by human activities such as intense cultivation. All stakeholders must bear in mind that sustainable development is closely linked biodiversity and should be approached concomitantly in the face of current global change (Johnson 1993).

Biodiversity in agricultural systems is well valued, and investigations into interactions between biological diversity and agricultural systems have gained much recognition as a growing field of research (Altieri 1999). Considerable research effort has been put into activities taking place at the interface between agriculture and the natural environment to underpin the need for balance between agricultural production and ecological stability which is the role of biological diversity (Tscharntke *et al.* 2005). Biodiversity has been identified as playing the following key roles in the ecosystem; nutrient cycling, pest and diseases control, soil protection and pollination. (Díaz *et al.* 2006; Perrings *et al.* 2006). This is achieved by interactions between flora and fauna, which co-occur in space and time (Tylianakis *et al.* 2010) and are involved with biological processes. Sustainability of the agro-ecosystem is

therefore closely linked to biodiversity, and high levels of biodiversity are needed to sustain productivity in these systems in the long term, without external interference (Tylianakis *et al.* 2010). When the biological internal links are large and diverse enough, processes of self renewal and adaptation to environmental change are made possible. This ability is lost in intensive agricultural systems, which result in simplified narrow and artificial systems that have to be supported by constant human intervention and external inputs (Altieri 1999).

Agricultural intensification at different spatial scales has brought about a significant decline in agricultural land biodiversity (Robinson & Sutherland 2002; Tscharntke *et al.* 2002, 2005). For example there is currently a wealth of evidence to show that agricultural intensification over the last four decades has led to widespread declines in European farmland bird populations (Krebs *et al.*, 1999; Donald *et al.* 2006) and reductions in abundance and diversity of several plant and invertebrate taxa (Donald 1998; Wilson *et al.* 1999; Preston *et al.* 2002). In spite of the well publicized adverse effect of agricultural intensification, it has been argued that a simplified perspective of agricultural land use as just habitat destruction is incomplete, as agricultural management has been shown to enhance biodiversity although this is less acknowledged by biased traditional opinions that only pristine ecosystems can rescue current biodiversity loss (Pimental *et al.* 1992; Jackson & Jackson 2002; Rosenzweig 2003; Tscharntke *et al.* 2005). Bridging these gaps to conserve biodiversity in the face of current global change will therefore involve a holistic approach to conservation that involves both managed and pristine land (Ings *et al.* 2009; Tylianakis *et al.* 2010).

### **Decline of pollinators**

Pollination of plants by insects is a critical ecosystem service that is of utmost importance to both crop and wild plants species worldwide. About 60-80% of the 250 000 flowering plants

rely on animal pollinators that range from mostly insects, small mammals, bats, to birds for pollination, (Kremen *et al.* 2007) of which insects are the largest group of animal pollinators.

Insects are responsible for 80-85 % of all pollinated commercial hectares with fruits, vegetables oil seeds, legumes and fodder representing about one-third of global food production (Williams 1996; Klein *et al.* 2007). Pollinators also sustain wild plant communities which provide food and shelter for several other types of wildlife (Ashman *et al.* 2004; Aguilar *et al.* 2006). Thus, plant pollination by insects being one of the most widespread and important ecosystem services, is essential to human health, global food webs and biodiversity protection (Black *et al.* 2007) especially in Africa where floral diversity is known to depend on pollinators (Eardley 2001).

Insect pollinators are diverse, and are represented by a wide range of taxa, including Lepidoptera (moths and butterflies), Hymenoptera (bees and wasps), Coleoptera (beetles) and Diptera (flies). These are made up of several thousands of species. According to Michener (2000), bees alone are represented by about 20 000 species worldwide, with an estimated 4 000 species in United States. Of this very diverse group, the honey bee (*Apis mellifera*) has been reported to contribute the most to crop pollination on a commercial scale (Williams 1994; Klein *et al.* 2007). However, recent evidence is suggesting that the importance of the honeybee for pollination service might have been over rated and that wild pollinators are more important (Breeze *et al.* 2011)

In recent times, human activities or human-induced activities such as urbanization, intensive agriculture, invasive plants, climate change, as well as diseases and parasites, have been shown to impact negatively on diversity and abundance (success) of insect pollinators (National Research Council 2006, Ricketts *et al.* 2008; Schweiger *et al.* 2010). It has been argued that these multifaceted aspects of global change such may be acting in synergy to drive the current loss of insect pollinators (Didham *et al.* 2007; Potts *et al.* 2010). The

strongest evidence of global pollinator crisis are from Europe and North America, and calls have been made for studies on other continents to enable a global mapping of the ubiquity of this phenomenon and to seek possible solutions (Banaszak 1995; National Research Council, 2006; Potts *et al.* 2010). Recent studies are however documenting the value of pollination services as well as the dynamics that shape pollinator communities and the ecosystem services they provide in South African agricultural landscapes (Allsopp *et al.* 2008; Carvalheiro *et al.* 2010, 2011).

In agricultural lands, decreasing landscape heterogeneity and habitat fragmentation have been associated with reduced pollination of crops and non-crop plants among other impacts on ecological functions (Cunningham 2000a; 2000b; Richards 2001; Kremen *et al.* 2002; Tilman *et al.* 2002; Tschardtke *et al.* 2005). Hence, the need for conservation and restoration efforts to sustain pollinator diversity in agricultural and natural ecosystems (Eardley 2001). Proximity of farmland to natural habitat is very important for the diversity of pollinators (Bergman *et al.* 2004; Kremen *et al.* 2004; Morandin & Winston 2006; Hendricks *et al.* 2007; Winfree *et al.* 2007, Carvalheiro *et al.* 2010). These natural areas serve as source of pollinators for crop pollination and also as refugia for pollinators that are able to recolonize degraded agricultural areas (Ockinger & Smith 2007).

In South Africa, the dry western part is known for high levels of diversity and endemism of aculeate hymenoptera (Gess & Gess 1993) and the Cape Floristic Region is a known centre of bee diversity (Donaldson 2002; Kuhlmann 2009). This region is also home to over two-thirds of South Africa's pollinator dependent deciduous fruit sector with insect pollination service and managed pollination valued at US\$358.4 and US\$312.1 million respectively (Allsopp *et al.* 2008). There have been few reports of factors such as introduced species of plants and insects as threats to populations of indigenous pollinators in South Africa (Rebelo 1987; Henning *et al.* 1997) with calls being made for research into impact of land use for

pollinators that can provide information on best practices for conservation of pollinators in agricultural landscapes in this region (Donaldson 2002).

### **Conserving insect diversity across landscape**

On a large scale, landscapes are often fragmented and include elements such as matrixes, habitat patches and corridors (Forman & Gordon 1986) as a result of which many populations are spatially structured in discrete local populations (Wu & Loucks 1995). Fragmentation and habitat reduction can limit species richness and relative abundance of species (Carleton & Taylor 1983; Diamond 1984). The challenge across these landscape mosaics is to reduce the probability of local extinctions and this is overcome by exchange of individuals among the patch populations (metapopulation). This provides a rescue effect that allows for recolonization even when there are local extinctions (Fahrig 2003).

Predictions have shown that various characteristics of land patches within landscapes such as patch size, isolation and disturbance frequency/intensity all influence species diversity in terms of richness, population density and survival probability (Fahrig & Jonsen 1998; Fahrig 2003). Several authors have predicted that patch isolation has a negative effect on population density and species richness, because the rate of rescue and recolonization are reduced as isolation of patches widens (Kareiva 1987; Kruess & Tschardtke 1994; Roland & Taylor 1995). However, in like manner, increasing disturbance frequency is usually thought to decrease population density and survival probability, hence species richness (Grossman *et al.* 1982; Sousa 1984; Norton *et al.* 1995). An approach to insect conservation should take into consideration both insect assemblages and diversity of behavioral interactions across different species and within the life stages of the same species (Fisher 1998). This will therefore involve a habitat based conservation approach that emphasizes functional species interactions.

Conserving biodiversity across the landscape involves the use of tools such as information about presence or absence of insect species. This involves species which can be used as monitors or indicators of environmental and biodiversity change or health (Disney 1986; Kremen 1992; Kremen *et al.* 1994). Values of essential and unique insect behaviour, functional roles and ecological processes such as pollination, predation and other crucial ecosystem services (Fisher 1998) may also be of importance. Study of plant-pollinator interactions as an important ecosystem process can be used to promote our understanding habitat loss and other effects of landscape fragmentation. This is possible because pollinators are known to be influenced by the geometry (such as size, density and shape) of plant populations (Handel 1983). Linhart & Feinsinger (1980) demonstrated this when they investigated the effect of island-induced reductions in pollinator diversity on generalized and specialized plants species. It has been proposed that pollinators, their population dynamics and the systems within which they interact should be priority group for case studies in the conservation and sustainable use of agricultural biodiversity. Bees are particularly known to be potential keystone indicator species because they can be easily monitored (Eardley 2001).

### **Viticulture in South Africa**

Viticulture in South Africa dates back to the 17<sup>th</sup> century from French origin. It mainly takes place at a latitude of 34° South in the Coastal region of Western Cape. However there are fragments of viticultural practices further east, towards the north, the Klein Karoo, Olifant river and Orange river areas which are warmer and drier (see Figure 1.1

[www.vineyardvarieties.com](http://www.vineyardvarieties.com)).

The Cape Floristic Region one of the global biodiversity hotspots (Myers *et al.* 2000), is influenced by two oceans and has a great diversity of topography and mesoclimate conditions which influence viticulture (WOSA 2009). All over the world, identification of a



viticultural terroir is of utmost importance to wine grape farming. According to Rouget *et al.* (2003), agriculture has transformed approximately a quarter of the landscape in the CFR (CFR) and this will continue to grow over the next 20 years. 90% of South Africa's 110 000 hectares under grape vine farming is in the Western Cape (Rogers 2006), where wine production is of great economic importance (Cape Wine Academy 2002). Even though vineyards, as a form of agro-ecosystem, are known to provide suitable habitats for high number of insect species (Isaia *et al.* 2006), loss of heterogeneity and biodiversity may result if vineyards are subject to intensification (Ruthsatz & Haber 1982; Harms *et al.* 1987). South Africa, like other parts of the world, has, however, established some agri-environment schemes aimed at sensitizing farmers towards sustainable agricultural practices with regards to biodiversity. Integrated Production of Wine Scheme (IPW) and Biodiversity and Wine Initiative (BWI) were established to promote viticultural practices aimed towards reducing the impact of the wine industry on the natural environment, preventing further habitat loss due to vineyard expansion and increasing the amount of protected natural habitats in vineyard landscape (Biodiversity and Wine Initiative 2011; Tromp 2006). Biodiversity and Wine Initiative (BWI) has successfully conserved 112 500 hectares i.e about 100 % of the total area under vineyard through the support of South African grape vine farmers.



Figure 1.1 Wine regions in South Africa (From [www.wineweb.com](http://www.wineweb.com))

### **Organic viticulture**

The drive for sustainable agro-ecosystems has brought about more extensive systems, such as organic farming, which is targeted towards mitigating the negative effects of modern agriculture and to enhance biodiversity (Krebs *et al.* 1999; Reganold *et al.* 2001; Tybirk *et al.* 2004).

Organic viticulture as described by The Oxford Companion to Wine is “a system of grape growing which is based not on the plant but on an attempt at rational management of the living parts of the soil, regarded as a complex, living environment interacting closely with the flora and fauna which inhabit it, while respecting biological cycles and the environment” (Smart & Williams 1994). Despite favorable climatic conditions, the development of organic viticulture in South Africa, has been slow and fragmented, due to years of political isolation (Waldin 1999, 2004). However, environmental concerns in the past few years have led to an increase in conversion to organic viticultural practices.

**Rationale for this research project**

There are various agri-environment schemes such as Countryside Stewardship Scheme (CSS) and Rural Stewardship Scheme (RSS) in Europe, as well as The Integrated Production of Wine scheme (IPW) and the Biodiversity and Wine Initiative (BWI) in the CFR, all which are aimed at sensitizing farmers towards sustainable agricultural practices with regards to biodiversity. Organic farming has been shown to provide a clear advantage over such schemes in that the farm as a whole is subject to the organic standards, rather than to the limited areas on conventional farms that may be exposed to environmental management under these schemes (Hole *et al.* 2005). This is termed the holistic approach, in which the farm is seen not merely as a production system, but rather as an ecosystem aimed at enhancing the resilience and stability of the system while reducing the need for human interference (Rigby & Cáceres 2001; Madge 2007).

Though proponents of organic farming have elucidated the environmental benefits that can be delivered by these management systems, substantial evidence to back up their claims have been questioned (Hole *et al.* 2005). The Soil Association (2000) has emphasized the need for further research into the quantification of the total biodiversity benefits of widespread organic farming.

Krebs *et al.* (1999) suggest that biodiversity in agro-ecosystems depends on both landscape heterogeneity and farm management. Contrasting findings may arise when differences between farming systems result from associated differences in landscape rather than directly from farming practices (Bengtsson *et al.* 2005). Few studies carried out at landscape scale reported modification of the effect of organic farming by landscape variation on plants (Roschewitz *et al.* 2005), on diversity of spiders in vineyards (Isaia *et al.* 2006) as well as diversity of bees, butterflies, carabids and spiders (Weibull *et al.* 2000, 2003; Kremen *et al.* 2002; Schmidt *et al.* 2005; Holzschuh *et al.* 2007). Most of these studies have organic

farming in annual crop systems, with few studies looking at perennial systems (Bruggisser *et al.* 2010). Also, to date few studies have looked at interaction networks or food webs of parasitoid-host and pollinator plant interactions in between organic and conventional farming systems (Macfadyen *et al.* 2009; Carvalheiro *et al.* 2010, 2011; Power & Stout 2011). According to WOSA (2009), the European Union, which currently makes up the majority of South Africa's export wine market, is working towards enforcing environmentally sustainable wine production systems on countries wishing to export wine to the EU. Hence, these research efforts will be particularly beneficial in the CFR, where the organic wine sector is at its infancy and on a small scale, with only a few producers managing their vineyards organically or biodynamically (Rosenthal Duminy 2004), owing to dearth of research findings to ascertain whether organic farming in vineyards can deliver anticipated biodiversity benefits (Gaigher 2008).

This study therefore seeks to assess the impact of vineyard management practices and landscape composition on flower-visiting insects and their interaction networks and to assess the potential of extensive vineyard management to mitigate loss of insect pollinator diversity and their interactions in the CFR. This objective will be achieved by asking the following questions:

### **Research questions**

1. What is the effect of vineyard management practices (organic vs. conventional) on species richness and abundance of endemic insect pollinators?
2. What is the effect of landscape context on species richness and abundance of endemic insect pollinators?
3. Are there interactive effects of vineyard management and landscape context on endemic pollinators?

4. Are there taxon specific effects of vineyard management and landscape context on endemic pollinator taxa?
5. Are there differences in insect-flowering plant interaction networks between organic and conventional vineyards and natural vegetation areas in terms of important network properties?
6. Do organic and conventional vineyards homogenize insect-flowering plant interaction networks across CFR agricultural landscape?
7. Do organic and conventional vineyards homogenize flower-visiting insect assemblage networks across CFR agricultural landscape?
8. Do organic and conventional vineyards homogenize networks of important pollinators of indigenous plants across CFR agricultural landscape?
9. What are the patterns of species turnover in space and time in organic and conventional vineyards compared to natural vegetation areas?

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## **Chapter 2 - Endemic pollinator response to organic vs. conventional farming and landscape context in the CFR**

### **Abstract**

The effects of organic management and landscape context on two highly endemic and important pollinator taxa (bees and monkey beetles) were studied in the Cape Floristic Region (CFR) biodiversity hotspot, where a quarter of the land is intensively managed mostly for grape production. This functional group was chosen as there is concern worldwide over pollinator decline, particularly in regions of high levels of endemism of both flora and fauna such as the CFR. Species richness and abundance of bees and monkey beetles were sampled and compared between organic and conventional vineyards, with natural vegetation as reference habitat. Species richness of monkey beetles, but not the species richness of bees, benefited from organic compared to conventional vineyard management. Our findings support the hypothesis that the effects of landscape context and management may be taxon dependent. Monkey beetles have been suggested as strong indicators of disturbance. These beetles are among the most important pollinator guilds in the CFR, as they help to sustain the endemic vegetation of this region. Protection of the remaining natural habitat in close proximity to farmland is an imperative conservation strategy in this region.

### **Introduction**

Fragmentation of the landscape, isolation and reduction of patch size, as well as intensive management practices, are among the various aspects of agriculture that impact significantly on biodiversity (Robinson & Sutherland 2002; Tilman *et al.* 2001; Tschamtker *et al.* 2002, 2005). Declines in species richness and abundance have been reported for various taxa (Tim *et al.* 2003), including pollinators (Kearns & Inouye 1997; Kearns *et al.* 1998; Kevan 1999;

Aizen & Feinsinger 2003; Biesmeijer *et al.* 2006; Goulson *et al.* 2008), across various spatial and temporal scales (Tschardtke *et al.* 2007). Improving the value of agricultural land for biodiversity should therefore involve approaches that decrease the intensity of management, while having the potential to promote species rich habitat through increased resource availability (Tschardtke *et al.* 2007).

This drive for sustainable land use in agricultural areas is of utmost importance for conservation, particularly in regions of the world such as the Cape Floristic Region (CFR), characterized by high species richness and endemism of flora and fauna (Myers *et al.* 2000). This region is known as a centre of diversity and endemism for several pollinator taxa, including bees (Apidae) and monkey beetles (Scarabaeidae) (Donaldson 2002; Kuhlmann 2009), which play vital roles in the region's plant speciation (Waterman *et al.* 2009; Van der Neit & Johnson 2009). The CFR has a unique global identity as the only place on earth where a centre of bee diversity and of plant diversity hotspot coincide (Kuhlmann 2005, 2009). In terms of vegetation type, the region is dominated by fynbos, a sclerophyll hard leaf vegetation type adapted to low soil fertility, frequent fires and winter rainfall. The fynbos is made up of four growth forms known as restoids (reed-like plants), ericoids (heath-like shrubs), geophytes (bulbous herbs) and sclerophyllous proteoids (tall shrubs with large leaves) (Cowling & Richardson 1995). Also among the CFR vegetation types is renosterveld, a low shrub layer, comprised mainly of renosterbos *Elytropappus rhinocerotis*, with a ground layer of grasses and seasonally active bulbs (Conservation International 2010). The Biodiversity and Wine Initiative (2009) reported that the CFR is under increasing threat from agriculture, urban development and invasion from alien plant species, with only 9% of the original extent of renosterveld and lowland fynbos remaining.

Organic management and other extensive farm management approaches improve the value of agricultural land by supporting higher biodiversity than intensive management approaches, in annual crop fields (Letourneau & Goldstein 2000; Holzschuh *et al.* 2007, 2010; Kovács-Hostyánszki *et al.* 2011a, 2011b) at both the habitat and landscape spatial scales. Contrary results, showing no significant benefits from organic farming to biodiversity relative to conventional farming, have also been reported (Hald & Reddersen 1990; Younie & Armstrong 1995; Weibull *et al.* 2000; Kleijn *et al.* 2001). This makes the debate on the value of organic farming open-ended, and calls for more research to give a clearer understanding of the value of organic farming for biodiversity conservation (Hole *et al.* 2005). In particular, there is a paucity of knowledge on the effect of organic farming on biodiversity in perennial crop systems (Bruggisser *et al.* 2010), especially those involving an endemic-rich biota. There is growing interest among viticulturists to assess biodiversity at the farm and landscape scales to ascertain the impact of management practices and the character of landscapes on biodiversity, as well as to identify best practices for wine production that are not at the expense of biodiversity (Earthwatch 2010). However, variable results have been obtained from the few studies which investigate management and landscape effects on biodiversity in vineyards and other perennials (see for example Costello & Daane 1998; Hadjicharalampous *et al.* 2002; Steffan-Dewenter & Leschke 2003; Isaia *et al.* 2006; Eilers & Klein 2009; Bruggisser *et al.* 2010). Brittain *et al.* (2010) found no significant difference in benefits for pollinators and pollination service in organically managed compared to conventional vineyards. A lack of significant effect of organic management on three trophic levels (plant, insect herbivore and predator) was ascribed to the intermediate disturbance hypothesis in a study carried out in Swiss vineyards (Bruggisser *et al.* 2010).

The effects of landscape context (proportion of natural habitat and distance to natural habitat) and management on biodiversity can be taxon dependent (Weibull & Östman 2003;

Batáry *et al.* 2008; Carvalheiro *et al.* 2010). While bee (Apidae) pollination is very important for about 80% of crops globally (Klein *et al.* 2007), beetles (Coleoptera) pollinate 88% of all flowering plants (Buchmann & Nabhan 1996). In evolutionary terms, beetles are among the earliest flower visitors (Bernhardt & Thien 1987) and they still remain the largest set of pollinators today, owing to their numbers (United States Department Agriculture (USDA) 2010). In the Fynbos biomes of the Western and Northern Cape Provinces of South Africa, monkey beetles (Scarabaeidae, Hopliini) are among the most important pollinators (Picker & Midgley 1996; Colville *et al.* 2002). They are endemic to these regions and are effective pollinators (Henning 1985; Goldblatt *et al.* 1998).

Here we compare the impact of local management and the landscape context of organic and conventional vineyards on the endemic-rich biodiversity using natural vegetation as a reference, for the two most important insect pollinator taxa (bees and monkey beetles) in the CFR, where agriculture, especially vine cultivation, is the main driver of transformation of about a quarter of the landscape. The following hypothesis were tested: i) organic vineyards have a greater species richness and abundance of bees and monkey beetles than do conventional vineyards, ii) the extent of geographical isolation of vineyards from natural vegetation patches and proportion of natural patch around vineyards has a significant effect on the species richness and abundance of bees and monkey beetles, and, iii) the effect of management and landscape variables is taxon dependent.

## **Methods**

### ***Research area and experimental design***

The study area was the Western Cape Province, South Africa (centered at 33°S, 18°E). It is in a Mediterranean-type ecosystem in the CFR, where vine cultivation is a major activity (Rogers 2006). Six pairs of organic and conventional vineyards were selected, along with six

adjacent patches of natural habitat made up of both fynbos and renosterveld, which were the reference sites (Figure 2.2). All organic vineyards were certified 'organic' by certification bodies such as Bio-Dynamic and Organic Certification Authority (BDOCA), Quality Certification Services (QCS) and Société Generale de Surveillance (SGS). They were planted with cover crops such as *Hypochoeris radicata*, *Raphanus raphanistrum*, *Erodium moschatum*, *Bidens pilosa*, *Avena fatua* and *Vicia* spp. between the vine rows, and were given no agrochemical treatment except fungicides. Features of organic and biodynamic practices are shown in Table 2.1. The six conventional vineyards were registered with the Integrated Production of Wine Scheme (IPW). Conventional vineyards were all treated with various agrochemicals at the IPW recommended rates (see IPW agrochemical coding from [ipw.co.za](http://ipw.co.za)). Organic and conventional vineyard pairs were within 0.1-1 km apart and were similar in terms of average size, vine height, between row and within row space (Table 2.2). Each natural site was within the same locality as the vineyard pair with which it was compared; within a range of 0.3-4 km. Natural sites were 5-20 ha in size.

### ***Insect and vegetation sampling***

There were two sampling periods (late August and late October in the spring season of 2009) during which insects and flowering plants on all vineyards and natural habitat sites were sampled. Two trapping methods were used, pan and window trapping, as these are well suited for sampling flower-visiting insects and aerial fauna in open landscapes (Duelli *et al.* 1999). Twelve yellow pan traps (1000 ml each) and two window-intercept traps (window dimension 0.5 x 0.5 m) were placed at each site (Figure 2.1). The traps were half filled with water, and a few drops of detergent were added to break the surface tension of the water. The traps were placed in pairs 20 m away from the edge of each field. Traps in a pair were 2 m apart and pairs were at least 20 m apart to prevent interaction. The traps were operational in the field for five days during each sampling period, after which the insects were removed and

preserved in 75% ethanol. Traps were placed at the height of the vegetation in each study site. Bees and monkey beetles were sorted, pinned and identified to species or, where not possible, to genus and morphospecies levels. All samples were pooled for both insect trapping methods per site. Survey flowering plants in each field were done along two 50 m transects in the centre of each field. Plants that were flowering within 2 m x 2 m swathes at 5 m intervals along the transects were recorded, during each insect sampling period. Transects in vineyards were between the vine rows. The average number of flowering plant morphospecies recorded for all plots within each field was calculated.



Figure 2.1 (a) Yellow pan trap and (b) window intercept traps placed at vegetation height for insect sampling.

### ***Landscape survey***

The proportion of different land use types around each study site were mapped in field surveys. This was done within a radius of 1000 m, 500 m and 250 m from the centre of each field for all 18 study sites. Topographic maps from the Western Cape Department of Agriculture, Elsberg, were used for field surveys, and estimation of different land use

proportions was done with Geographic Information System, (Arc GIS 9, Environmental Systems Research Institute, ESRI, California). The proportion of surrounding natural habitat around the organic, conventional and natural sites (referred to here as proportion of natural habitat) were not significantly different (Kruskal Wallis ANOVA), Kruskal-Wallis chi-squared = 2.2105,  $df = 2$ ,  $P = 0.3311$ ). Distance from the centre each vineyard from to natural habitat was also calculated with ArcGIS. Habitat area and isolation are meaningful explanatory variables for predicting response of insect communities to landscape structures (Zabel & Tschamtkke 1998; Steffan-Dewenter & Tschamtkke 2000). Proportion of natural habitat at 1000 m, 500 m and 250 m radii were highly correlated (Table 2.2) but distance to natural habitat was not correlated with proportion of natural habitat at any of the radii ( $P > 0.05$ ). Hence Proportion of natural habitat at 1000 m radius and distance to natural habitat were used in further analysis.



Table 2.1 Comparison of Organic and Biodynamic farming practices

Management practice	Use in Organic/Biodynamic farming
i. Reduced tillage	Used in both farming systems
ii. Restricted use of agrochemicals	Used in both farming systems
iii. Sensitive management of non-crop vegetation	Used in both farming systems
iv. Organic instead of synthetic fertilizers	Used in both farming systems
v. Application of lunar and cosmic rhythms	Unique to biodynamic farming
vi. Use of biodynamic preparations (fermented herbal preparations as compost additives and field sprays Carpenter-Boggs <i>et al.</i> 2000)	Unique to biodynamic farming

Table 2.2 Characteristic of the twelve (six pairs) organic and conventional vineyards that were sampled

	Organic	Conventional
Average field size (ha)	4.85 ± 1.76	4.43 ± 1.37
Range of vine height (m)	1.5 – 2.0	1.5 – 2.0
Range of between row spaces (m)	2.5 – 3.5	2.5 – 3.5
Range of within row spaces (m)	1.2 – 1.4	1.2 – 1.4
Irrigated	2 fields	All 6 fields
Fertilizer use	Organic fertilizer	Synthetic fertilizer

Table 2.3 Pearson's product-moment correlation scores and *P* values for proportion of natural habitat at 1000 m, 500 m, and 250 m radii around the study sites

	1000 m		500 m	
	Correlation score	<i>P</i> value	Correlation score	<i>P</i> value
1000 m	-	-	-	-
500 m	0.918	$2.129e^{-07}$	-	-
250 m	0.845	$1.968e^{-05}$	0.927	$8.637e^{-08}$

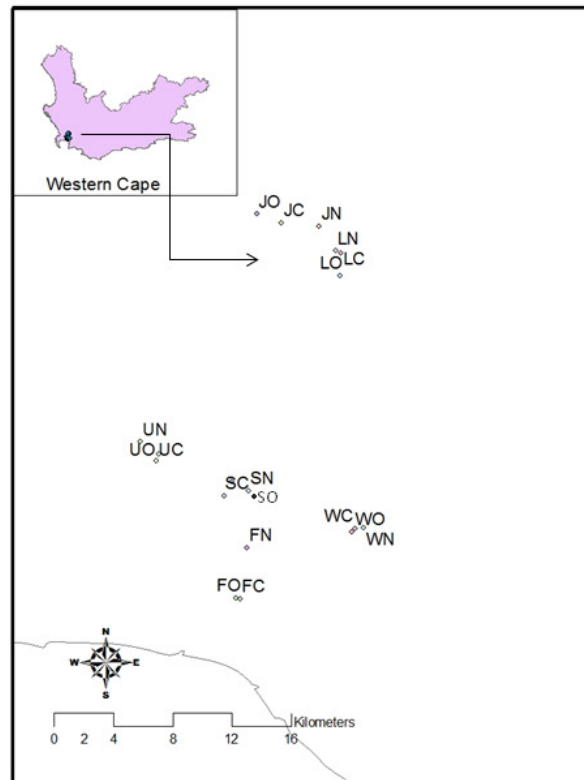


Figure 2.2 Study sites in the Western Cape region of South Africa. JO=Joostengerg Organic vineyard, JC=Joostengerg Conventional vineyard, JN=Joostenberg Natural vegetation, LN=Laibach Organic vineyard, LC=Laibach Conventional vineyard, LN=Laibach Natural vegetation, WO=Waterford Organic vineyard, WC=Waterford Conventional vineyard, WN=Waterford Natural vegetation, FO=Firgroove Organic vineyard, FC=Firgroove Conventional vineyard, FN=Firgroove Natural vegetation, UO=Uitzicht Organic vineyard, UC=Uitzicht Conventional vineyard, UN=Uitzicht Natural vegetation, SO=Spier Organic vineyard, SC=Spier Conventional vineyard, SN=Spier Natural vegetation.

### ***Data analyses***

Data were analyzed with generalized linear mixed effects models (Poisson error distribution). Management, landscape variables (proportion of natural habitat and distance to natural habitat), morphospecies richness of flowering plants (referred to here as ‘flower diversity’) and all their possible interactions were fixed factors. To account for the locality based nature of the sampling design, locality was used as random factors in the models. Two sets of models (each of which is here referred to as a ‘model-set’) were computed for each taxon of bees and monkey beetles. One model-set was computed by incorporating data from all the sites. The other model-set was computed excluding data from the natural sites. This allowed for the testing of distance to natural site as a landscape variable. The value of this variable for the natural sites was always zero since they were the reference point for this measurement. Therefore, including data from the natural sites will imply co-variance with the management variable. All analyses were done in R (R development core team 2010). Model simplification was done by first removing non-significant interactions of fixed factors ( $P > 0.05$ ) and then non-significant main effects were removed. Model assumptions of normality, heterogeneity and independence were validated by plots of normalized residuals against fitted values and against each independent variable (Zuur *et al.* 2007).

### **Results**

A total of 4 754 monkey beetle individuals in 15 species, and 1 910 bee individuals in 25 species were sampled throughout the study. The average bee abundance was not significantly different between organic and conventional vineyards and natural sites ( $P > 0.05$ ). In contrast, the mean species richness of bees was significantly higher at the natural sites ( $t = 2.728$ , d.f. = 10,  $P = 0.021$ ) compared with both organic and conventional vineyards (Figure 2.3). The landscape variables ‘proportion of natural habitat’ and ‘distance to natural habitat’ and all

their possible interactions with management variable had no significant effect on the abundance and species richness of bees ( $P > 0.05$ ). These results were consistent between the model-set that included data from the natural sites and the model-set where natural site data were excluded.

The model explaining the variation in monkey beetle species richness showed the significant effect of management and distance to natural habitat, but a non significant effect of proportion of natural habitat. Species richness of monkey beetles was significantly higher in organic vineyards and natural sites compared to conventional vineyards ( $t = 3.40$ , d.f. = 7,  $P = 0.011$ , Figure 2.4a). Abundance of monkey beetles was highest in natural sites ( $t = 2.215$ , d.f. = 10,  $P = 0.05$ , Figure 2.4b), but not significantly different between organic and conventional vineyards ( $P > 0.05$ ). In the model-set where data from natural sites were excluded to test effect of distance to natural habitat, species richness ( $t = -5.063$ , d.f. = 4,  $P = 0.007$ ) and abundance ( $t = -3.677$ , d.f. = 4,  $P = 0.021$ ) of monkey beetles decreased with increasing distance from natural habitat in both organic and conventional vineyards (Figure 2.5a and 2.5b), with 50% of species richness and abundance lost at 1650 m and 500 m respectively (see Figure 2.5 for equations).

Flower diversity was significantly higher in natural sites than in organic or conventional vineyards ( $t = 3.5760$ , d.f. = 8,  $P = 0.0072$ ) but was not significantly different between organic and conventional vineyards, as well as in the models explaining variation in bee and monkey beetle species richness and abundance ( $P > 0.05$ ). All fixed factor interactions had no significant effect in the models.

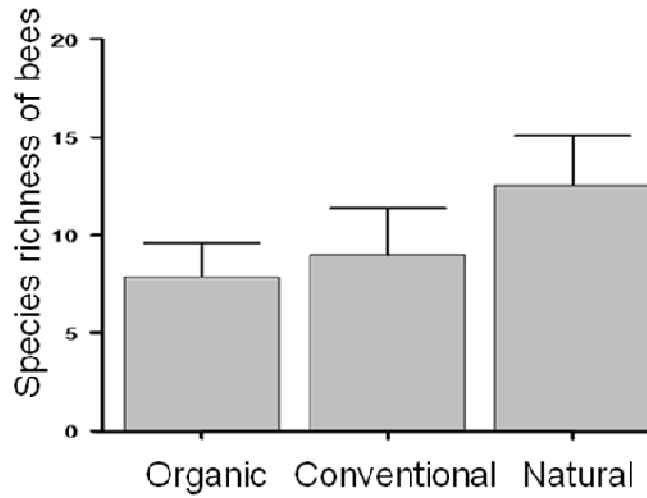


Figure 2.3 Mean ( $\pm$  SE) species richness of bees sampled in different vineyard management types and natural sites.

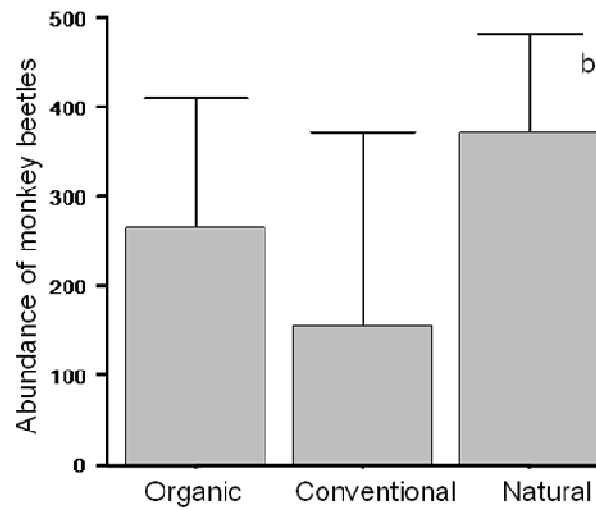
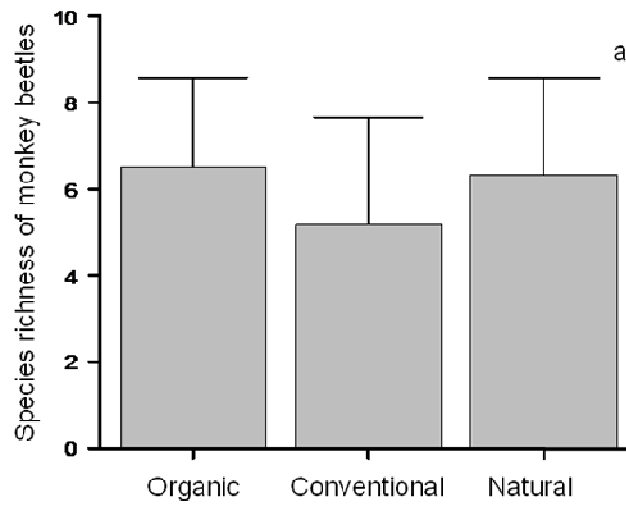


Figure 2.4 Mean ( $\pm$  SE) (a) species richness and (b) abundance of monkey beetles sampled in organic and conventional vineyards and natural sites.

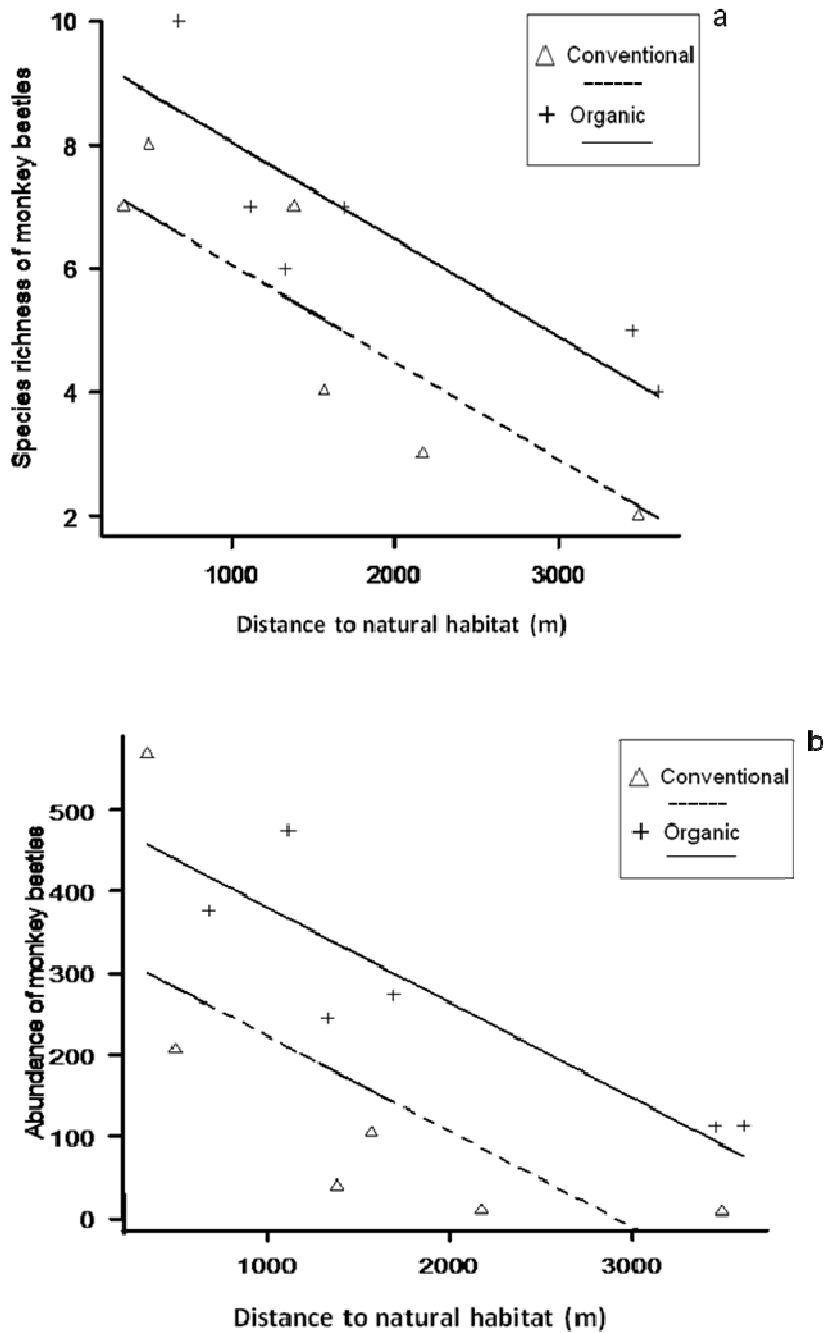


Figure 2.5 Mean ( $\pm$  SE) (a) species richness and (b) abundance of monkey beetles, with distance to natural habitat in organic and conventional vineyards. Regression lines shown for significant values at 5% level. Model equations are: Species richness =  $7.6494 - 0.001579 \times$  Distance to natural habitat (m), Abundance =  $340.5515 - 0.1174 \times$  Distance to natural habitat (m)



Table 2.4

List of bees and monkey beetles sampled during this study

Family	Genus	Species
Andrenidae	<i>Andrena</i>	<i>notophila</i>
Apidae	<i>Apis</i>	<i>mellifera</i>
Apidae	<i>Xylocopa</i>	<i>rufitarsis</i>
Apidae	<i>Allodape</i>	<i>tridentipes</i>
Apidae	<i>Pasites</i>	morphospecies
Apidae	<i>Amegilla</i>	<i>fallax</i>
Apidae	<i>Anthophora</i>	<i>diversipes</i>
Apidae	<i>Anthophora</i>	<i>wartmanni</i>
Apidae	<i>Tetraloniella</i>	<i>junodi</i>
Colletidae	<i>Scrapper</i>	<i>heterodoxus</i>
Colletidae	<i>Colletes</i>	morphospecies 1
Colletidae	<i>Colletes</i>	morphospecies 2
Halictidae	<i>Lasioglossum</i>	morphospecies 1
Halictidae	<i>Lasioglossum</i>	morphospecies 2
Halictidae	<i>Lasioglossum</i>	morphospecies 3
Halictidae	<i>Lasioglossum</i>	morphospecies 4
Halictidae	<i>Halictus</i>	morphospecies 1
Halictidae	<i>Patellapis</i>	morphospecies 1
Halictidae	<i>Patellapis</i>	morphospecies 2
Melittidae	<i>Melitta</i>	<i>arrogans</i>
Megachilidae	<i>Megachile</i>	<i>fulvohirta</i>
Megachilidae	<i>Lithurge</i>	<i>spiniferus</i>
Megachilidae	<i>Megachile</i>	<i>malangensis</i>
Megachilidae	<i>Hoplitis</i>	sp1
Megachilidae	<i>Pseudoanthidium</i>	<i>tuberculiferum</i>
Scarabaeidae	<i>Heterochelus</i>	morphospecies 1
Scarabaeidae	<i>Heterochelus</i>	morphospecies 2
Scarabaeidae	<i>Heterochelus</i>	morphospecies 3
Scarabaeidae	<i>Heterochelus</i>	morphospecies 4
Scarabaeidae	<i>Heterochelus</i>	morphospecies 5
Scarabaeidae	<i>Heterochelus</i>	morphospecies 6
Scarabaeidae	<i>Heterochelus</i>	morphospecies 7
Scarabaeidae	<i>Anisomyx</i>	<i>ursus</i>
Scarabaeidae	<i>Anisomyx</i>	<i>ditus</i>
Scarabaeidae	<i>Anisomyx</i>	morphospecies 1
Scarabaeidae	<i>Peritrichia</i>	morphospecies 1
Scarabaeidae	<i>Peritrichia</i>	morphospecies 2
Scarabaeidae	<i>Peritrichia</i>	morphospecies 3
Scarabaeidae	<i>Pachynema</i>	<i>crassipes</i>
Scarabaeidae	<i>Platycheilus</i>	morphospecies

## Discussion

From the few results to date in perennial cropping systems, organic farming does not always have significantly different beneficial effects for biodiversity when compared with conventional farming. While some studies report significantly different benefits (Isaia *et al.* 2006; Gaigher & Samways 2010), others found non-significantly different effects (Brittain *et al.* 2010; Bruggisser *et al.* 2010). In this study, natural sites had higher bee species richness than organic or conventional vineyards, which were similar in species richness. Brittain *et al.* (2010) reported similar non-significant effects of organic management and landscape context (proportion of uncultivated land) on bees in Italian vineyards. Also, abundance and diversity of flower-visiting insects were found to be similar in organic and conventional mango orchards in South Africa (Carvalho *et al.* 2010). Despite the non-significant effect of landscape variables on abundance and species richness of bees, the higher species richness in the natural sites found here emphasizes their importance as insurance habitats in agricultural landscapes. The protection of the remaining natural vegetation in the CFR has been shown to be more important than organic farming for the conservation of surface-dwelling arthropods in this region (Gaigher & Samways 2010). To optimize arthropod conservation, preference for native vegetation over agricultural land was also stressed in a meta-analysis by Attwood *et al.* (2008). However, consideration has to be given to connectivity of these habitats within the landscape mosaic (Brittain *et al.* 2010) to ensure the delivery of the biodiversity benefits they hold.

Interestingly, organic and conventional vineyards had similar bee species richness. Similarly, high diversity of plants, grasshoppers and spiders, representing three trophic levels, was not found in organic farming in Swiss vineyards. However, higher grasshopper species richness was found in conventional rather than organic vineyards (Bruggisser *et al.* 2010). It has been suggested that perennials such as vineyards may require intermediate levels of disturbance provided by more intensive management systems, such as conventional management in this case, to provide conditions that allow the co-existence between species

supported by low disturbance and stress-tolerant species (supported by high disturbance), hence supporting greater biodiversity (Huston 1979; Reganold *et al.* 1987; Townsend & Scarsbrook 1997; Mackey & Currie 2001; Bruggisser *et al.* 2010).

Taxon dependent effects of landscape context and management on biodiversity have been reported (Weibull & Östman 2003; Bengtsson *et al.* 2005; Batary *et al.* 2008). This study examined the effect of management and landscape on monkey beetles, a pollinator taxon which has not been well studied in this regard. Contrary to the results obtained for bees, monkey beetle species richness was significantly higher in organic than in conventional vineyards, as well as in landscapes closer to natural habitat than in those that were more isolated. Monkey beetles have been suggested as strong indicators of disturbance due to their close association with both vegetative and edaphic factors (Colville *et al.* 2002), this may explain the management effects found in this study. Insect mobility has been shown to mediate effects of landscape features on insect diversity (Weibull & Östman 2003; Carvalherio *et al.* 2010). Arguably, bees and monkey beetles are similar in their dispersal abilities, however, monkey beetles are slower dispersers compared to some large rapid flying bees (Goldblatt & Bernhardt, 1990; Goldblatt *et al.* 1995; Goldblatt *et al.* 1998). This may possibly explain the significant effect of distance obtained here for monkey beetles but not for bees. However, proportion of natural habitat in the landscape did not contribute significantly to the model. It has been shown that in biodiversity-rich regions, small fragments of natural habitat of 8% cover within 2000 m radius sustained population of flower-visiting insects such that there was no significant effect of natural habitat proportion on flower visitors, despite effects of distance to natural habitat (Carvalheiro *et al.* 2011). In this study, the lowest proportion of natural habitat obtained was 2% cover in a radius of 1000 m and this was shown not to be limiting for the abundance and species richness of these endemic pollinators. However, caution has to be exercised, as the monkey beetle model shows that when these natural habitats are too isolated their benefits may be lost in

agricultural landscapes. *Carvalho et al.* (2010) also noted that when agricultural fields are too large, effect of pollinator declines may be felt in even regions of high biodiversity.

Protection of the remaining natural habitat in the CFR region should therefore be a priority. Efforts are being made by the BWI, in partnership with vineyard owners, to conserve sizeable areas of natural habitat around vineyards. Having natural habitats in close proximity to farmland may increase possibility of connectivity of those habitats in the agricultural landscape, with promising benefits for both crop and non-crop vegetation. Holistic conservation efforts involving both protection of the remaining natural vegetation and environment-friendly agricultural practices that promote within farmland floral resources should therefore be encouraged by all stakeholders.

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## **Chapter 3 – Effects of vineyard management on mutualistic insect-flower interaction networks**

### **Abstract**

Few studies to date have used food web structures to assess performance of agri-environment schemes, despite their potential to provide more detailed clues to the dynamics shaping biodiversity patterns in agricultural landscapes. Insect-flower interaction networks were employed as a tool to examine biodiversity patterns between organic and conventional vineyards compared to natural reference sites. Networks were compared by analyzing important web structure indices from organic and conventional vineyards, as well as natural vegetation control sites in five different landscapes in the Cape Floristic Region (CFR). Results showed that number of insect-flower interactions was higher in organic vineyards than in conventional vineyards. Other network indices were similar between organic and conventional vineyards. There is need for expanded knowledge on the effect of different management practices for the conservation of biodiversity in these perennial systems particularly in the endemic rich CFR. Vineyard ecosystems, when well managed in this region, have the potential to support conservation of flower-visiting insects and their interaction networks, by serving as refugia, reservoirs or alternative sites with rich resource benefits. In this way, these vineyards contribute significantly to the emerging holistic approach of conserving interaction biodiversity in natural and managed systems.

### **Introduction**

The use of mutualistic interaction networks and webs has been stressed as vital to the science and practice of conservation in recent years. This is especially so with globally beneficial interactions such plant-pollinator networks which have played a major role in maintaining the Earth's biodiversity (Bascompte *et al.* 2006; Klein *et al.* 2007). It has been shown that land use change, habitat modification and other potential drivers of global change may alter

network properties even in the absence of biodiversity loss (Tylianakis *et al.* 2007; Laliberté & Tylianakis 2010). Services such as pest-control and pollination, which are critical for maintaining functioning ecosystems, are synthesized from complex networks of interactions of species in plant and animal communities. Macfadyen *et al.* (2009) suggested that attention should be given to investigating how these species interactions are structured within communities and how community interactions shape functionality.

Approaching biodiversity conservation from a holistic perspective should involve focus on both species and the networks within which they interact (Tylianakis *et al.* 2010). These interaction networks are of three types depending on the organisms and interactions they contain: traditional webs, host-parasitoid/predator webs (both categorized as antagonistic interactions) and mutualistic webs (e.g. plant-pollinator) in which both partners benefit (Ings *et al.* 2009). For interactions between species to occur, it is important that the species must not only be present, but also to co-occur in space and time (Tylianakis *et al.* 2010). Both the identity and frequencies of interactions that form the building block of an interaction network are vital in characterizing the structure of ecological networks (Laliberté & Tylianakis 2010). Frequency of interaction in mutualistic networks has been shown to be one of the key determinants of plant reproductive success and to be important in the prediction of pollination service by a pollinator species to a plant population with which it interacts (Morris 2003; Vázquez *et al.* 2005)

Globally, attention has been drawn to the potential benefits of organic farming in mitigating biodiversity loss at both habitat and landscape scales (Letourneau & Goldstein 2001; Clough *et al.* 2007; Holzschuh *et al.* 2007; Holzschuh *et al.* 2008). Species interactions have been found to be higher in organic systems with prey-predator-parasitoid networks (Macfadyen *et al.* 2009). However, whether these higher species richness and interactions translate into more benefits in terms of ecosystem services remains uncertain (Macfadyen *et al.* 2009). Most studies that assess the benefits of organic farming in terms of species richness and interactions have been limited to annual cropping systems, with increased calls to extend

investigations to perennial systems, as it is uncertain if patterns obtained in annual systems can be generalized (Marshall *et al.* 2003; Bruggisser *et al.* 2010; Sanguaneko & León 2011).

Vine (*Vitis vinifera*) cultivation, being a major feature of the Cape Floristic Region (CFR) (Rogers, 2006), has contributed to the agriculturally driven fragmentation of the highly endemic flora and fauna of this region (Fairbank *et al.* 2004). Efforts are being sought to mitigate the impact this may have on biodiversity (Biodiversity and Wine Initiative, 2010). As reported in other parts of the world (Sanguaneko & León 2011), weeds in the form of cover crops and resident plant species, which may be annual or perennial, are integral to vineyard rows in the CFR. Sustainable weed management in vineyards involves controlling competitive weeds while leaving other vegetation without jeopardizing crop production. This creates favourable habitat for several organisms through acting as refugia, reservoirs or alternative sites that hold resource benefits for biodiversity (Marshall *et al.* 2003; Sanguaneko & León 2011). We therefore aim to test the following hypothesis: i) vineyards harbour rich composition of insect-flower interactions and are potential surrogate sites. ii) the composition and structure of insect-flower interaction networks in organic vineyards are more diverse and stable to habitat loss and random extinction than those of conventional vineyards. iii) the composition and structure of insect-flower interaction networks in the natural habitat are more diverse than those of vineyards.

## **Methods**

### ***Research area***

The study area was in the Stellenbosch wine region in the Western Cape Province (WCP) (33 55 S, 18 51 E) in the CFR, an area of 87,892 km<sup>2</sup>, notable for its high endemism among its flora and fauna (Rouget *et al.* 2003). For a Mediterranean-type ecosystem, the CFR has an exceptional floral richness, with  $\leq 100$  unique species of vascular plants within an area of 1000 km<sup>2</sup>. This compares favourably with many tropical rainforests, and is unparalleled in most other Mediterranean-type ecosystems (Goldblatt & Manning 2002).

Five pairs of organic and conventional vineyards, which differed in farm management practices, were selected. Close to each pair, a natural fynbos or renosterveld vegetation area was used as a natural reference site. All sets of three sites were within 5 km to each other. All organic vineyards were managed either organically or biodynamically for at least four years and were officially certified as such by Bio-Dynamic and Organic Certification Authority (BDOCA), Quality Certification Services (QCS) and Société Generale de Surveillance (SGS). The conventional vineyards were registered with the Conventional Production of Wine Scheme (IPW). Biodiversity concerns have led to environmentally friendly approaches to management of vegetation within vineyards which has resulted in various species of annual and perennial cover crops, and weeds, becoming an integral part of the vineyards.

#### *Sampling of insect-flower interactions*

Sampling was in spring, early October to mid December 2010, with an average minimum temperature of 15 °C and minimum rainfall of 18 mm. Plants in vineyard rows and native vegetation in natural sites were flowering during this period. A 100 m X 50 m plot was demarcated at each site. From the 100 m length of each plot, I divided the plots into three, giving three subdivisions of approximately 30 X 50 m each. Three 50 m-long transects (one from each subdivision) were randomly sampled during each visit to a site. I walked through each transect for 20 min during each visit. Transect walks were done on days with no rain, no wind (beaufort scale 0-1) and minimal cloud (< 5 %) cover at all sites. Records of insect-flower interactions were taken within 2 m swathe along each transect. Flower-visiting insect samples were collected to confirm identification to species level in the laboratory. Sampling was done weekly at each site for a total of eight weeks within the sampling period. Insect taxa sampled here were those well known to be involved in mutualistic interactions with flowering plants i.e. bees, butterflies, hoverflies and beetles.

Vegetation sampling was done along the same transects where insect-flower interactions were recorded. Species richness of flowering plants and abundance of floral units were recorded in six 1 m<sup>2</sup> quadrats at 10 m intervals along each 50 m transect. Flower units



were defined in terms of bee movement such that a medium-sized bee has to fly to move between separate flower units/heads (Dicks *et al.* 2002).

### ***Data analyses***

Insect-flower interactions data matrices were set up for each study site. Web structure for each site was plotted, and twenty-four web structure indices were computed for each site with plotweb and network level functions in bipartite package in R (version 2.12, R development core team 2010).

I analyzed some qualitative and quantitative web structure indices to determine variations in interaction networks from the three land use management types: organic and conventional vineyards, and natural vegetation, here referred to as ‘land use(s)’. Qualitative indices chosen were; insect and flowering plant species richness and abundance, and number of insect-flower interactions. Quantitative indices weight interactions by their frequency, which reduces their sensitivity to sampling intensity and network size (Dormann *et al.* 2009; Power & Stout 2010). The following quantitative indices were chosen for analysis: quantitative connectance, generality, vulnerability, nestedness, compartmentalization. The effect of land use on each of the network indices was tested with linear mixed effects models with land use as a fixed factor. Locality was used as a random factor to account for the locality based sampling design. Models were validated, with plots of standardized residuals and fitted values, to confirm normality, heterogeneity and independence.

### **Results**

The 15 food webs from the study sites contained a total of 45 species of flower-visiting insects, 41 species of flowering plants, 1 692 insect-flower interactions and 191 links.

Bipartite network plots showed the higher trophic level community (insects) at the top level and the lower trophic level community (plants) at the bottom level. These are shown for all organic and conventional vineyards and natural sites sampled (Figure 3.1 – 3.5).

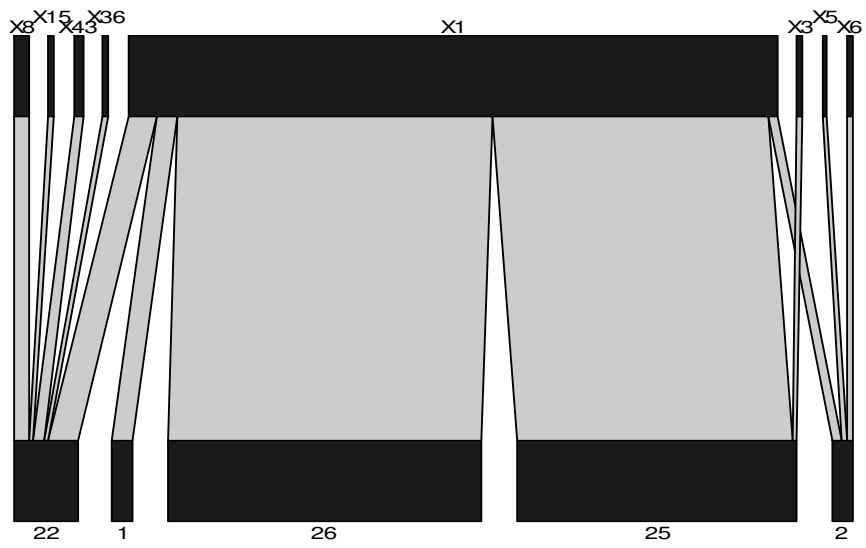
Abundance ( $P = 0.110$ ) and species richness ( $P = 0.978$ ) of flower-visiting insects as well as abundance ( $P = 0.480$ ) and species richness of flowering plants ( $P = 0.533$ ), were not significantly different between organic and conventional vineyards. The models also showed significantly higher number of insect-flower interactions in organic compared to conventional vineyards, ( $t = 3.477$ , d.f. = 7,  $P = 0.016$ , Figure 3.6). The observed difference in number of interactions was positively related to abundance of flowering plants ( $t = 3.171$ , d.f. = 7,  $P = 0.023$ ). None of the quantitative indices was significantly different between organic and conventional vineyards in the models ( $P > 0.05$ , Table 3.1). Species richness of flowering plants was significantly higher in the natural sites ( $t = 5.728$ , d.f. = 8,  $P = 0.000$ ) compared to both organic and conventional vineyards. In terms of the quantitative indices, model showed that compartmentalization and connectance in the natural sites were similar to organic and conventional vineyards ( $P > 0.05$ , Table 3.1). However, nestedness was significantly lower in the natural sites compared to organic and conventional vineyards ( $P = 0.039$ , Table 3.1).

Table 3.1 Mean ( $\pm$  SE) of network indices of organic and conventional vineyards and natural vegetation obtained from Linear Mixed Effects Models

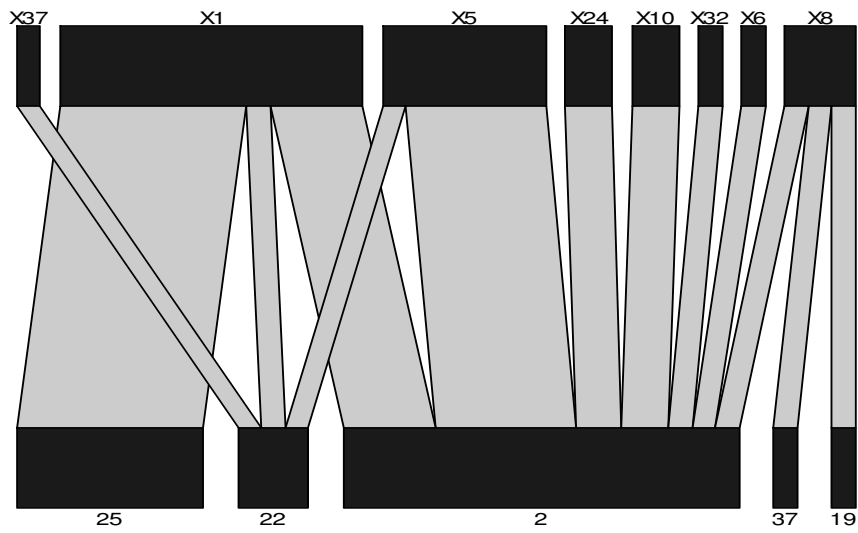
	Organic	Conventional	Natural	d.f	t	<i>P</i> value
Connectance	0.30 $\pm$ 0.06	0.29 $\pm$ 0.06	0.24 $\pm$ 0.06	-	-	NS
Compartmentalization	1.75 $\pm$ 0.31	2.18 $\pm$ 0.26	2.71 $\pm$ 1.18	-	-	NS
Nestedness	23.4 $\pm$ 7.04	21.86 $\pm$ 13.4	6.12 $\pm$ 8.71	8	-2.47	0.039*
Vulnerability	2.06 $\pm$ 0.88	2.64 $\pm$ 0.98	2.12 $\pm$ 0.61	-	-	NS
Generality	1.9 $\pm$ 0.57	2.30 $\pm$ 0.80	1.78 $\pm$ 1.11	-	-	NS

\*significant at the 5% level, NS not significant

Organic



Conventional



Natural

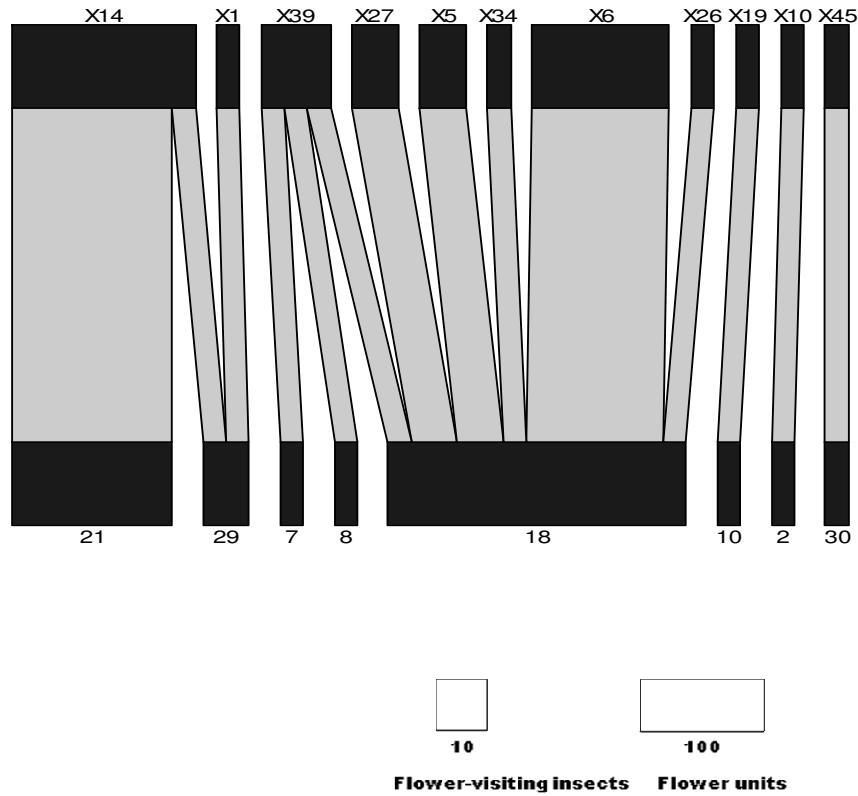
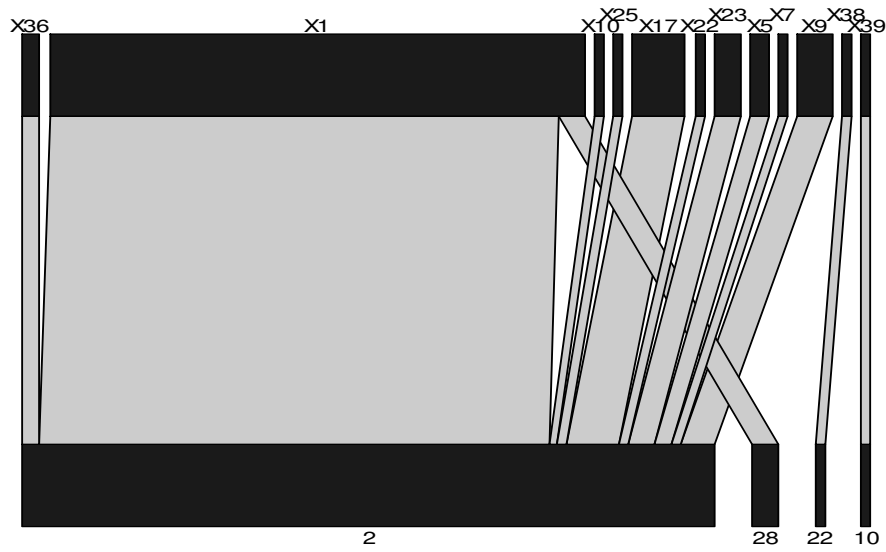
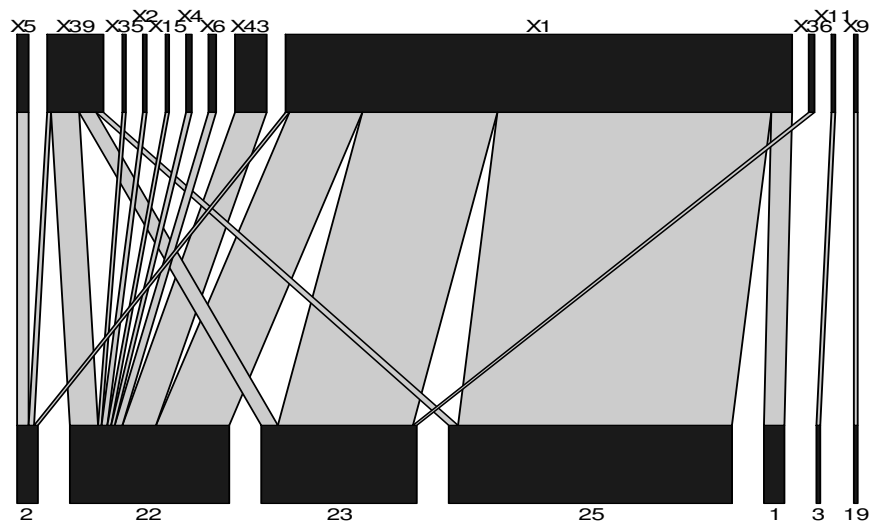


Figure 3.1 Insect-flower interaction networks of organic and conventional vineyards and a natural site from one of the landscape (named Laibach). The top levels are the insect species which visit plants species at the bottom level. The lines between the two levels represent the interactions between the two levels and the thickness of the line shows the frequency of interactions. The width of the upper and lower rectangles indicate the quantity of insects and plants involved in visitations respectively. Insect and plant codes (numbers) interpreted in Tables 3.2 and 3.3 respectively.

Organic



Conventional



Natural

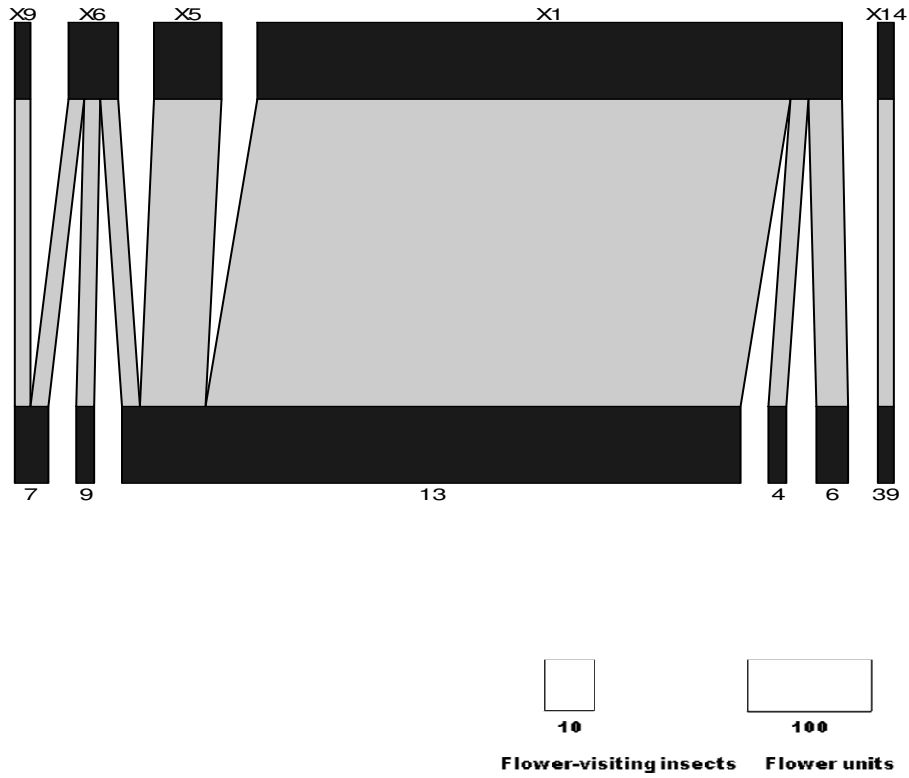
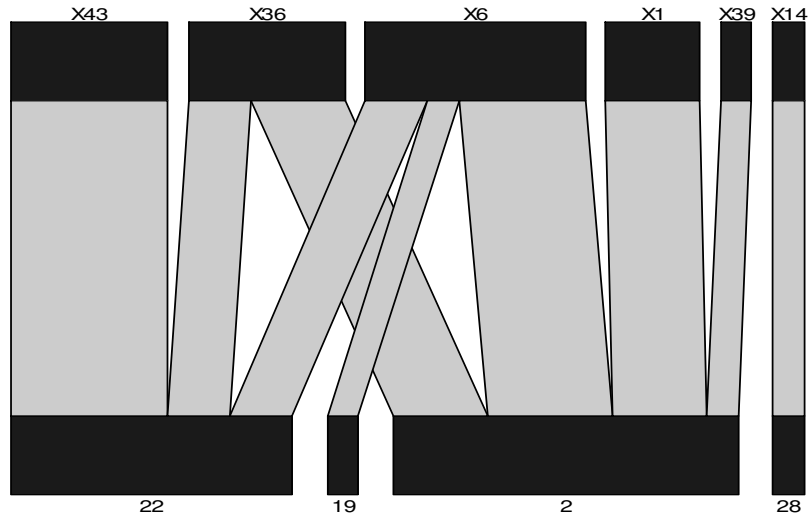
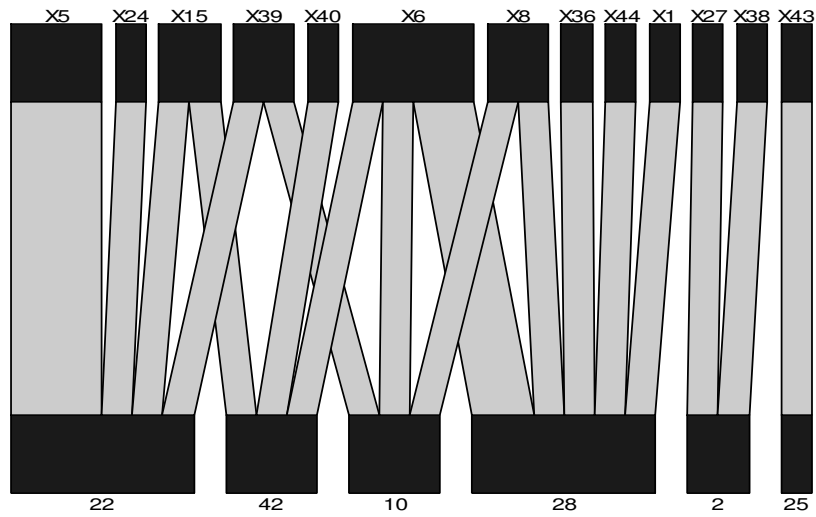


Figure 3.2 Insect-flower interaction networks of organic and conventional vineyards and a natural site from one of the landscape (named Firgrove). The top levels are the insect species which visit plants species at the bottom level. The lines between the two levels represent the interactions between the two levels and the thickness of the line shows the frequency of interactions. The width of the upper and lower rectangles indicate the quantity of insects and plants involved in visitations respectively. Insect and plant codes (numbers) interpreted in Tables 3.2 and 3.3 respectively.

Organic



Conventional





Natural

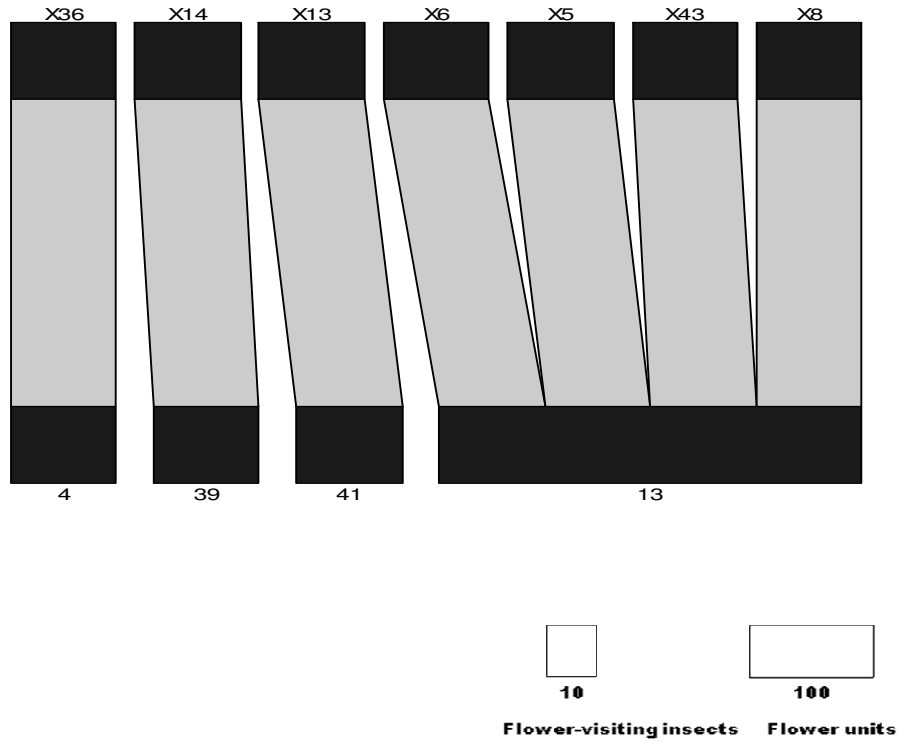
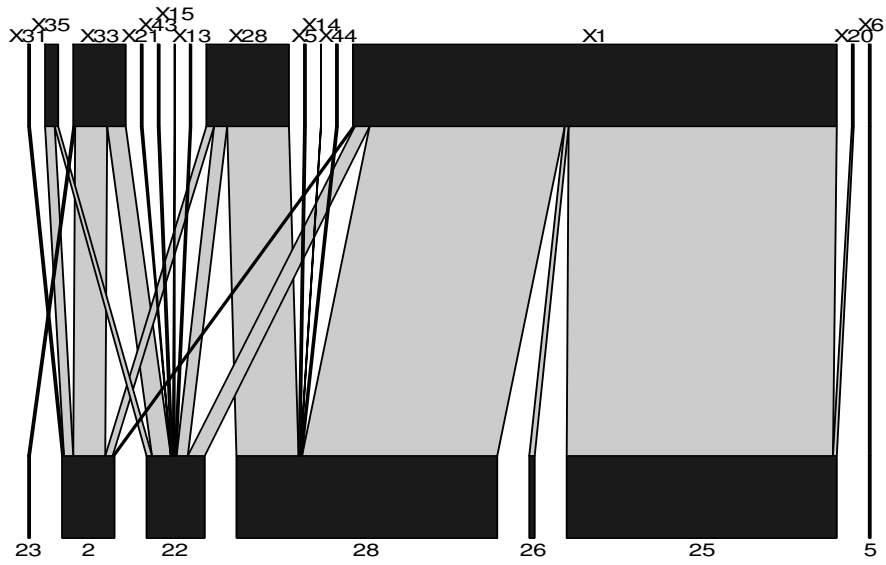
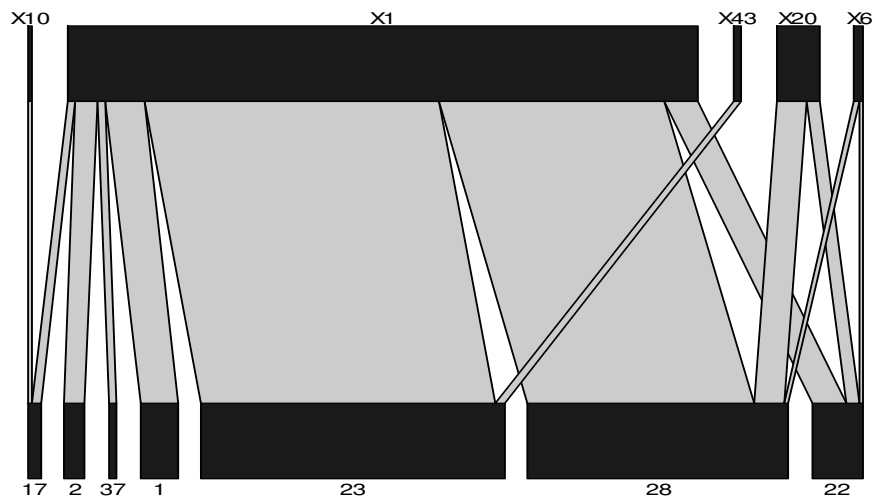


Figure 3.3 Insect-flower interaction networks of organic and conventional vineyards and a natural site from one of the landscape (named Joostenberg). The top levels are the insect species which visit plants species at the bottom level. The lines between the two levels represent the interactions between the two levels and the thickness of the line shows the frequency of interactions. The width of the upper and lower rectangles indicate the quantity of insects and plants involved in visitations respectively. Insect and plant codes (numbers) interpreted in Tables 3.2 and 3.3 respectively.

Organic



Coventional



Natural

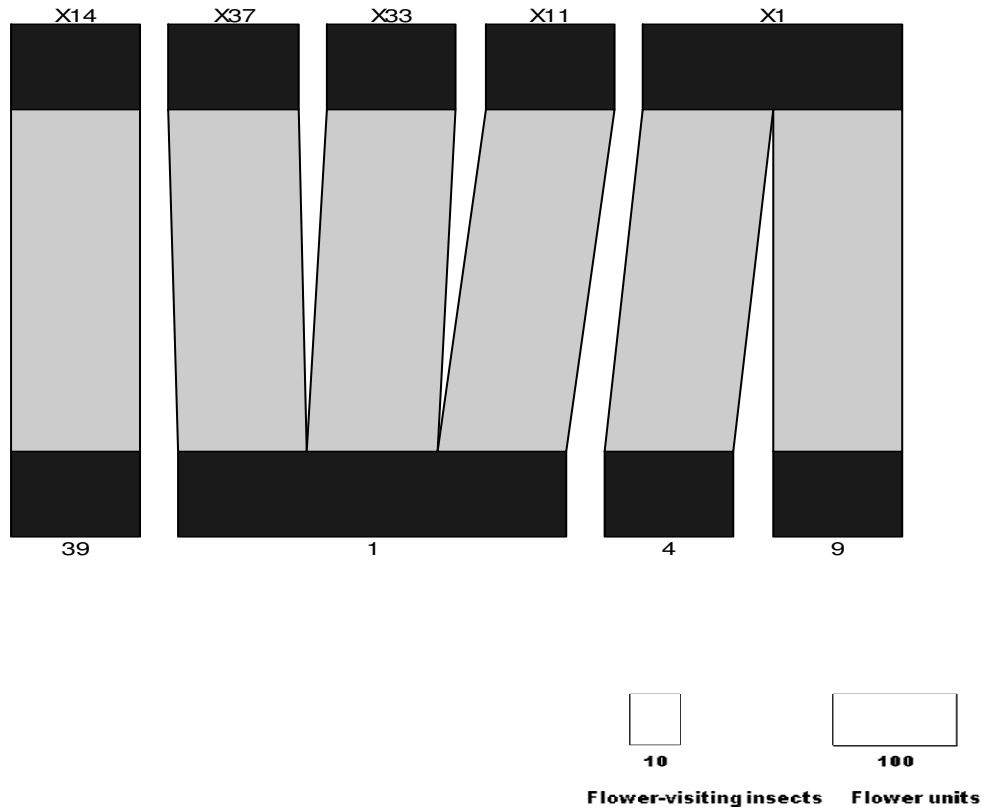
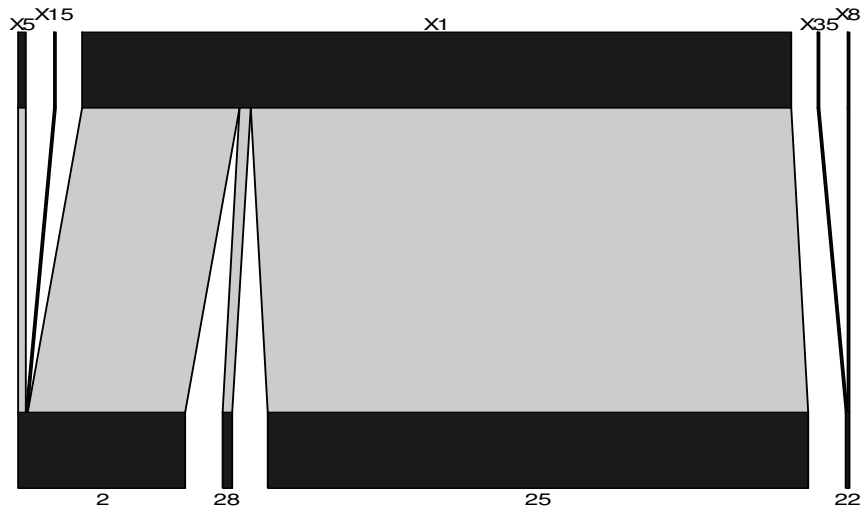
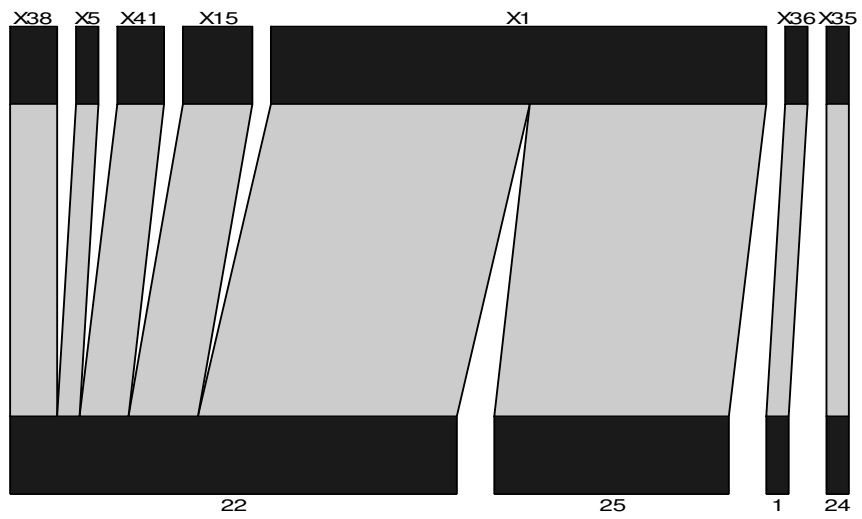


Figure 3.4 Insect-flower interaction networks of organic and conventional vineyards and a natural site from one of the landscape (named Spier). The top levels are the insect species which visit plants species at the bottom level. The lines between the two levels represent the interactions between the two levels and the thickness of the line shows the frequency of interactions. The width of the upper and lower rectangles indicate the quantity of insects and plants involved in visitations respectively. Insect and plant codes (numbers) interpreted in Tables 3.2 and 3.3 respectively.

Organic



Conventional



Natural

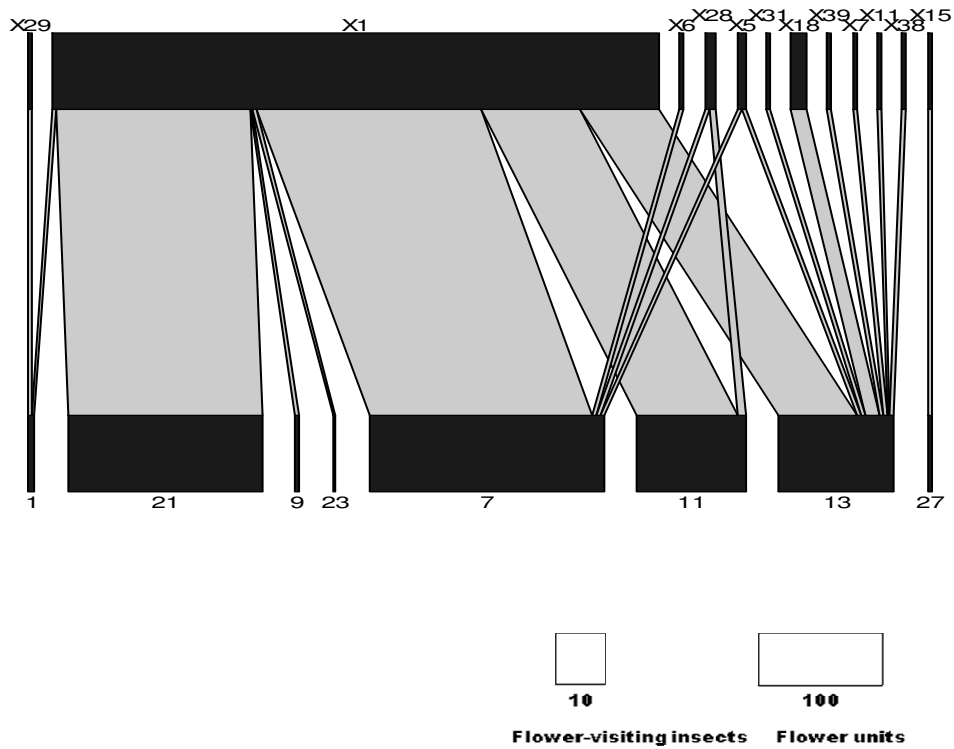


Figure 3.5 Insect-flower interaction networks of organic and conventional vineyards and a natural site from one of the landscape (named Uitzght). The top levels are the insect species which visit plants species at the bottom level. The lines between the two levels represent the interactions between the two levels and the thickness of the line shows the frequency of interactions. The width of the upper and lower rectangles indicate the quantity of insects and plants involved in visitations respectively. Insect and plant codes (numbers) interpreted in Tables 3.2 and 3.3 respectively.

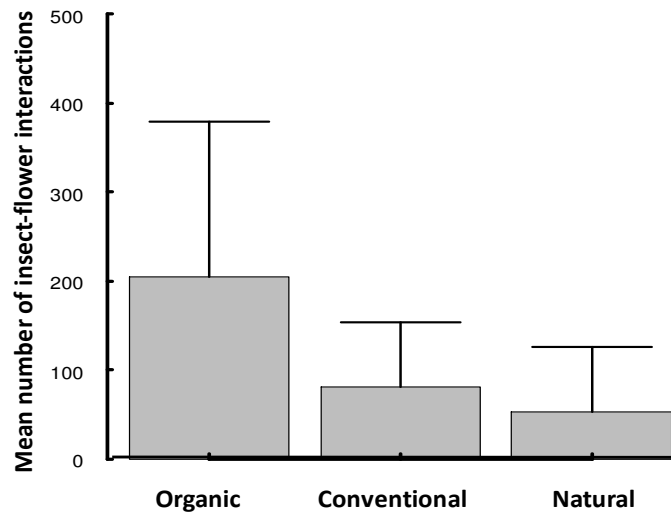


Figure 3.6 Mean ( $\pm$  SE) number of insect-flower interactions in organic and conventional vineyards and natural sites.

Table 3.2 Insect species and codes used in food webs plots

Food web codes	Order	Genus/Other	Species
X1	Hymenoptera	<i>Apis</i>	<i>mellifera</i>
X2	Hymenoptera	<i>Melitta</i>	<i>arrogans</i>
X3	Hymenoptera	<i>Amegilla</i>	<i>spilostoma</i>
X4	Hymenoptera	Halictidae	spp
X5	Hymenoptera	<i>Halictus</i>	spp1
X6	Hymenoptera	<i>Halictus</i>	spp2
X7	Hymenoptera	<i>Lasioglossum</i>	spp1
X8	Hymenoptera	<i>Lasioglossum</i>	spp2
X9	Hymenoptera	<i>Lasioglossum</i>	spp3
X10	Hymenoptera	<i>Lasioglossum</i>	spp4
X11	Hymenoptera	<i>Pseudoanthidium</i>	spp1
X12	Hymenoptera	<i>Megachile</i>	frontalis
X13	Hymenoptera	<i>Anthophora</i>	<i>wartmanni</i>
X14	Hymenoptera	<i>Xylocopa</i>	rufitarsis
X15	Hymenoptera	<i>Patellapis</i>	spp1
X16	Coleptera	Hopliini	spp1
X17	Coleptera	Hopliini	spp2
X18	Coleptera	Hopliini	spp3
X19	Coleptera	Hopliini	spp4
X20	Coleptera	Hopliini	spp5
X21	Coleptera	Hopliini	spp6
X22	Coleptera	Hopliini	spp7
X23	Coleptera	Hopliini	spp8
X24	Coleptera	Hopliini	spp9
X25	Coleptera	Hopliini	spp10
X26	Coleptera	Hopliini	spp11
X27	Coleptera	Hopliini	spp12
X28	Coleptera	Hopliini	spp13
X29	Coleptera	Hopliini	spp14
X30	Coleptera	Hopliini	spp15
X31	Coleptera	Hopliini	spp16
X32	Coleptera	Hopliini	spp17
X33	Coleptera	Hopliini	spp18
X34	Coleptera	Hopliini	spp19
X35	Diptera	<i>Syrphus</i>	spp1
X36	Diptera	<i>Agarcina</i>	spp1
X37	Diptera	Syrphidae	spp1
X38	Diptera	<i>Eristalis</i>	spp1
X39	Diptera	Syrphidae	spp2
X40	Diptera	Syrphidae	spp3
X41	Diptera	<i>Syritta</i>	spp1
X42	Diptera	<i>Syritta</i>	spp2
X43	Lepidoptera	<i>Cynthia</i>	cardui
X44	Lepidoptera	<i>Utetheisia</i>	pulchella
X45	Hymenoptera	Masaridae	spp

Table 3.3 Plants species and codes used in food web plots

Food web codes	Family	Genus	Species
1	Asteraceae	<i>Senecio</i>	<i>burchelli</i>
2	Asteraceae	<i>Senecio</i>	spp
3	Asteraceae	<i>Senecio</i>	<i>pterophorus</i>
4	Asteraceae	<i>Helichrysum</i>	<i>teretifolium</i>
5	Asteraceae	<i>Helichrysum</i>	<i>cymosum</i>
6	Asteraceae	<i>Helichrysum</i>	<i>grandiflorum</i>
7	Asteraceae	<i>Helichrysum</i>	spp
8	Asteraceae	<i>Helichrysum</i>	<i>indicum</i>
9	Asteraceae	<i>Helichrysum</i>	<i>patulum</i>
10	Asteraceae	<i>Arctotis</i>	<i>acaulus</i>
11	Asteraceae	<i>Berkheya</i>	<i>macrocephala</i>
12	Asteraceae	<i>Chrysanthemoides</i>	<i>monilifera</i>
13	Asteraceae	<i>Athanasia</i>	<i>trifurcata</i>
14	Asteraceae	<i>Felicia</i>	<i>filifolia</i>
15	Asteraceae	<i>Corymbium</i>	<i>glabrum</i>
16	Asteraceae	<i>Oncosiphon</i>	<i>suffruticosum</i>
17	Asteraceae	<i>Ursinia</i>	spp
18	Asteraceae	<i>Ursinia</i>	<i>paleacea</i>
19	Asteraceae	<i>Picris</i>	<i>echioides</i>
20	Fabaceae	<i>Aspalathus</i>	<i>globulus</i>
21	Fabaceae	<i>Aspalathus</i>	<i>quinquefolia</i>
22	Fabaceae	<i>Raphanus</i>	<i>raphanistrum</i>
23	Fabaceae	<i>Trifolium</i>	<i>angustifolium</i>
24	Fabaceae	<i>Medicago</i>	<i>polymorpha</i>
25	Fabaceae	<i>Vicia</i>	<i>benghalensis</i>
26	Fabaceae	<i>Indigofera</i>	spp
27	Fabaceae	<i>Aspalathus</i>	<i>cordata</i>
28	Boraginaceae	<i>Echium</i>	<i>plantagineum</i>
29	Iridaceae	<i>Micranthus</i>	<i>alopecuroides</i>
30	Iridaceae	<i>Aristea</i>	spp
31	Iridaceae	<i>Hesperantha</i>	<i>falcata</i>
32	Anacardiaceae	<i>Searsia</i>	<i>laevigata</i>
33	Anacardiaceae	<i>Rhus</i>	<i>angustifolia</i>
34	Crassulaceae	<i>Crassula</i>	<i>flava</i>
35	Gentianaceae	<i>Chironia</i>	<i>baccifera</i>
36	Polygonaceae	<i>Rumex</i>	<i>cordatus</i>
37	Plantaginaceae	<i>Plantago</i>	<i>lanceolata</i>
38	Myrsinaceae	<i>Anagallis</i>	<i>avensis</i>
39	Lamiaceae	<i>Salvia</i>	<i>africana</i>
40	Rutaceae	<i>Agathosma</i>	spp
41	Tecophilaeaceae	<i>Cyanella</i>	<i>hyacinthoides</i>



## Discussion

There was no difference in species richness of insects and flowering plants in organic farming compared to conventional farming systems. Some studies have reported benefits from organic farming in vineyards for biodiversity (Isaia *et al.* 2006; Gaigher & Samways 2010). However, others have found similar species richness of insects and plants in organic and conventional vineyards (Brittain *et al.* 2010b; Bruggisser *et al.* 2010). It has been argued that reduced level of background disturbance associated with perennial crops such as vineyards imply that they require intermediate levels of disturbance provided by more intensive management systems, to provide conditions that allow for co-existence between species supported by low disturbance and stress-tolerant species (supported by high disturbance), to provide greater support for biodiversity (Townsend & Scarsbrook 1997; Mackey & Curie 2001; Bruggisser *et al.* 2010).

Proponents of the use of interaction networks for assessing effects of habitat modification on biodiversity have suggested that traditional measures such as species richness, abundance etc. may give inconclusive assessments of variations that might be inherent in floral and faunal assemblages as a result of management activities (Tylianakis *et al.* 2007). In spite of this identifiable gap, few studies to date have examined food web structure between organic and conventional farming (Macfadyen *et al.* 2009; Power & Stout 2011). Variation in the pattern of interactions of ecological communities across a gradient of habitat modification has been shown to drive functional activities of species and ecosystem services (McCann 2007; Tylianakis *et al.* 2007). These results confirm these variations in terms of significant differences in some web structure indices of insect-flower visitation networks between organic and conventional vineyards and natural sites. There was higher number of insect-flower interactions in organic vineyards than conventional vineyards, which could imply higher frequency of interaction between insects and flowering plants. Frequency of interaction in mutualistic networks has been shown to be one of the key determinants of plant reproductive success and to be important in the prediction of pollination service by a

pollinator species to a plant population with which it interacts (Morris 2003; Vázquez *et al.* 2005; Carvalheiro *et al.* 2010). Plant mediated community process which has been reported in agricultural landscapes (Holzschuh *et al.* 2007; Carvalheiro *et al.* 2011) was supported by this study. This is shown by the significant contribution of flower abundance to the number of insect-flower interactions' model. Flower abundance may however be acting in synergy with other factors as it is shown not to be significantly different between organic and conventional vineyards. Other forms of disturbance such as pesticides use can be limiting for insect communities in intensive management systems (Dormann *et al.* 2007; Brittain *et al.* 2010a). Perennial agricultural fields with rich floral rewards, such as the vineyards in this study, could be of important conservation value in supporting pollinator diversity on a local and landscape scale. This may eventually benefit the natural vegetation especially in a region such as the CFR where the plants are well known for specialized pollination systems (Colville 2002; Goldblatt *et al.* 1998; Johnson & Steiner 2003)

Although these networks here were similar in terms of compartmentalization and connectance values, networks of natural sites were surprisingly less nested and had lower number of insect-flower interaction compared to organic and conventional vineyards. For mutualistic networks, such as the ones studied here, nestedness has been shown to enhance stability of networks against random extinction and habitat loss (Memmott *et al.* 2004; Fortuna & Bascompte 2006; Burgos *et al.* 2007). A large proportion of the indirect effect of connectance and diversity on network persistence has been shown to be mediated through nestedness and modularity (compartmentalization) (Thébault & Fontaine 2010). Quality as well as quantity of the natural and semi-natural vegetation patches that are left for the purpose of conservation in agricultural landscape is also important, with quality shown to be a better predictor (Potts *et al.* 2003; 2005). When the quality of these patches is low, they may have little value for the conservation of insect-flower interactions (Kleijn & Van Langevelde 2006). In the CFR, the dominant fynbos vegetation has a fire dependent community process (Mucina & Rutherford 2006). It is important for fynbos vegetation to

burn every 20 – 30 years to prevent loss of species and maintenance of its floral and faunal integrity (SANparks 2011). Site specific monitoring of the state of fynbos patches to determine the value patches can offer as source habitat and controlled burning to resuscitate flora and fauna diversity in these remnant fynbos patches within agricultural landscapes are recommended. Fire has been found to benefit insect and plants communities (Huntzinger 2003; Hartley *et al* 2007) especially on a long term when well managed (Black *et al.* 2007). This may involve rotational burning that leaves unburned, potential micro-refuges or insurance patches to make provision for re-colonization of burned patches. This will potentially incorporate both short term and long term benefits for flower-visiting insect diversity and their interaction network in CFR agricultural landscape.

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## **Chapter 4 - Assessment of biotic homogenization of insect-flower interaction networks in organic and conventional vineyards vs. natural vegetation**

### **Abstract**

The world's biota, and important species interaction networks such as insect-flower networks, are facing threats of biotic homogenization. It is unclear whether environmentally friendly farming approaches, such as organic farming can reduce the effects of these threats caused by intensive management in agricultural landscapes. This study examined biotic homogenization of insect-flower interactions. These interactions are of significant conservation value in the fragmented Cape Floristic Region (CFR) agricultural landscape. Assessment was done using permutational distance-based test for homogeneity of multivariate dispersion to determine whether organic and conventional vineyards contributed to loss of diversity across the landscape through homogenization of their insect-flower interaction networks. Neither organic nor conventional vineyards homogenized interaction networks compared to natural sites. Even when important pollinators of indigenous plants were separated out for analysis, networks were still not homogenized for both types of vineyard management. The vineyard ecosystem holds promise for these evolutionarily important networks which are important determinants of plant speciation in the CFR.

### **Introduction**

Various human enhanced activities lead to break down of natural biogeographic barriers (Olden & Poff 2003) and faunal mixing, resulting in the concept of biotic homogenization. This phenomenon is associated with the gradually increasing spatial and temporal semblance of the Earth's biota (Lockwood & McKinney 2001; Olden & Poff 2003). Patterns of biotic

homogenization were reported to be driven by two key processes; extinction of native species and invasion of species in a community (McKinney & Lockwood 1999; Lockwood & McKinney 2001, Olden & Poff 2003). However, recent findings demonstrate re-organisation of native species as an active part of this phenomenon, potentially suggesting that biotic homogenization may be a more widespread phenomenon than previously thought (Keith *et al.* 2009). There are multiple consequences associated with biotic homogenization ranging from loss of functional diversity to diminished geographical isolation, these may affect important evolutionary process such as allopatric speciation. Future biodiversity might also be threatened through the fusion of evolutionary lineages through hybridization and introgression (Perry *et al.* 2002; Olden *et al.* 2004).

The increasing importance of examining biotic homogenization across multiple trophic levels is currently being stressed (Olden *et al.* 2004). This will incorporate knowledge on higher levels of ecological effects such as the relationship between biotic mixing and food web structure and dynamics across flora and fauna communities that occupy landscapes and regions (Olden *et al.* 2004; Laliberté & Tylianakis 2010).

Limited number of studies has verified the effect of organic farming; a globally accepted wildlife friendly farming system (Soil Association 2000; Hole *et al.* 2005) for not just local diversity but also diversity across sites and landscapes (Clough *et al.* 2007). However, Clough *et al.* (2007) reported that measures of diversity beyond local diversity contribute substantially to species richness and should be included in evaluations of the effectiveness of agri-environment schemes such as organic farming. Doing this at a multitrophic scale tallies with the aims of Olden *et al.* (2004) and Laliberté & Tylianakis (2010) to examine biotic homogenization dynamics at multiple trophic levels. To date few studies have compared food web structures and indices in organic farming systems with those of more intensive systems of farming (Macfadyen *et al.* 2009; Power & Stout 2010). Some of these studies have

reported differences in the web structure and indices of parasitoid-host and insect-flower interactions between organic and conventional farming systems, with organics having larger and more robust web structures. However, most of the studies were limited to local diversity patterns of these webs and were done in annual crops systems. Laliberté & Tylianakis (2010) reported homogenization of interaction frequency and composition of parasitoid-host networks for annual rice and pasture fields but not for perennial agro forests, when the two land use types were compared with natural forest sites. Examination of possible homogenization caused by vineyard cultivation is important, especially in the Cape Floristic Region (CFR) with rich flora and fauna and high levels of endemism (Cowling 1990; Donaldson 2002; Kuhlmann 2009) and its high priority for conservation (Conservation International 2010). Biodiversity in this region is under increasing threat from agriculture particularly vine cultivation (Biodiversity and Wine Initiative 2010) and various activities ranging from protection of natural areas around vineyard, to integrated production of wine (IPW) and lately organic farming are being introduced to combat biodiversity loss. In this study the following hypothesis were examined i) interaction frequencies and composition (presence-absence) of insect-flower networks in organic and conventional vineyards are homogenous compared to natural sites. ii) abundance and composition (presence-absence) of flower-visiting insects in organic and conventional vineyards are homogenous compared to natural sites. iii) interaction frequencies and composition (presence-absence) of important insect pollinators of indigenous plants in organic and conventional vineyards are homogenous compared to natural sites.

## **Methods**

### ***Study sites***

Five pairs of organic and conventional vineyards and five natural sites were chosen for this study such that there was one natural site in the same landscape with the vineyard pairs as reference site. Detailed description of the study sites and study area are in chapter 3.

### ***Insect-flower interaction sampling***

Sampling and data collection was between early October and mid December, 2010 as described in details in chapter 3.

### ***Data analyses***

Testing for homogeneity or differences in multivariate dispersion of communities usually involves using distance or dissimilarity measures that emphasize either changes in presence-absence (composition) data or changes in relative abundance data (Anderson *et al.* 2006; Laliberté & Tylianakis 2010). I checked for homogenization of interaction frequency and flower-visiting insect abundance data, as well as their presence-absence data among organic and conventional vineyards and the natural sites. Hellinger distance was chosen for the interaction frequency and abundance data. This distance measure can represent species and sites in biplots, reach an asymptote for sites that have no species in common, gives low weight to rare species and interactions and has been reported to give better representation than the more popular Bray-Curtis distance. (Legendre & Legendre 1998; Legendre & Gallagher 2001; Laliberté & Tylianakis 2010). For the presence-absence (binary) data of both interactions and flower-visiting insect species, I used the Jaccard dissimilarity measures, which is a well used dissimilarity measure in ecological studies involving biotic homogenization (Olden & Rooney 2006; Laliberté & Tylianakis 2010). Hellinger and Jaccard measures were computed with the “vegdist” and “decostand” functions in R vegan package

(R development core team 2010). The dissimilarity matrices were used for the computation of the distance of each site to its group centroid in multivariate space. This was done with the `betadisper` function for multivariate dispersion analysis in R `vegan` package. This R program was written based on PERMDISP2 procedure for the analysis of multivariate homogeneity of group dispersions (variances) (Anderson 2006). This test is a multivariate analogue of Levene (1960) test for homogeneity of variance. It is reported to be a robust method for analyzing relative abundance, as well as presence-absence data (Anderson 2006; Laliberté & Tylianakis 2010). The distances were then analyzed with one-way permutational ANOVA to check for multivariate dispersion between groups of sites i.e. organic and conventional vineyards and natural sites. According to Johnson & Steiner (2003), bees, especially oil-collecting bees and hopliine beetles, are among the important pollinators of South African plants, so the data of the abundance of these taxa and their interaction frequencies as well as their presence-absence data were extracted for separate analysis. I confirmed that sampling design did not bias the results obtained by checking for homogeneity of geographical dispersion among groups of sites using latitude and longitude coordinates of each site (Zuur *et al.* 2007; Laliberté & Tylianakis 2010). Post hoc comparison using the Tukey Honestly Significant difference test for equal sample sizes was made.

## **Results**

There was no homogenization of insect-flower networks in terms of interaction frequency in organic and conventional vineyards compared to the natural sites across the landscapes.

Average distance of organic and conventional vineyard networks to the group centroids was not significantly different from average distance of natural sites to the group centroid for insect-flower interaction frequency ( $P > 0.05$ , Figure 4.1). Analysis of flower-visiting insect abundance showed no significant difference in average distance to group centroids when

organic and conventional vineyards were compared with natural sites ( $P > 0.05$ , Figure 4.2). Similar patterns emerged from the analysis on the presence-absence data matrix of insect-flower interactions and presence-absence of flower-visiting insects, with no significant difference in average distance to group centroids of organic and conventional vineyards and natural sites ( $P > 0.05$ ). Results of multivariate dispersion analysis, when important pollinators of indigenous plants were separated out, showed that both organic and conventional vineyards did not homogenize interaction networks across the landscape. This is shown by similarity in the dispersion of networks of the vineyards and the natural sites in multivariate space (Figure 4.3). Average distance of organic and conventional vineyard networks to the group centroids was not significantly different from average distance of natural sites to the group centroid, for insect-flower interaction frequency of the important pollinators of indigenous plants ( $P > 0.05$ , Figure 4.5). Organic and conventional vineyards also showed no homogeneity in terms of abundance of these taxa when compared to the natural sites in their average distance to the group centroids ( $P > 0.05$ , Figure 4.6). These results were also confirmed by the non-significant difference of the average distance to group centroids at organic and conventional vineyards and natural sites for the analysis on their presence-absence data ( $P > 0.05$ ). The results were not biased by the sampling design, as spread of sites in geographical space, i.e. their spatial dispersion in the study region, did not differ significantly between organic and conventional vineyards and the natural sites ( $P = 0.989$ ).

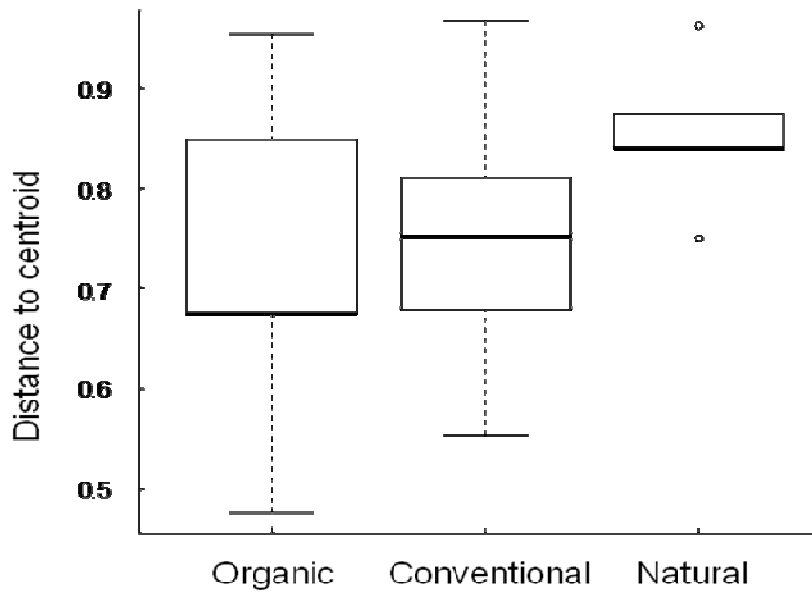


Figure 4.1 Boxplot showing distance of individual organic and conventional vineyards and natural sites to their group centroids based on insect-flower interaction frequency.



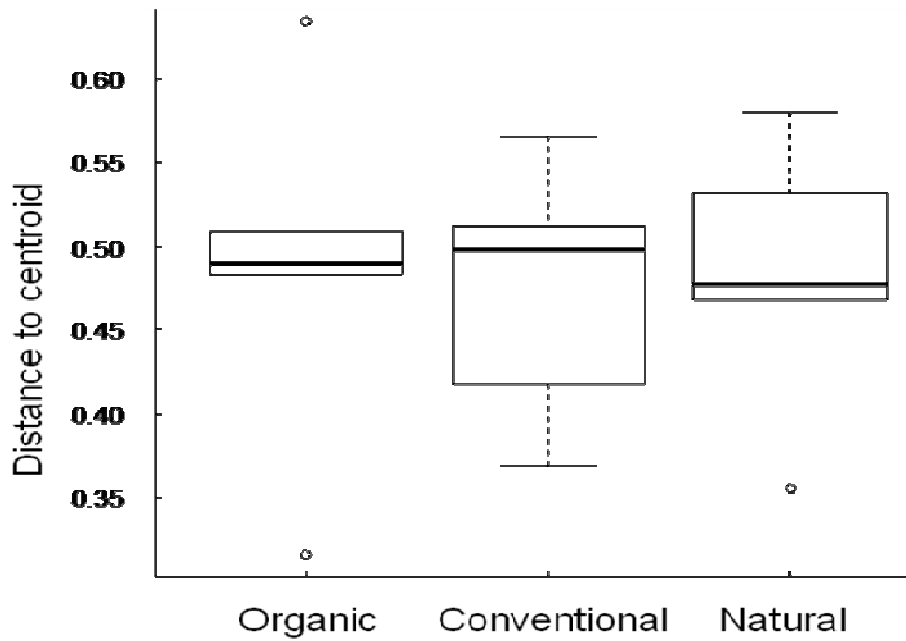


Figure 4.2 Boxplot showing distance of individual organic and conventional vineyards and natural sites to their group centroids based on abundance of flower-visiting insects.

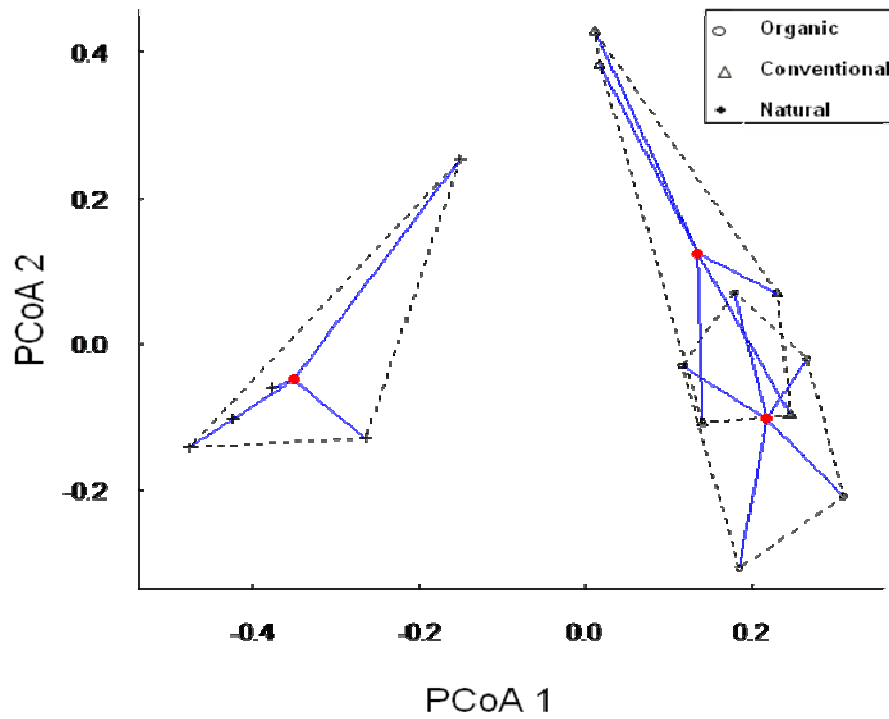


Figure 4.3 Principal coordinate analysis (PCoA) plot of insect-flower interaction frequency of important pollinator of indigenous plants based on Hellinger distance, showing proximity of sites to the group centroid and to other sites of the same management type but from different landscapes for organic and conventional vineyards and natural sites.

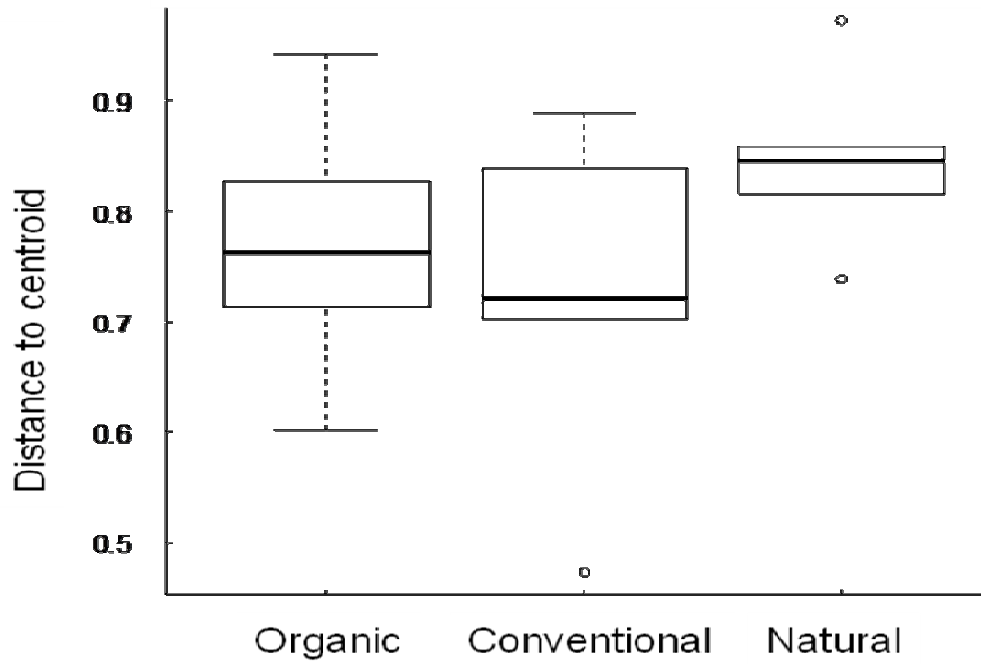


Figure 4.4 Boxplot showing distance of individual organic and conventional vineyards and natural sites to their group centroids interaction frequency of important pollinators of indigenous plants.

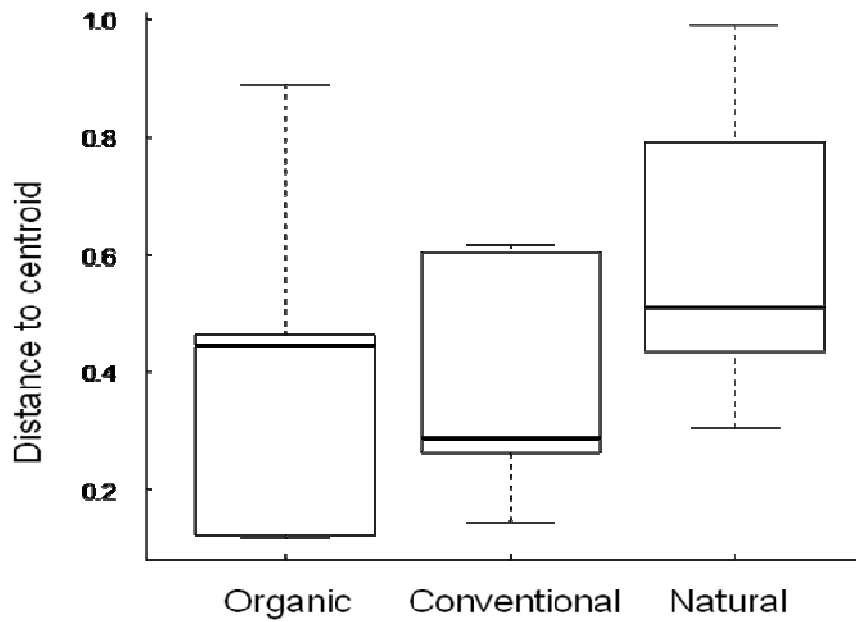


Figure 4.5 Boxplot showing distance of individual organic and conventional vineyards and natural sites to their group centroids based on abundance of important pollinators of indigenous plants.

## Discussion

Assessments of how habitat modification contributes to the shaping of biodiversity patterns, which are limited to local or habitat scale have been shown to be incomplete (Veech *et al.* 2002; Legendre *et al.* 2005; Clough *et al.* 2007). This is because total diversity is often a combination of  $\alpha$ -diversity, average within plot diversity,  $\gamma$ -diversity, total diversity across plots and  $\beta$ -diversity, the difference between ( $\gamma$ ) total and ( $\alpha$ ) local diversity, which measures variation in species composition between plots (Clough *et al.* 2007). This study shows that organic and conventional vineyards did not homogenize variation of insect-flower interaction frequency compared to natural sites. In a comparison of host-parasitoid interaction networks in different types of land use systems, Laliberté & Tylianakis (2010) reported homogenization of interaction frequency and composition for annual rice and pasture fields but not for perennial agro forests, when the two land use types were compared with natural forest sites. It is not surprising that both organic and conventional vineyards did not homogenize insect-flower interactions. Perennial systems such as the vineyards studied here have been shown to hold special value for conservation irrespective of whether they were under organic or conventional management (Bruggisser *et al.* 2010). The results also show that interaction and abundance of important pollinators of indigenous plants were also not homogenized by organic and conventional vineyards. In Europe, vineyards have been reported to be host to many rare and threatened species, and usually have high biodiversity (Costello & Daane 1998; Gliessman 2001; Isaia *et al.* 2006). Bruggisser *et al.* (2010) has called for the protection of vineyard habitats and for expanded knowledge on the effect of different management practices that will help maximize the conservation benefits vineyards may hold for biodiversity. This is especially important in CFR, a region of global ecological importance, for floral and fauna diversity, that has to be rescued from threats of biotic homogenization, while maintaining agricultural production.

Homogenization of interaction networks occur when communities of interacting species become homogenized in agricultural landscapes, as has been reported for bee communities in Europe, where homogenization was due to loss of semi-natural habitats and high levels of pesticides at landscape and regional levels (Dormann *et al.* 2007; Brittain *et al.* 2010). Both Organic and conventional vineyard practices being used by viticulturist in the CFR have the potential of sustaining reasonable semi-natural vegetation within vineyard rows and hedges. Prospects for achieving these and similar conservation goals will also be enhanced in this region by conventional growers who have adopted the integrated farming approach and organic growers both of whom employ measures that support minimum levels of pesticide applications unlike intensive conventional practices that spray indiscriminately. Greater awareness campaigns should be launched in the CFR conservation sector as has been achieved by the BWI, which has successfully protected about 100% of the vineyard footprint in the CFR as natural vegetation around vineyards. This will bring a holistic approach of both preservation of natural habitats, and environmentally friendly resource use in the agricultural areas, both aimed at combating biodiversity loss and biotic homogenization. Catering for the regional insurance of flower-visiting insect biota as well as their interactions, which has been difficult to maintain under spatially and temporally changing environmental conditions (Loreau *et al.* 2003) can become a reality with well managed vineyard ecosystems in the CFR.

Examining if results obtained here are consistent across temporal scales as has been suggested for other types of land use (Tylianakis *et al.* 2005) will help in the predictions of future patterns, and give more insight for adequate conservation planning.

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## **Chapter 5 - Vineyard management effects on species turnover of bees and flowering plants**

### **Abstract**

Partitioning diversity into its various components helps cater for spatial and temporal turnovers of species. This can provide more detailed understanding when the performance of agri-environment schemes is being assessed. Additive partitioning was used to obtain local (alpha) and spatial and temporal (beta) components of total diversity of bees and alpha and spatial beta diversity of flowering plants. These diversity partitions were compared between organic and conventional vineyards, as well as with reference natural vegetation sites. Local, spatial and temporal components of diversity contributed meaningfully to total diversity. All components of bee and flowering plant diversity were similar between organic and conventional vineyards. Local and temporal components of bee diversity, as well as local and spatial components of flowering plant diversity, were significantly higher in the natural vegetation compared to organic and conventional vineyards. The integrated approach used by conventional viticulturists in this region creates sufficient heterogeneity within vineyards, and holds promise for biodiversity if improved upon. However, temporal scale analysis shows that both organic and conventional viticulturists need to put in place practices that are sustainable for the future of biodiversity in their vineyards.

### **Introduction**

Additive partitioning of diversity reveals the different components that make up the diversity of flora and fauna communities. These components may be spatial or temporal partitions of overall diversity (Lande 1996; Tylianakis *et al.* 2005; Clough *et al.* 2007). McKinney (2005) reported that caution should be exercised to avoid narrow perception of the earth's biota at local scale when considering issues of biodiversity dynamics. Valuable consideration of how

biodiversity is partitioned spatially is of increasing importance as the world faces biotic reorganization referred to in literature as “New Pangea” (Rosenzweig 2001a, 2001b; Olden & Poff 2003; McKinney 2005). Much of the growing literature on the spatial and temporal patterns and consequences of taxonomic homogenization has focused on a relatively restricted set of taxa, with little known for other groups such as insects in this respect (Shaw *et al.* 2010).

Knowledge of spatial pattern of diversity has been used in the understanding of macro ecological patterns (Blackburn & Gaston 1996; Novotny & Weiblen 2005) and identified as valuable in selecting spatial scale for species conservation (Gering *et al.* 2003). This would be of particular importance in agricultural landscapes, where conservation goals are being extended to both natural and managed habitats that are well connected in landscape mosaics (Thompson 1994; Ings *et al.* 2009; Tylianakis *et al.* 2010; Ekroos *et al.* 2010). The occurrence and spatial arrangement of semi-natural habitats have been shown to play a key role in determining patterns of diversity in agricultural landscapes (Holzschuh *et al.* 2009).

Agri-environment schemes that encourage wildlife friendly agriculture are being proposed as potential solution to the pressing need of conserving biodiversity at different scales while sustaining agricultural yield (Clough *et al.* 2011). Organic farming is one of such important approaches, and has received attention on its potential impact in addressing the challenges of halting the scale of biodiversity loss for various taxa (Schmidt *et al.* 2005; Roschewitz *et al.* 2005; Holzschuh *et al.* 2007). Only recently have studies examining the effect of organic farming included spatial partitions (component) of diversity, that goes beyond the habitat or plot sampled (Clough *et al.* 2007). Partitioning of diversity into alpha and beta components has been used to show that diversity decreases in intensively managed fields (Tylianakis *et al.* 2005) and agricultural landscapes (Ekroos *et al.* 2010). Management factors can have a heterogenous effect on species richness. Futhermore, beta diversity was

shown to contribute significantly to total richness at varying proportions for different taxa (Clough *et al.* 2007) when organic wheat fields were compared with conventional ones.

I examined here the effect of land use (organic and conventional vineyards and natural sites) on alpha, temporal beta and spatial beta diversity of bees and on alpha and beta diversity of flowering plants. It has been proposed that pollinators, their population dynamics and the systems within which they interact should be a priority group for case studies in the conservation and sustainable use of agricultural biodiversity. Bees in particular are well known to be potential keystone indicator species because they can be easily monitored (Eardley 2001).

## **Methods**

### ***Study site***

Detailed description of study sites is found in Chapter 2

### ***Insect and flowering plant sampling***

Bee sampling took place during spring season between August and December of years 2009 and 2010. Sampling in year 2009 was done with twelve yellow pan traps each of 1000 ml capacity and two window traps each of dimension 0.5 x 0.5 m on each site. Traps were half filled with water, and a few drops of detergent were added to break the surface tension of the water and enhance insect trapping. Traps were placed in pairs 20 m away from the edge of each field and traps in a pair 2 m apart to increase overall effectiveness of trapping. Further details of trapping are as shown in Chapter 2. Sampling in year 2010 was done by collecting bees visiting flowers along transects in the same study sites as 2009. 100 X 50 m plots were demarcated on each study site and three transects were placed randomly within the plots along which bees were sampled during each of eight sampling visits to each site. Bees were sorted and identified to species or morphospecies levels from samples of both years.

Detailed vegetation sampling, where plant samples were collected and identified to species level was done in 2010. Plant sampling was done along the same transects where insects were sampled. Species richness of flowering plants were recorded in six 1 m<sup>2</sup> quadrats at 10 m interval along each 50 m transect. Details of vegetation sampling is as shown in Chapter 3

### ***Data analyses***

Species diversity was partitioned to determine spatial and temporal turnover of bee species and spatial turnover of flowering plant species using the additive partitioning of species diversity approach;  $\gamma = \alpha + \beta$  (Lande 1996; Wagner *et al.* 2000; Veech *et al.* 2002). This technique is insensitive to differences in sampling effort among replicates, and therefore does not require rarefaction of data prior to analyses (Veech *et al.* 2002; Crist *et al.* 2003).

According to Tylianakis *et al.* (2005), we defined the different partitions of diversity for bees and flowering plants as follows. Alpha diversity was determined for bees as the mean species richness per site per year, and for flowering plants as the mean species richness per site.

Spatial beta diversity for bees and flowering plants was the total number of species sampled in a land use type over the entire sampling period minus the mean number of species per plot of that land use type over the entire sampling period. Temporal beta diversity between the years for bees was defined as the total number of bee species sampled in a plot over the two years minus the mean number of bee species per plot per year (alpha).

Effect of land use on alpha and beta diversities of each taxon was analyzed in a generalized linear mixed effects model with land use as fixed factor and locality as random factor, to account for the locality based sampling design. All data analyses were done in R (version 2.12, R development core team 2010, Lme4 package). Model validations were done with plots of residuals and independent variables as described in previous chapters.

## Results

All partitions of diversity contributed substantially to total diversity on organic and conventional vineyards as well as the natural sites. The highest proportion of total diversity on all the land use types was from spatial beta diversity which ranged from 19-65% for bees (Figure 5.1) and 33-72% for flowering plants (Figure 5.2). Alpha diversity had a contribution to total diversity ranging between 18-53% for bees (Figure 5.1) and 28-61% for flowering plants (Figure 5.2). Temporal species turnover for bees had a contribution 7-36% to total diversity (Figure 5.1).

Models showed that all partitions of diversity measured for bees i.e. alpha diversity (Figure 5.3), temporal beta diversity (Figure 5.4) and spatial beta diversity (Figure 5.5) were similar between organic and conventional vineyards ( $P > 0.05$ , Table 5.1). A similar pattern was shown for alpha (Figure 5.6) and beta (Figure 5.7) diversities of flowering plants, which were not significantly different between organic and conventional vineyards ( $P > 0.05$ ).

Alpha, spatial and temporal beta diversity of bees were also shown to be similar between organic and conventional vineyards as well as natural sites ( $P > 0.05$ , Figure 5.3 & 5.4, Table 5.1). However, the following partitions of diversity were higher in natural sites than in organic and conventional vineyards. These involved spatial beta diversity of bees (Table 5.1, Figure 5.5), alpha diversity of flowering plants (Table 5.1, Figure 5.6) and beta diversity of flowering plants (Table 5.1, Figure 5.7).

Table 5.1 Generalized Linear Mixed Effect output to test land use and landscape effects on diversity partitions of bees and flowering plants

Response variable	Land use ( <i>P</i> value)	Proportion of Natural habitat ( <i>P</i> value)	Land use x Proportion of Natural Habitat ( <i>P</i> value)	d.f.	AICc
<i>α</i> -diversity (bee)					
Model 1	ns	ns	ns	-	18.19
Model 2	ns	ns	-	-	14.27
Model 3	ns	-	-	-	12.28
<i>β</i> -diversity spatial (bee)					
Model 1	ns	ns	ns	-	26.35
Model 2	0.03	ns	-	8	22.35
Model 3(Best model)	0.022	-	-	8	20.46
<i>β</i> -diversity temporal (bee)					
Model 1	ns	ns	ns	-	20.64
Model 2	ns	ns	-	-	16.96
Model 3	ns	-	-	-	15.06
<i>α</i> -diversity (flower)					
Model 1	0.002*	ns	ns	8	19.01
Model 2 (Best model)	0.000*	ns	-	8	16.70
Model 3	0.000*	-	-	8	17.83
<i>β</i> -diversity (flower)					
Model 1	0.000*	ns	ns	8	17.27
Model 2 (Best model)	$2.27 \times 10^{-11}$ *	ns	-	8	15.43
Model 3	$6.76 \times 10^{-11}$ *	-	-	8	15.61

\*significant at the 5% level, NS not significant



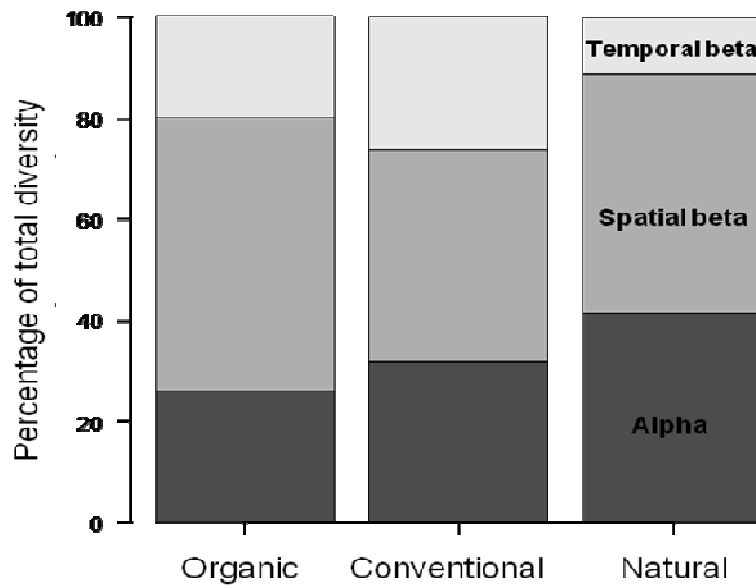


Figure 5.1 Percentage contribution of alpha, spatial beta and temporal beta diversities to species richness of bees sampled in organic and conventional vineyards and natural sites.

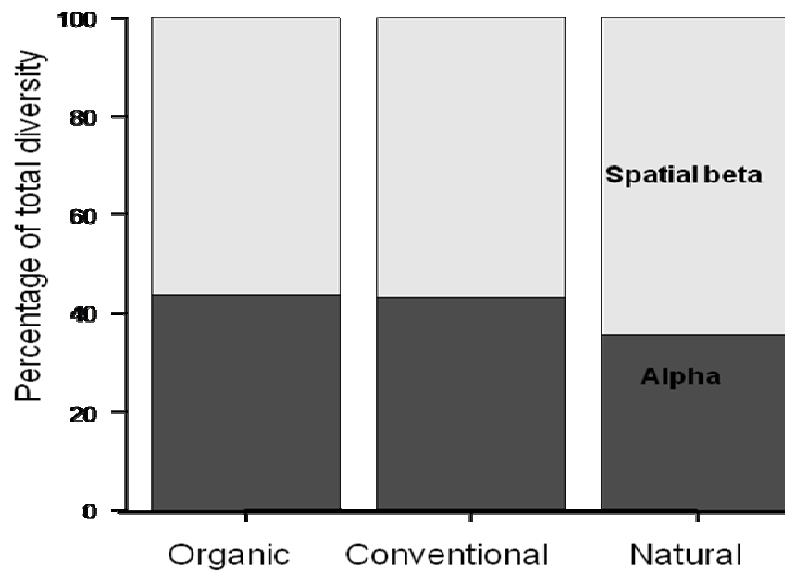


Figure 5.2 Percentage contribution of alpha and spatial beta diversities to species richness of flowering plants sampled in organic and conventional vineyards and natural sites.

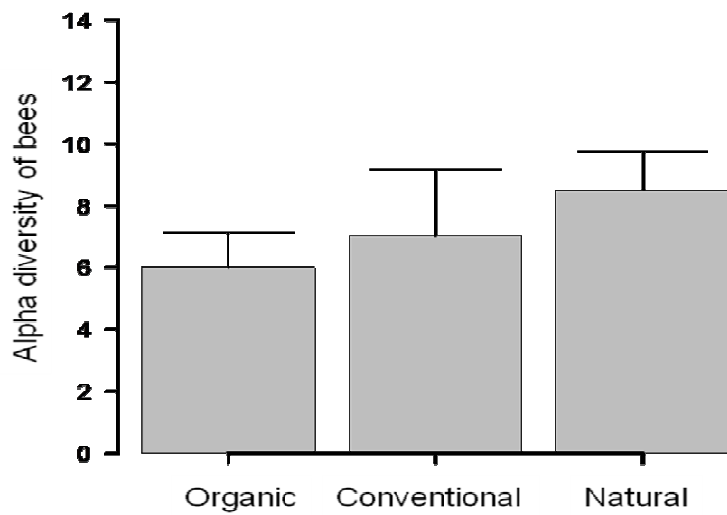


Figure 5.3 Barplots showing means and SE of alpha diversity of bees sampled in organic and conventional vineyards and natural sites.

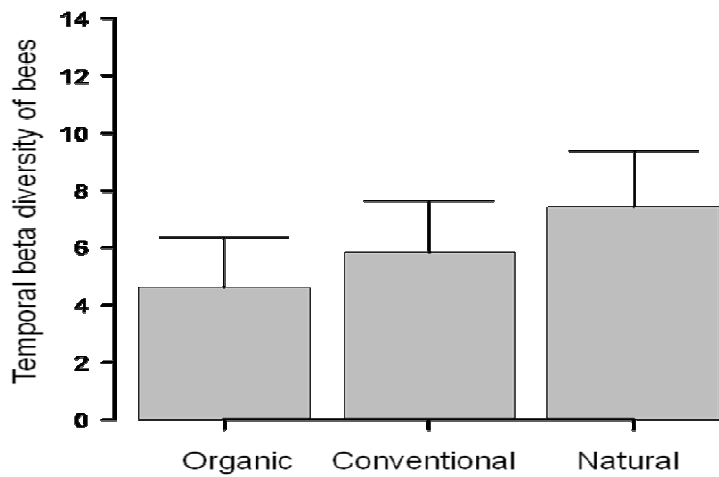


Figure 5.4 Barplots showing means and SE of temporal beta diversity of bees sampled in organic and conventional vineyards and natural sites.

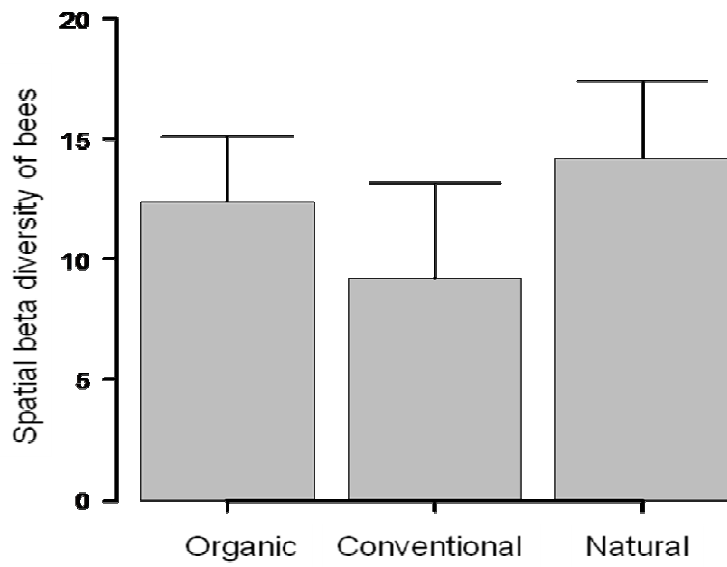


Figure 5.5 Barplots showing means and SE of spatial beta diversity of bees sampled in organic and conventional vineyards and natural sites.

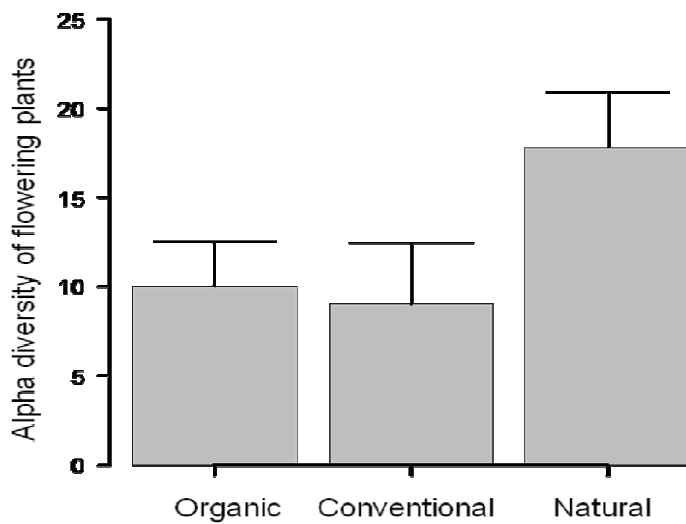


Figure 5.6 Barplots showing means and SE of alpha diversity of flowering plants sampled in organic and conventional vineyards and natural sites.

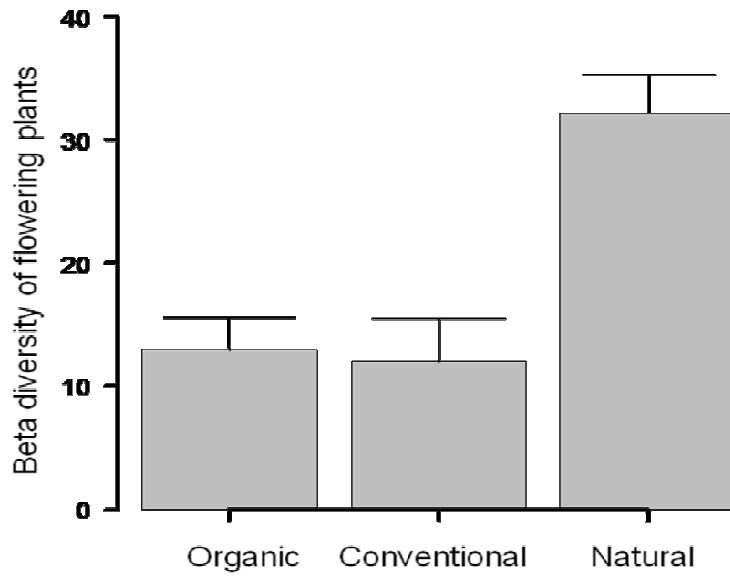


Figure 5.7 Barplots showing means and SE of beta diversity of flowering plants sampled in organic and conventional vineyards and natural sites.

## Discussion

The results showed that beta diversity contributed meaningfully to total diversity for both bees and flowering plants. High proportion of beta diversity for both groups of organisms was found irrespective of the type of land use. This is consistent with previous studies showing significant contribution of beta diversity to total diversity (Wagner *et al.* 2000; Roschewitz *et al.* 2005, Clough *et al.* 2007). Additive partitioning informs our understanding of local-regional patterns of species diversity, as well as spatial and temporal dependence of diversity components (Loreau 2000; Gering & Crist 2002; Gering *et al.* 2003; Tylianakis *et al.* 2005; Klimek *et al.* 2008). Diversity partition showed consistently high proportion of beta diversity irrespective of differences in dispersal ability between highly mobile bees compared to plants. It has been reported that for both higher disperser and lower disperser organisms, habitat heterogeneity and habitat variability are crucial factors in the determination of spatial differentiation of species richness across landscapes (Wagner *et al.* 2000; Diekötter *et al.* 2008). This especially important in the CFR, known for its high spatial turnover due to the exceptionally high meta-community scale diversity with some range restricted species, many occurring in small and isolated populations (Cowling & Lombard 2002). According to Clough *et al.* (2007), between-landscapes heterogeneity provides explanation for high beta diversity while within-landscape heterogeneity explains high alpha diversity. Perennials, which provide greater heterogeneity within study fields than do annual crop systems, are a major source of species richness in agroecosystems (Dauber *et al.* 2003; Jeanneret *et al.* 2003; Clough *et al.* 2005; Gabriel *et al.* 2005; Clough *et al.* 2007). This may explain the equally high alpha diversity as shown in this study. Appreciable interannual species turnover as contribution to total diversity of bees in organic and conventional vineyards and natural sites was also recorded. It has been speculated that interannual temporal turnover may also explain a significant portion of species richness (Tylianakis *et al.* 2005). This was confirmed in this study as temporal turnover of bees was significantly higher in the natural sites than the

vineyards. However, I propose that studies should be replicated across years more than two years used here. This may provide more detailed information about temporal aspects of species richness as the world faces temporal scale global changes from increasing human population and climate change (Tilman *et al.* 2001)

Local (alpha) diversity, temporal (beta) diversity and beta (spatial) diversity of bees and flowering plants were similar in organic and conventional vineyards. This confirms but extends results from previous studies that have reported non-significant differences in local diversity (Brittain *et al.* 2010; Bruggisser *et al.* 2010, Carvalheiro *et al.* 2010) but not species turnover as is reported here for plants and insects in organic compared to conventional vineyards. It has been suggested that reduced level of background disturbance associated with perennial crops such as vineyards imply that they require intermediate levels of disturbance, provided by more intensive management systems, to provide conditions that allow for co-existence between species supported by low disturbance and stress-tolerant species (supported by high disturbance), to provide greater support for biodiversity. (Townsend & Scarsbrook 1997; Mackey & Curie 2001; Bruggisser *et al.* 2010).

It was shown from the results that spatial beta diversity of bees was similar between organic and conventional vineyards. Sustainable weed management has been shown to benefit diversity of flower visiting insects and enhance delivery of the ecosystem service they supply (Carvalheiro *et al.* 2011). In vineyards, carefully maintained vine row vegetation that is not detrimental to crop production as reported in other parts of the world (Sanguaneko & León 2011), has made weeds in the form of cover crops and resident plant species, which may be annual or perennial, integral to vineyard rows in the CFR. This creates favorable habitat for several organisms by acting as refugia, reservoirs or alternative sites that hold resource benefits for biodiversity (Marshall *et al.* 2003; Sanguaneko & León 2011).

The importance of natural and semi-natural elements as resource and nesting sites (Meek *et al.* 2002) is, however, revealed by the significantly higher spatial turnover (beta diversity) of

bees as well as higher alpha and spatial beta diversity of flowering plants in the natural sites. High diversity of natural sites in agricultural landscapes may provide seed rain that has the potential of benefiting useful weed diversity within agricultural landscapes (Gabriel *et al.* 2005).

While recognizing the potential benefit that low intensive organic farming has for biodiversity. The integrated approach to vineyard management, which is used by conventional viticulturist, holds promise of biodiversity benefits for the agricultural landscape in CFR. However, both organic and conventional viticulturist must be aware that current levels of biodiversity are not guaranteed for the future, as shown from the results of species turnover of these two important groups. The insurance provided by the natural habitat should be well guarded while improving on farmland biodiversity.

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## **Chapter 6 – General conclusion and management recommendations**

The key role of agriculture in driving current and future global change has been well reported. This compelling global scenario is of great concern for biodiversity in South Africa, especially in the Cape Floristic Region (CFR), well known for its high endemism, and high priority for conservation. It therefore became imperative to investigate the impact of land use on biodiversity, based on vineyard cultivation which is the predominant agricultural practice in this region.

The critical role played by insect pollinators for both wild and crop plants in this region makes them a keystone functional group. Hence the need to fill knowledge gaps on the effect of land use activities, and then suggesting best practices for sustaining their populations and networks within which they interact in the agricultural and natural landscapes.

Results on the potential benefit of organic farming for biodiversity conservation that have been obtained in most parts of the world may not be directly applicable to the CFR landscape, as most of the results obtained so far have been in annual crop systems in post-glacial environments. However, the CFR agricultural landscape is predominantly perennial, with vineyards in particular dominating the agricultural land mosaic and competing with areas of high priority for conservation.

I report in chapter 2 that organic farming in vineyards supports the important monkey beetle pollinator taxon, while benefits from organic farming were not significantly higher for bees, unlike results from annual crop systems. Reduced level of disturbance in organic vineyards should be encouraged, since greater number of insect-flower interactions was also recorded in organic vineyards than conventional vineyards in chapter 3. However, more careful attention has to be paid to what conditions in vineyards support local farm scale species richness of important groups in these perennial systems. What would be of keen

interest is the between row vegetation in vineyards, as this holds promise for flower-visiting insects. Practices that encourage rich and abundant flower resources should be promoted as evidence is provided in chapter 3 of this study where flower resources were shown to influence flower visitation rates. The choice of such practices would encourage both indigenous plants and flowering weed species that can be managed sustainably to provide floral rewards for flower-visiting insects. Conservation strategies such as this will become increasingly important as evidence is accumulating that wild pollinator species are important in spite of the presence of managed pollinators. This will ensure adequate and sustainable provision of pollination ecosystem function in agricultural and natural land. In the CFR, both the conventional viticulturists, who manage their vineyard with an integrated approach that regulates intensity of disturbance, and the organic viticulturists who place premium on farmland biota could become role players in achieving this target of wine production that is not at the expense of biodiversity. The platform of Integrated Production of Wine initiative, with which most conventional viticulturists in the CFR are registered, and the Biodynamic Agricultural Association of Southern Africa (BDAASA), with which organic/biodynamic farmers are registered, could be explored to advocate these environmentally friendly approaches at the local farm scale.

Also of importance is the need to pay attention to ethics governing use of agrochemicals. This has to be done in a sustainable way where levels are not harsh at the farm level diversity of pollinators, and using agrochemical application methods that reduce drifts into neighboring natural and semi-natural. This is important, since the results in chapter 2 show the natural and semi-natural areas are the insurance for species richness and abundance of bees and monkey beetles, which may also be representative of other endemic insects in this region.

At a landscape scale, results in chapter 2 show that in biodiversity-rich regions, though minimal proportion of natural land (such 2% natural land in a radius of 1000 m found in this

study) in the landscape might provide insurance habitat for insect pollinators in the agricultural landscape. However, care has to be taken to ensure these natural habitat patches are not too isolated in the landscape. From the results obtained here, we recommend land patches not to be farther than 500 m, as 50% abundance of an important insect pollinator taxon was lost beyond this distance. This also means that vineyard plots in this region should not be too large, as this might increase the chances of increased isolation.

Although natural habitats are important for the population and interaction networks of flower-visiting insects, surprisingly, the networks in these natural habitats were not as nested as those of organic and conventional vineyards, showing that they may be less stable against species loss than the vineyard networks in the face of perturbation. This advances an argument to examine the state of the natural and semi-natural land patches in this region, with a view to assessing their potential to deliver the expected benefit for the conservation of insect pollinators and their interaction with flowering plants. Quality and size of natural and semi-natural land patches have been proposed in the literature as potential limiting factors for the benefits obtainable from them for biodiversity. The fynbos-dominated natural vegetation of the CFR has a fire-driven community process. Controlled burning is highly recommended for the natural patches that have not burned for a long time. This will bring back floral and insect life into some of these conserved areas that have become moribund.

The results in chapters 3 and 4 show that organic and conventional vineyards have the potential to halt both taxonomic and interaction network level biotic homogenization of flower-visiting insects. These findings are very important in view of increasing concerns over biotic homogenization of the world's biota resulting from intensive agriculture. Viticulturist and conservationist in the CFR can be assured that if current levels of environmentally friendly viticultural practices can be improved on in both organic and conventional vineyards, the negative effect of biotic homogenization that has resulted from intensive vineyard management approaches in other parts of the world can be averted in the CFR. This is

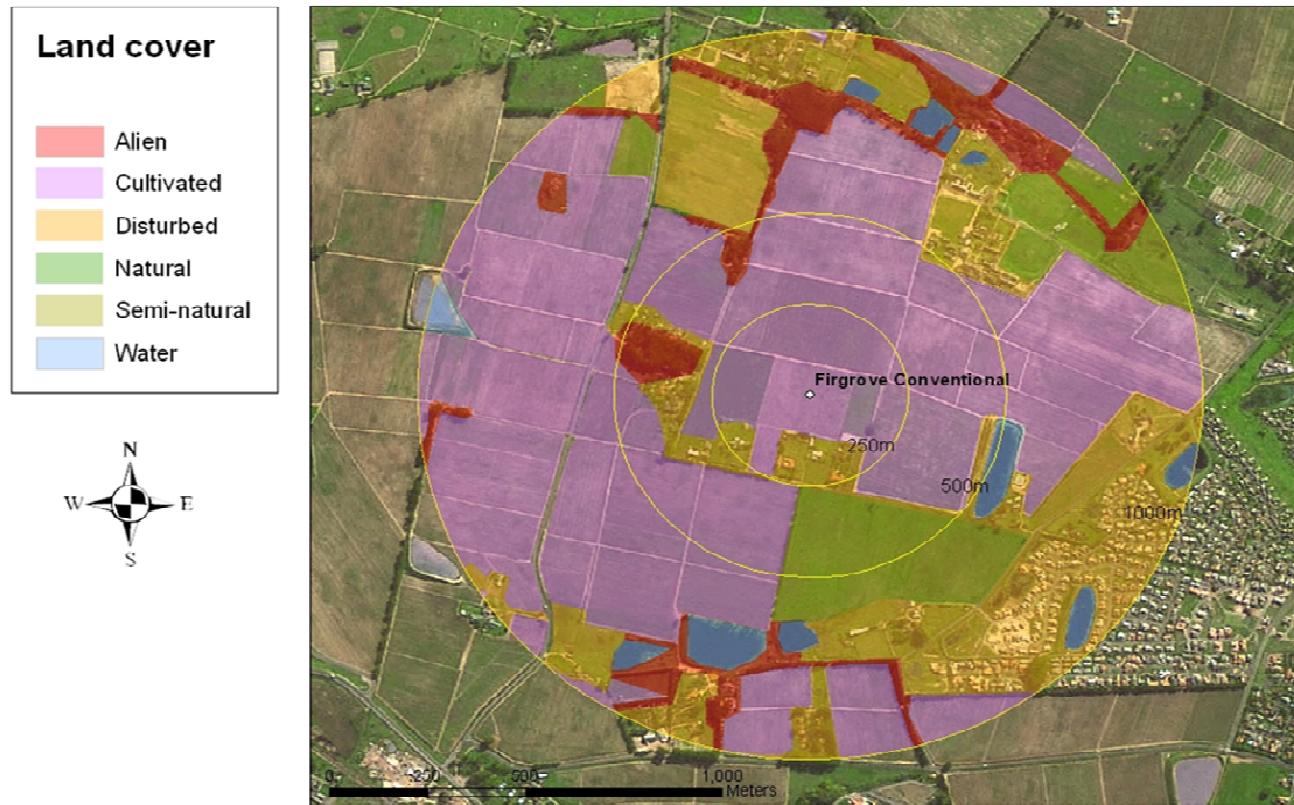
especially so as there is no replacement for the biota of this region if lost, due to the high level of local endemism. Caution has to be exercised, as mentioned earlier, not to allow current levels of intensity and disturbance to deteriorate, but to rather improve on it, as results of species turnover analysis showed difference in spatial turnover of the bee and flowering plant populations between the vineyards and the natural habitat. Progressively, the wine sector has shown interest in the conservation efforts in this region, with the ongoing partnership that exists with the conservation sector through the BWI and IPW.

Future research into whether benefits that have been found for species richness and insect-flower interactions are transferable for ecosystem services such as pollination and pest control, which has been investigated in other crops, are suggested. It will also be of interest to know what landscape elements are currently available to reduce isolation between natural and semi-natural land patches, and how these can be properly linked in the landscape mosaic to benefit regional biodiversity.

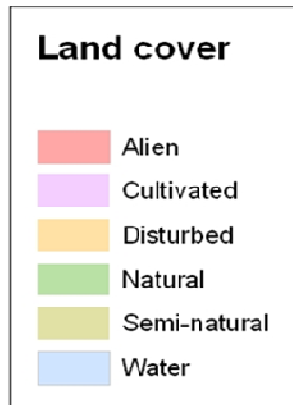


## Appendices

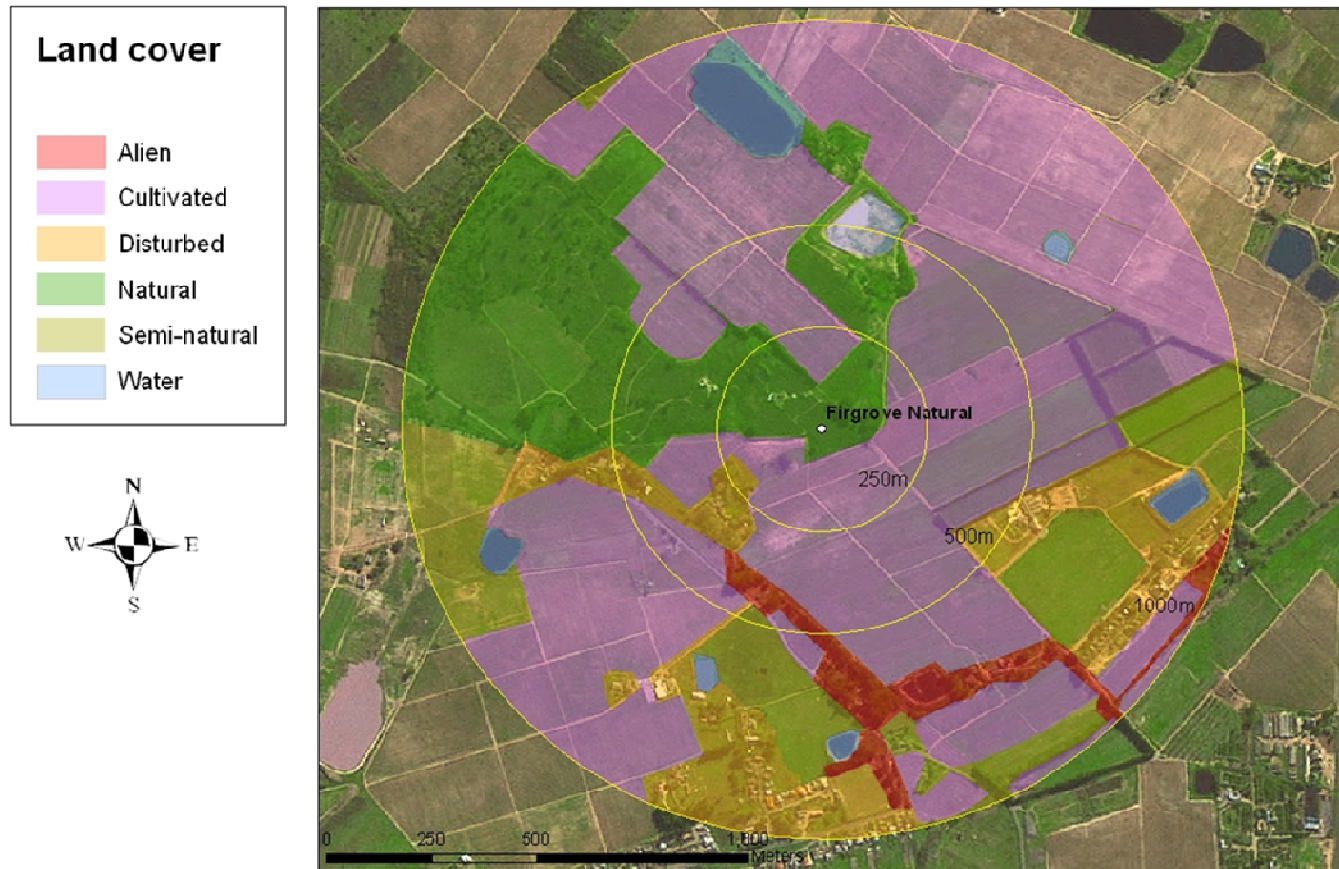
Field mapping of the landscapes mapping used in the computation of land cover shown in this appendices was done between November to December 2009. Data obtained were compiled by Estee Keyser, Department of Agriculture, GIS Unit, Elsenerg, Western Cape, South Africa. Images were obtained from SPOT5 2008.



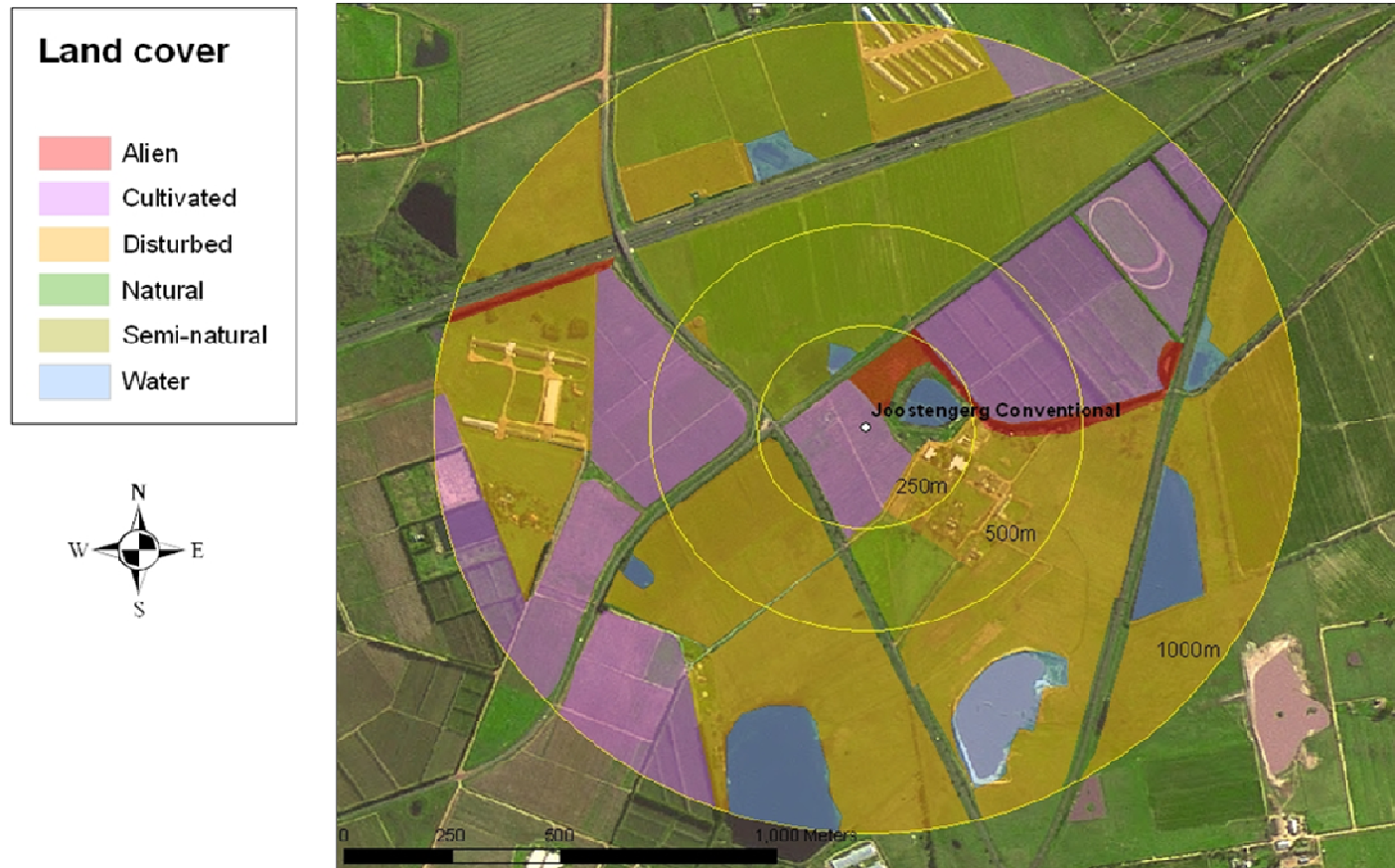
Appendix 1 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Firgrove landscape Western Cape, South Africa.



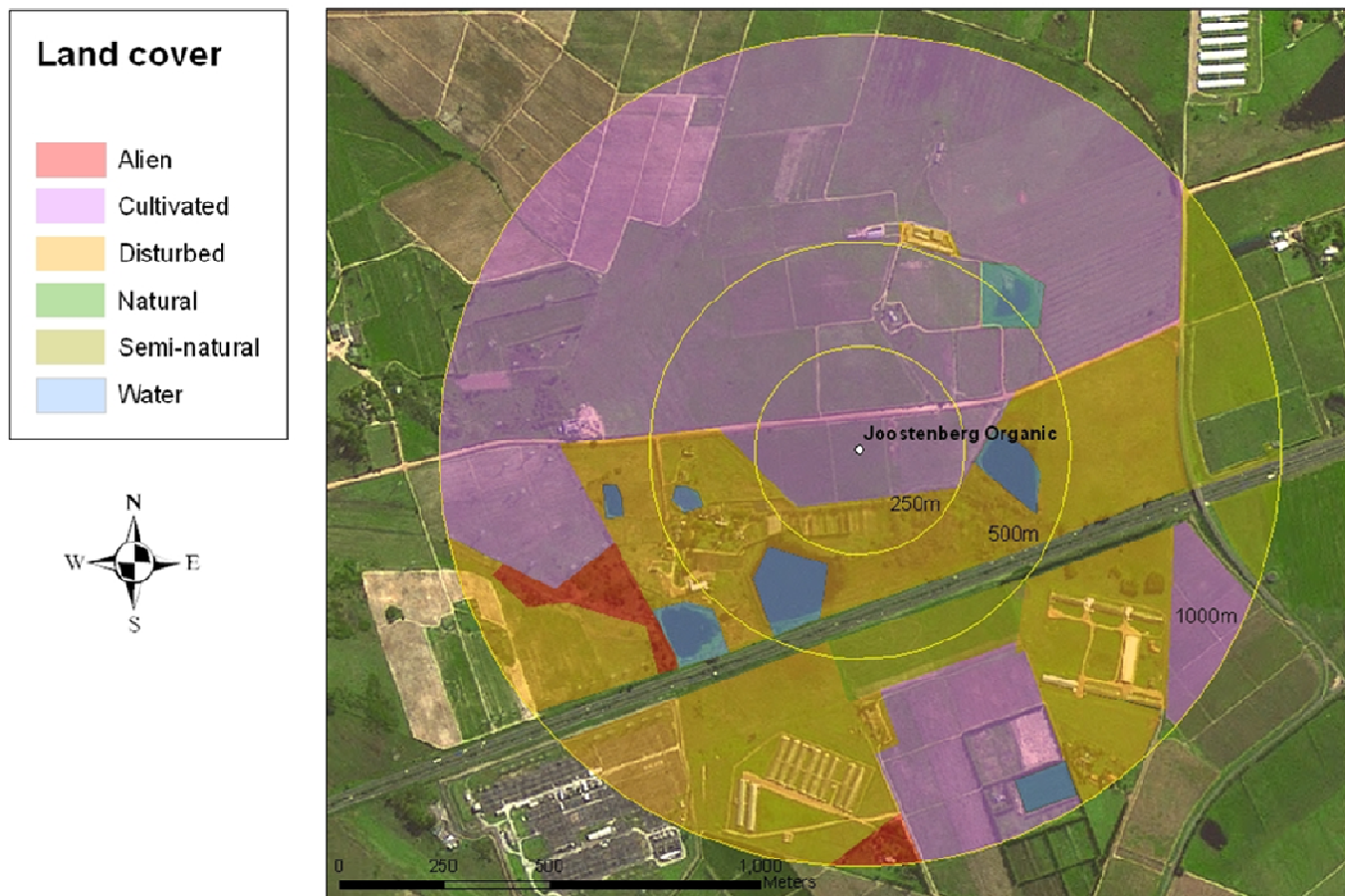
Appendix 2 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Firgrove landscape Western Cape, South Africa.



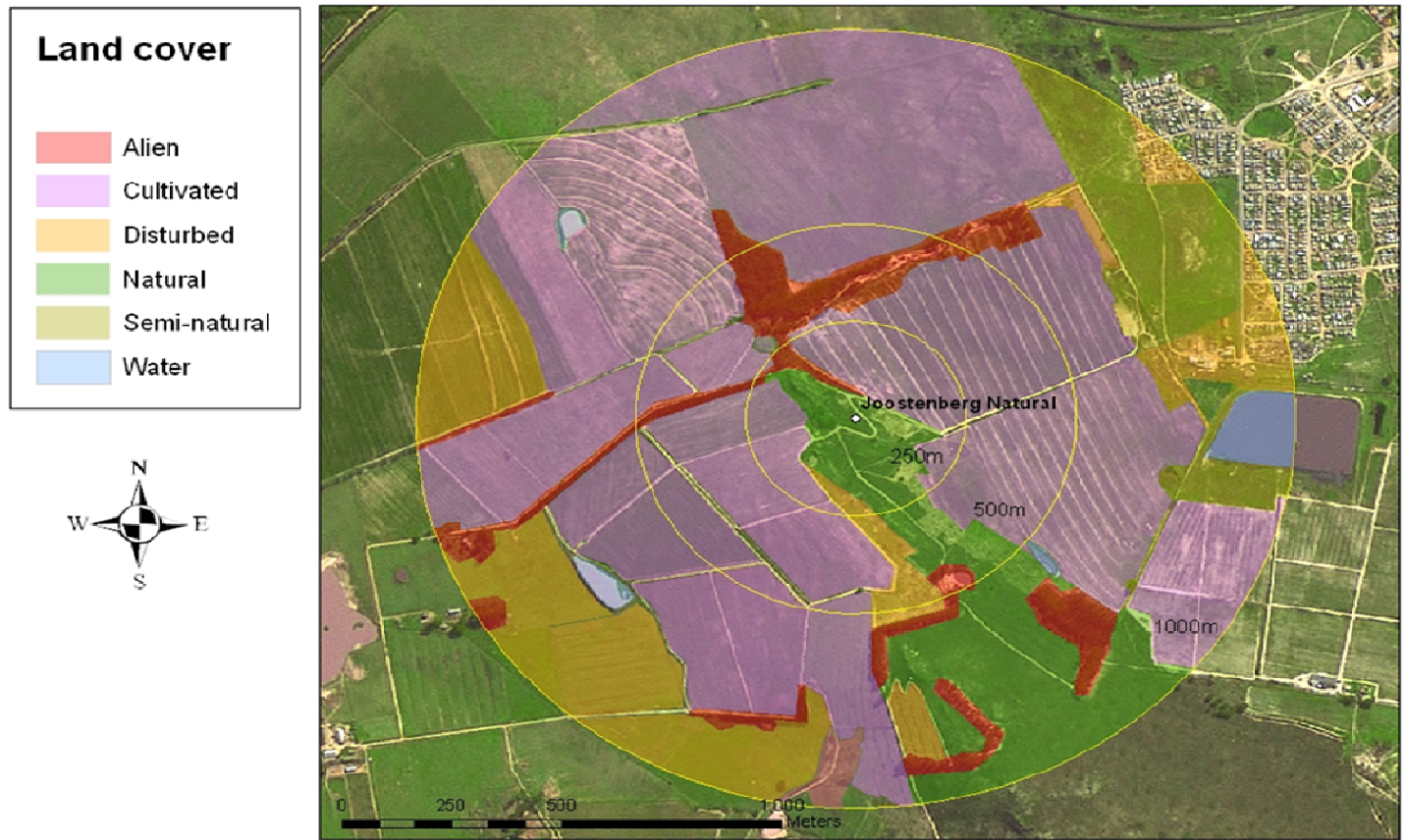
Appendix 3 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Firgrove landscape Western Cape, South Africa.



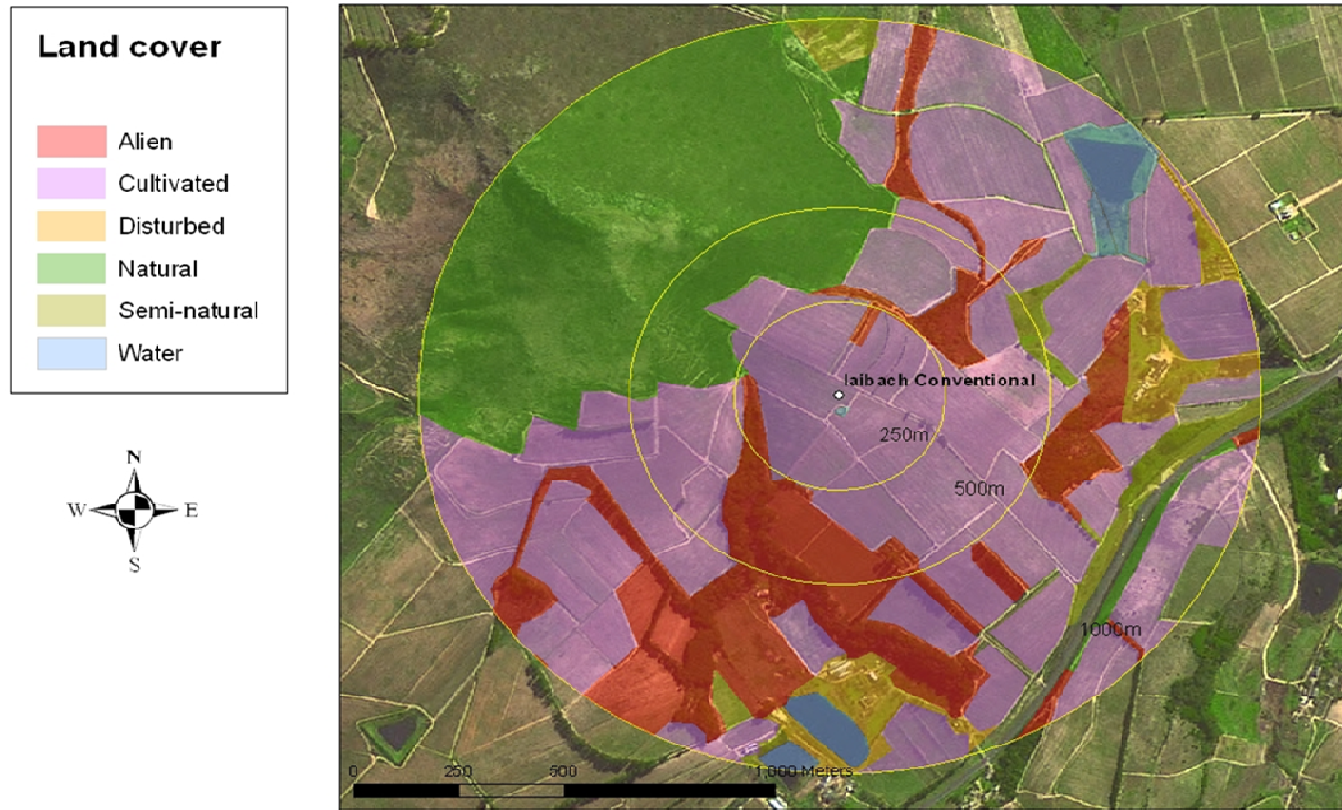
Appendix 4 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Joostenberg landscape Western Cape, South Africa.



Appendix 5 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Joostenberg landscape Western Cape, South Africa.

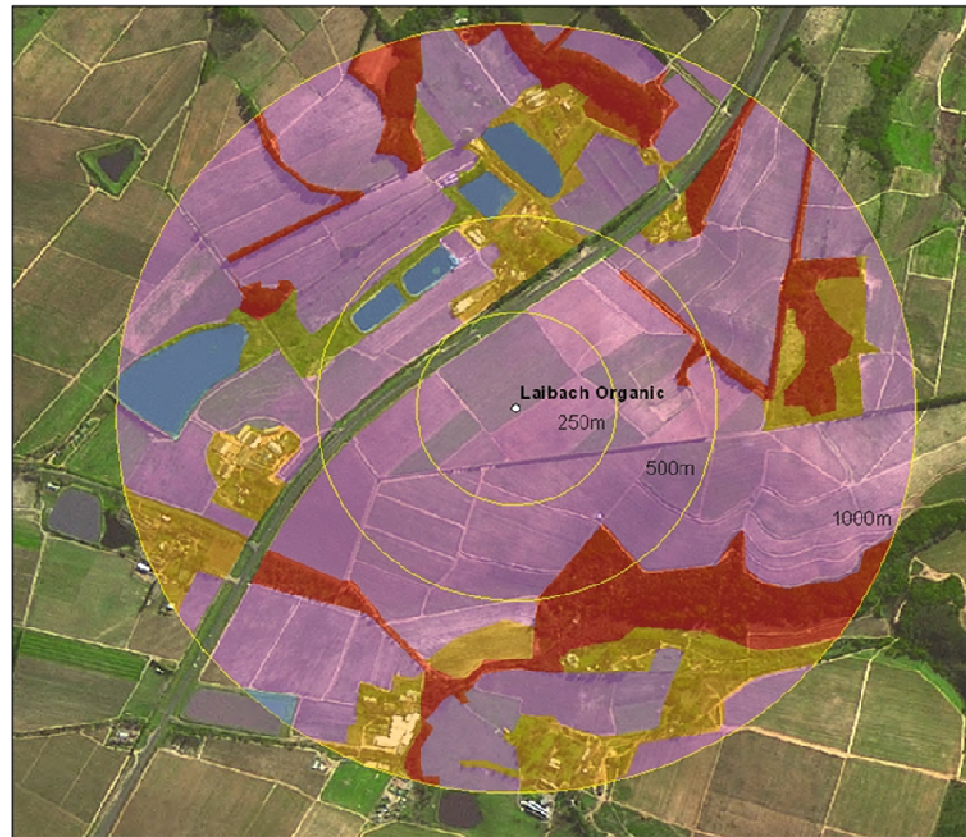
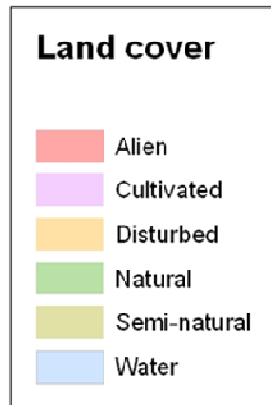


Appendix 6 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Joostenberg landscape Western Cape, South Africa.



Appendix 7 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Laibach landscape Western Cape, South Africa.





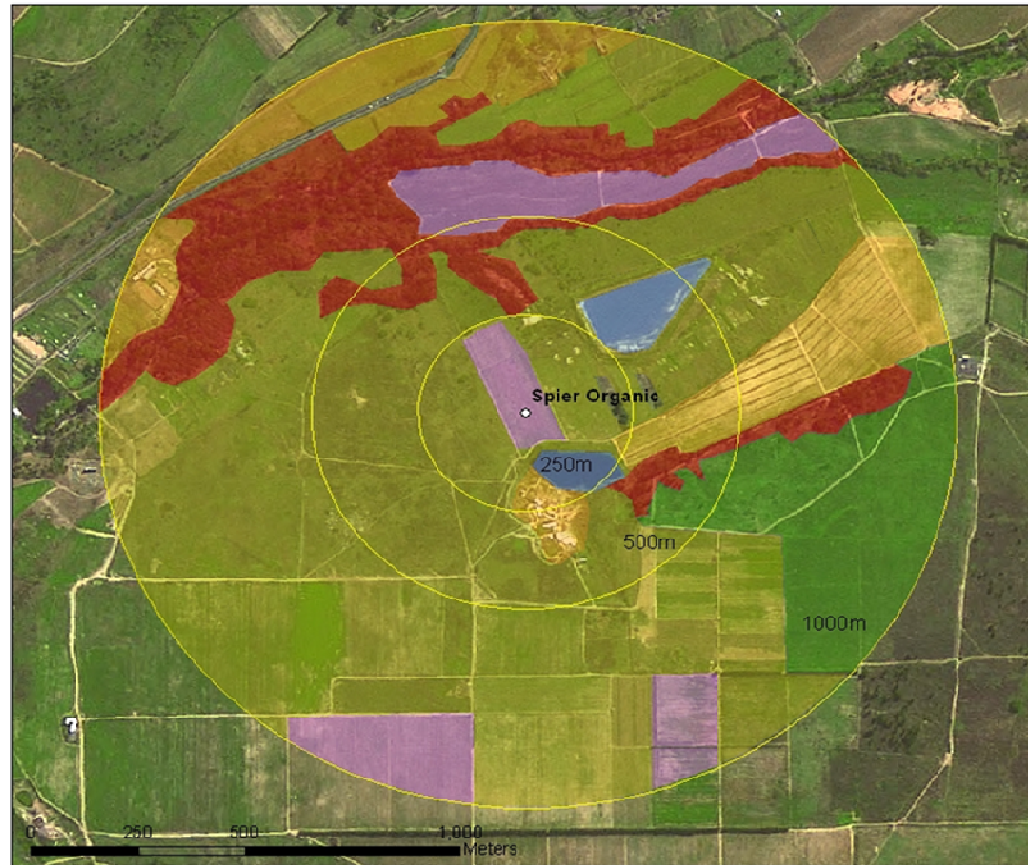
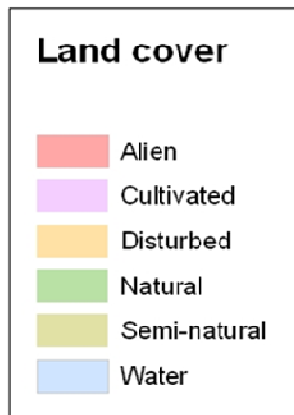
Appendix 8 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Laibach landscape Western Cape, South Africa.



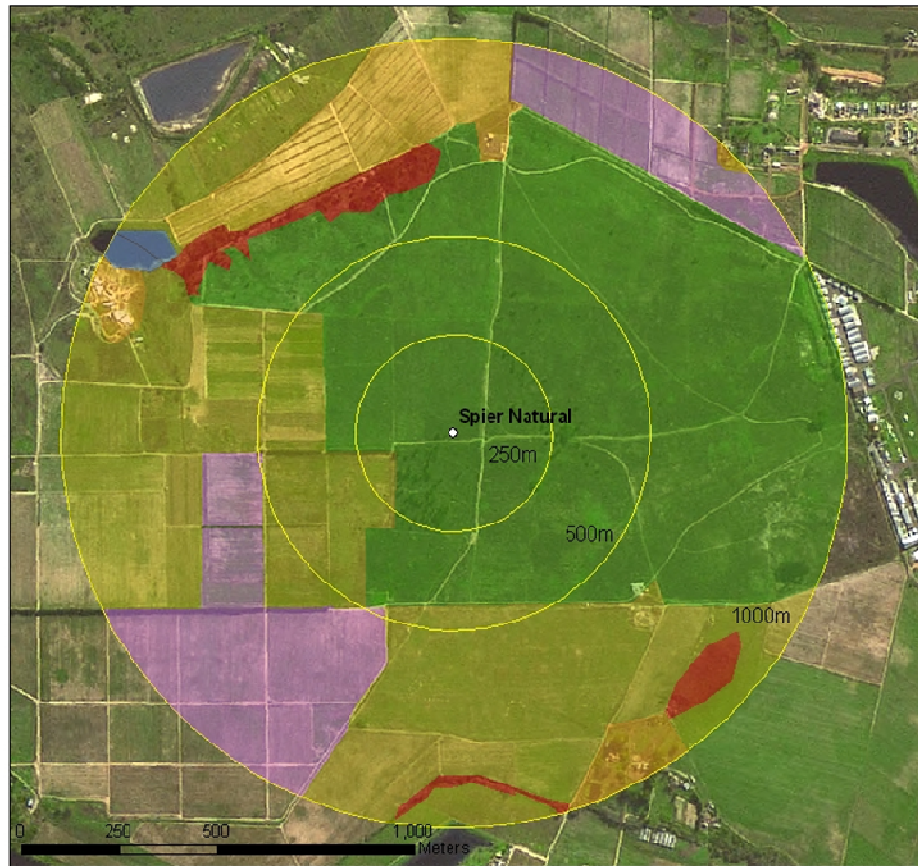
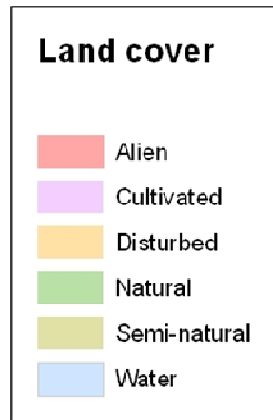
Appendix 9 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Laibach landscape Western Cape, South Africa.



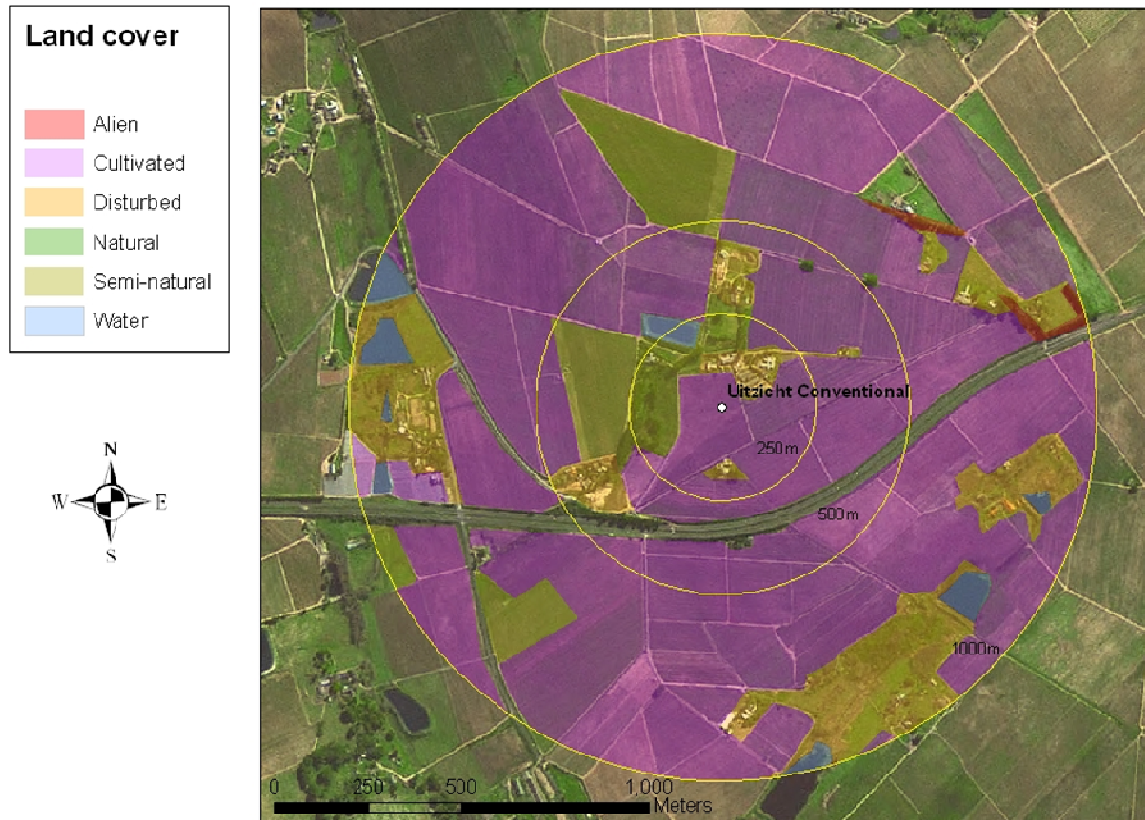
Appendix 10 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Spier landscape Western Cape, South Africa.



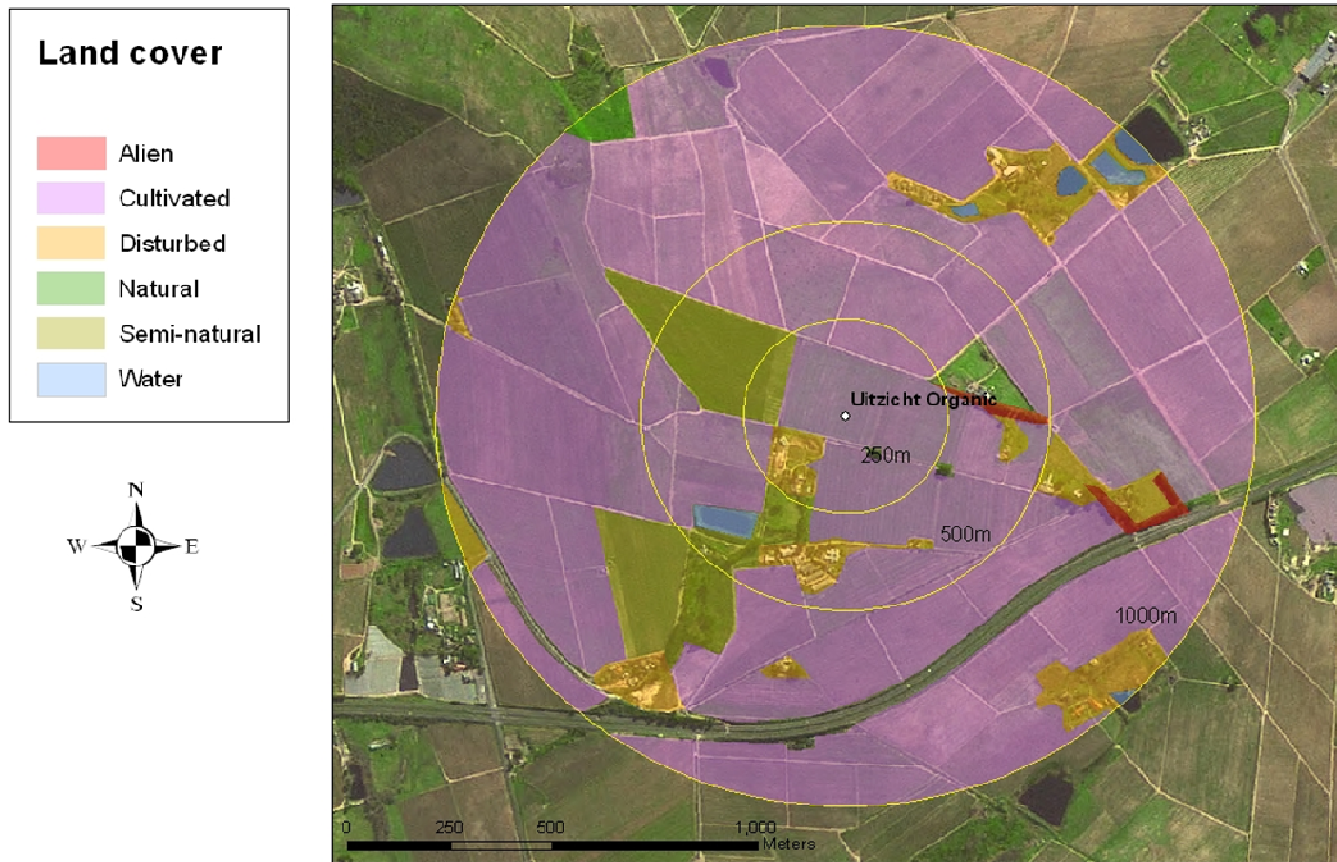
Appendix 11 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Spier landscape Western Cape, South Africa.



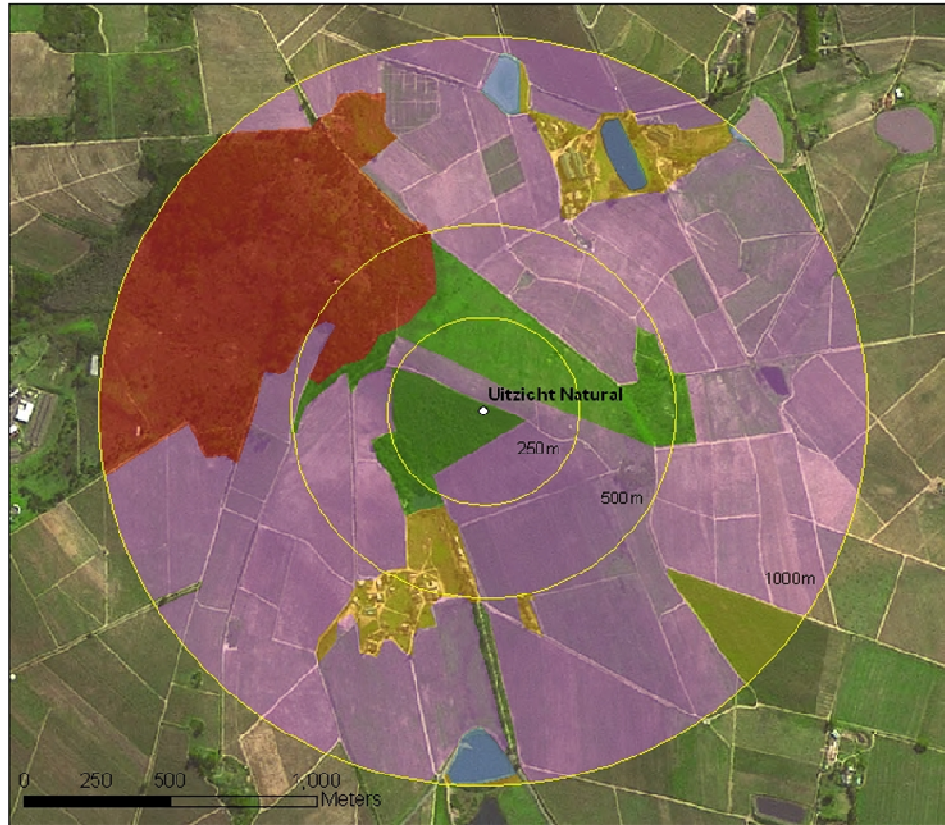
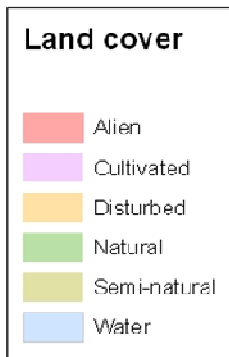
Appendix 12 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Spier landscape Western Cape, South Africa.



Appendix 13 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Uitzicht landscape Western Cape, South Africa.



Appendix 14 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Uitzicht landscape Western Cape, South Africa.



Appendix 15 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Uitzicht landscape Western Cape, South Africa.

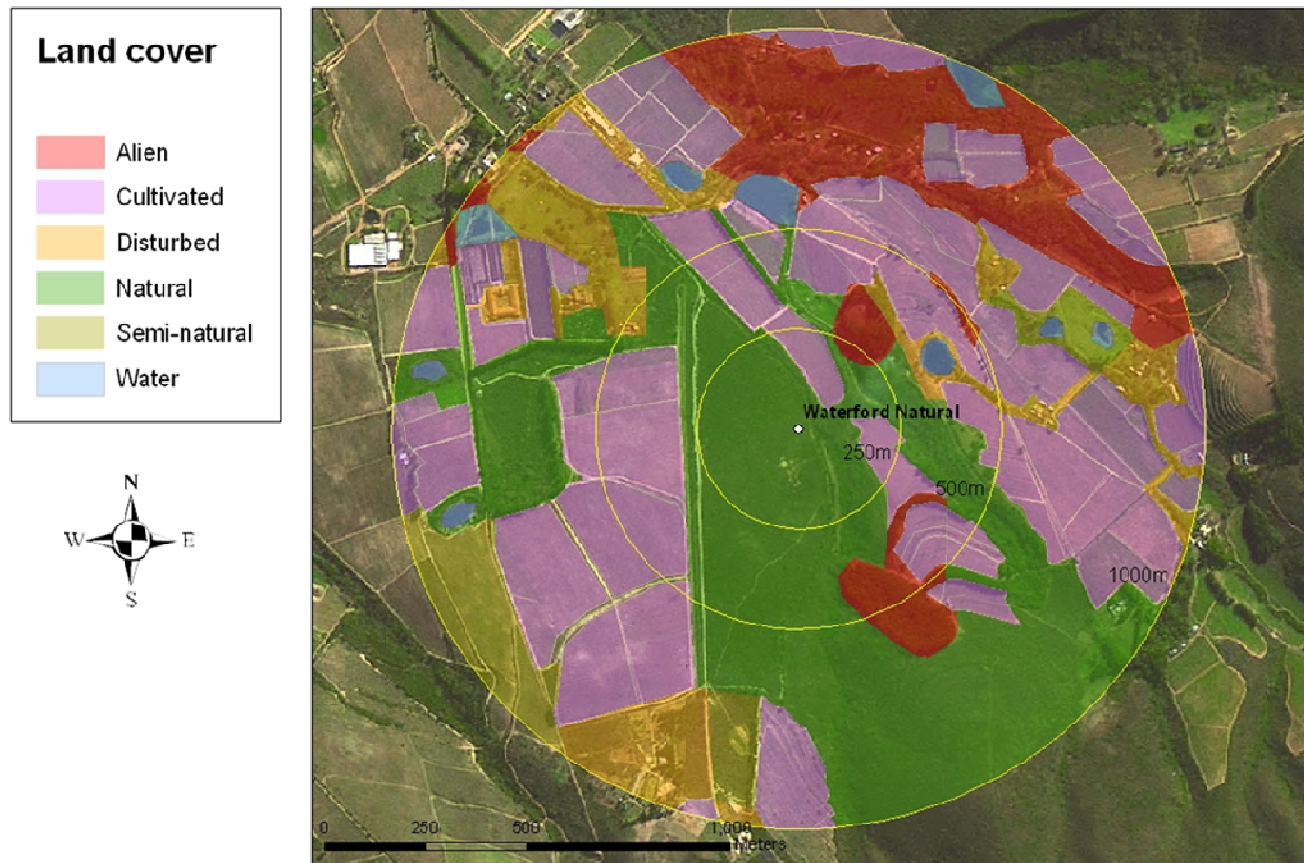




Appendix 16 Land cover within 250 m, 500 m and 1000 m radius of conventional vineyard in Waterford landscape Western Cape, South Africa.



Appendix 17 Land cover within 250 m, 500 m and 1000 m radius of organic vineyard in Waterford landscape Western Cape, South Africa.



Appendix 18 Land cover within 250 m, 500 m and 1000 m radius of natural vegetation in Waterford landscape Western Cape, South Africa.