Development of a Biotope Quality Index

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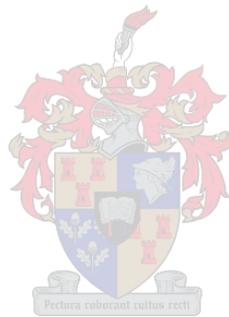
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Declaration

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E Bredenhand

Date ....27 September 2013....................

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ABSTRACT

As the world's human population increases, more pressure is placed on the management of natural resources. In response, we need an efficient means of monitoring, not only the quantity of these resources but also their quality. No comprehensive standard metric has been developed to assess environmental quality of a biotope, or to define the nature and extent of environmental degradation at this spatial scale. Currently in conservation management, various landscapes are being evaluated for spatial heterogeneity, by making use of species surrogates such as species richness, relative abundance, diversity indices and phylogenetic indices, as well as environmental surrogates. These values are then used towards conservation, where those systems with high intrinsic heterogeneity are usually considered more important than those with low heterogeneity at least when given the choice between the two. Yet, the actual quality of the biotopes within the landscapes is rarely taken into consideration. This study therefore develops and tests a Biotope Quality Index (*BQI*) to study this point in depth. The *BQI* makes use of arthropod assemblages as bioindicators of the level of disturbance within a biotope.

Firstly, I summarize the literature on the concept of environmental health, and define it as "An ecosystem is healthy, if it can sustain an optimal number of species with optimal population sizes and their ecological processes, thus providing and optimal heterogeneous sustainable system with sufficient resources, and indicated adequate resistance when under perturbational stress, but still allowing natural succession to take place" Against this background, I then review the use of certain Arthropoda as bioindicators, as arthropods are small, mobile, environmentally sensitive, easily sampled, and readily available. These features together make arthropods good subjects for testing the *BQI*.

I then compare the *BQI* with diversity indices currently used as surrogates of biotope quality. The outcome was that the *BQI* stood out as a significantly better indicator than the currently available indices for assessing environmental quality of a biotope. Furthermore, during the selection process, I also tested the use of guilds for *BQI* evaluation, and found that the scavenger (represented by Formicidae) and decomposer (represented by Collembola) guilds were the most significant. The effect of seasonality was also tested. I found the best results with the *BQI* were when data are pooled from all seasons of the year.

A case study, making use of the *BQI* evaluation, was conducted at a site in the Cape Floristic Region, South Africa (Jonkershoek Valley). *BQI* results suggested that the agricultural management and tourism within the locality might have an effect on biotope quality. This study has shown that use of the *BQI* is a useful and practical management tool for evaluating environmental quality of a biotope towards conservation management.

OPSOMMING

Met die vermeerdering van mense op die Aarde, wat meer druk plaas op ons natuurlike hulpbronne en omgewing is daar 'n aanvraag na doeltreffende maniere wat nie net die kwantiteit maar ook die kwaliteit van die hulbron evalueer. Geen betroubare standard bestaan om biologiese kondisies of die kwaliteit van n omgewing te meet nie. Heidiglik maak wetenskaplikes staat op die bepaling van diversiteit en ander voogde soos spesies rykheid en diversiteit indeksies as voog vir kwaliteit. Die waardes word dan gebruik binne die omgewings bestuur praktyke en bevooroordeel omgewings met 'n hoë diversiteit, terwyl die kwaliteit van omgewing skaarslik na gekyk word. Hiervolgens, onwikkel ons n Omgewings Kwaliteit Indeks (*OKI*), wat gebruik maak van Arthropoda saamestellings as bioindikator van die vlak van verval binne 'n omgewing. Verder sluit die tesis n literatuur studie van die omgesings gesondheid teorie, en die gebruik van arthropoda as bioindikators. As basis van die studie, definieër ons 'n gesonde omgewings as 'n omgewing wat' n optimal hoeveelheid spesies en hulle ekologiese prosesse kan handhaf, en daarom verwys na 'n diverse onderhoubare sisteem met genoegsame hulpbronne en kan genoegsame weerstand bied onder omgesings stres, maar gee geleentheid vir naturlike suksesie om plaas te vind

Ons het verder die *OKI* getoets teen ander diversiteit's indeksies, waar ons gevind het dat die *OKI* evaluering 'n statistiese beklemtonde verskil toon as bioindikator van omgewings kwaliteit. Verder het ons voorkeer getoets, in gedrags groupe en gevind dat die versamellaars groep (verteeenwoordig deur Formicidae) en die afbrekers groep (verteenwoordig deur Collembola) die beste resultate toon. Seisoene het ook 'n uitwerking en ons het gevind die groupeering van data ingesamel oor alle seisoene die beste resultate getoon.

'n Ondersoek studie wat gebruik maak van die *OKI* evalueering, was gekondakteer in die Jonkershoek valei en het getoon dat die landbou plaagbestuur en toerusme 'n negatiewe effek het op die omgewing. Verder het die *OKI* evalueering getoon dat aanplanting van Denne plantasies die kwaliteit van 'n omgewing verlaag. Die studie het verder getoon dat die *OKI* evalueering 'n betroubare evalueerings metode is vir die bestuur van 'n omgewings, terwyl die diversiteits indeksies nie geskik is as bioindikator van omgewings kwaliteit nie.

ACKNOWLEDGEMENTS

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Chapter 1

DEVELOPMENT OF AN BIOTOPE QUALITY INDEX USING TERRESTRIAL ARTHROPOD ASSEMBLAGES

1.1 INTRODUCTION

An ecosystem is a biological community of interacting organisms and their physical environment (Price *et al.,* 2011). We as humans do not know what the outcome will be when we randomly remove or disturb these interactions, yet we continue to do so for the benefit for our own species. Many of these interactions are vital not only to our survival, but also those of other species. If we are to manage current ecosystems for their long term future, we therefore need first to develop mechanisms for evaluating ecosystem integrity (its natural composition) and quality (its natural functioning). We can establish a sequence of procedures to either protect or restore our natural resources for generations to come. Currently, such a system exists in the medical field for evaluating human health (Rockström, 2011). Therefore, there is merit in adapting this concept for evaluating ecosystem health i.e. ecosystem integrity and quality.

Richardson *et al.* (1998) suggested management can be enhanced by the availability of various databases, legislation and management tools for planning, zoning and locating areas of high conservation status. Most important is the evaluation of the quality of these systems as well as their network ascendancy. Ulanowicz (1997) defined network ascendency as the product of the aggregate amount of material or energy being transferred in an ecosystem, multiplied by the coherency with which the outputs from the members of the system relate to the set of inputs to the same components.

A first step is to define what is meant by 'ecosystem quality' or, as is the aim here, 'biotope quality'(Chapter 2, p. 12-46). This is all part of establishing a clearer idea of the quality and health of natural environments for bringing our regulatory mandates in line with legislative procedures (Haskell *et al*., 1992). A 'biotope' is defined as the smallest geographical unit of the biosphere or a habitat that can be delimited by convenient boundaries and is characterized by its biota (Lincoln *et al*., 1998). No comprehensive standard has been developed to assess the biological condition, or measure the quality of

biotopes, or even to define the nature and extent of degradation at this spatial scale (Karr, 1992; 2004). Great confusion around the concept and relative terminology exist within the literature, and in Chapter 2 (p.12-46) I report these definitions and conclude with a functional definition that I could use for the underlining criteria needed to base the Biotope Quality Index (BQI) on (Chapter 4, p.111-131).

Currently, environmental quality is evaluated by making use of heterogeneity, in the form of various species surrogates such as species richness, relative abundance, diversity indices, phylogenetic indices, as well as environmental surrogates, at the landscape level (Faith & Walker, 1996; Clarke &Warwick, 1998; Reyers *et al*., 2002). These values are then used in the management of ecosystems, where systems with high heterogeneity are considered more important than systems with low heterogeneity. Yet, quality of the biotopes at the landscape level is rarely taken in consideration (Dennis *et al*., 2007). It is the aim in Chapter 4 (p.111-131) to explore environmental integrity and quality at the important managerial scale of the biotope and compare the developed index to the heterogeneity indices currently surrogated for as measurement of environmental health.

Relative species abundance and species richness are key elements of biodiversity (Hubbell, 2001). Species richness is the basic unit according to which an environment's heterogeneity is assessed. Yet species richness alone is principally an arbitrary number (being based on what is actually apparent and counted at the time of sampling). However, calculations for measuring the estimated species richness (the Chao index for example), can be used to give a more realistic estimation of the actual number of species present within a biotope (Henderson, 2003). Relative abundance of species refers to how rare or common a species is relative to other species in a given biotope or community (McGill *et al*., 2007). Relative abundance of species is described for a single trophic level, where species will potentially compete for similar resources (Hubbell, 2001). This is similar to Tokeshi's (1990) point that the approach to species abundance distributions uses niche-space, i.e. available resources, as the mechanism driving abundances of the species present.   In comparison, species diversity indices are statistical analyses which are intended to measure the differences among individuals of a data set consisting of various types of objects (Cover & Thomas, 1991).

Diversity indices assign different weights to factors such as proportion of individuals, evenness, species richness and abundance. The diversity indices most commonly used in

conservation ecology are the Simpson-Yule index (Simpson, 1949), Shannon-Wiener function (Shannon, 1948), Berger-Parker dominance index (Henderson, 2003), McIntosh diversity measure (McIntosh, 1967) and the Briliouin index (Stilling, 1999). All of these measures are important in ecology but only in conjunction with other indices to prevent bias. They can be applied to Bratton's (1992) climax theory, whereby a community is at its best (in perfect health) when the successional sequence has maximal biomass, the most complex nutrient cycles, greatest productivity, greatest species diversity, and is able to maintain itself indefinitely when freed from major disturbances.

One of the ways we can assess the quality of the environment is to look at the individuals present within the system. Bioindicators (a species or group of species that readily reflects the abiotic or biotic state of an environment, or represents the impact of environmental change on a habitat, community or ecosystem) are the obvious choice for such an assessment (McGeoch, 2007). Original use of the term 'biological indicator' in aquatic systems referred to detection and monitoring of changes in biota to reflect changes in the environment (Wilhm & Dorris, 1968). These individuals or assemblages of species can then be used to examine various factors at a number of levels of organisation including genetic, species or ecosystems levels (Noss, 1990), for example, ecosystem health at the biotope level. In Chapter 3 (p.47-110) a literature review on arthropods used as bioindicators is done to identify possible taxonomic groups that would make good bioindicators for the development of a Biotope Quality Index.

Insects in particular have been flagged as promising bioindicators for over three decades because of their significant contribution to global species richness, biomass and ecological function, as well as their responsiveness and extensive life history and behavioural diversity (McGeoch, 2007). Terrestrial insects from a variety of taxa have been used as bioindicators for various biotopes, habitats and environmental scenarios (Kremen *et al*., 1993; McGeoch, 1998).

Samways *et al.* (2010) identified some attributes in invertebrates that make them significant for the use of biomonitoring, namely:

- being small they are often highly sensitive to higly local conditions
- being mobile and reactive to changing conditions, they often respond by moving away from, or towards, adverse or optimal conditions respectively

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- with their usually rapid breeding rates and short generation times, they are often highly responsive numerically to changes, with their great variety of growth rates, life history styles, body sizes, food preferences, and ecological preferences
- they can, at the species level, often be linked to specific environmental variables
- fluctuations in their abundance provide an extra sensitivity layer over that of species richness for indicating subtle changes in environmental conditions
- many are relatively easy to survey

However, not all invertebrates show the same sensitivity to change, and therefore not all can be used as bioindicators on equal terms. McGeoch (1998) established nine steps involved in the identification, testing, and eventual adoption of a bioindicators.

1. Determine the broad objective (environmental, ecological or biodiversity indication)
2. Refine objectives by making them scenario-specific, and clarify the end-point (by identifying the final desired outcome of the process)
3. Select a potential indicator based on accepted *a priori* suitability criteria
4. Accumulate data on the proposed bioindicator using appropriate sampling or experimental design protocols
5. Collect quantitative relational data (weather, habitat quality)
6. Establish statistically the relationship between the indicator and the relational data (information on the environmental stressor of interest)
7. Based on the nature of the relationship, either accept (preliminarily) or reject the species, higher level taxon or assemblages as a potential indicator
8. Establish the robustness of the indicator by developing then testing appropriate hypotheses under different conditions
9. If the null hypotheses are rejected, make specific recommendations, based on the original objectives, for the use of the (now realized) bioindicator and further development of the bioindicator system

Another approach is to classify bioindicators into guilds, a concept defined by Simberloff & Dayan (1991) as a group of species that exploit the same class of environmental resources in a similar way, and according to Price *et al.* (2011), should exhibit similar ecologies. The term 'guild' groups together species without regard to taxonomic position, but rather on a significant overlap in their niche requirements (Simberloff & Dayan, 1991). Although taxonomic position is not important in defining a guild, we find in ecology that

guilds are often composed of groups of closely related species that all arose from a common ancestor and which exploit resources in similar ways as a result of their shared ancestry and evolutionary biogeography (Flannery & Thompson, 2007). Guilds are useful if, and only if, they provide us with generalizations that would be missed by taxonomic studies alone (Speight *et al*., 1999). In Chapter 6 (p.144-175) various guilds are compared to see if selecting a specific guild would improve the Biotope Quality Index.

Terrestrial arthropods divided into ecological guilds might include the herbivore, fungivore, carnivore, omnivore and decomposer guilds. Such division into guilds depends on the type of food upon which the organisms feed (plant, fungal or animal) as well as the status of the food (dead, recently deceased or alive) (Price *et al*., 2011).

Studies in phenology indicate that most invertebrates have short life spans, with variations in life stages to overcome diverse seasonal variations within an environment. Eggs and pupae are a developmental stage to overcome extreme cold or warm conditions (Speight *et al*., 2009). Palmer (2010) indicated that rainfall has a significant effect on invertebrate behaviour, and when rain is absent, certain species become more active, while other species are more active during the cooler, rainy season.   Furthermore, Thompson & Townsend (1999) showed that species richness, number of food web links, connectance strength, mean chain length, average number of links down the chain and prey:predator ratio within the food web, show significant variation across seasons. Similarly, various studies in aquatic systems have showed variation in results owing to seasonality (Bagatini *et al*., 2010), while other environments showed no differences (Vonk *et al*., 2010). Therefore, in Chapter 7 (p.176-187), seasonality and testing the effects of seasonal change are both considerations which must be taken in to account when developing an index which assesses environmental quality.

1.2 AIMS OF STUDY

The aims of this study are to:

➢ Summarize concepts of, and views on, environmental health, and to examine methods of measurement and interpretation of these concepts (Chapter 2, p.12-46).
➢ Develop a definition of 'environmental health' for the purpose of creating an index to assess biotope health (quality and integrity)(Chapter 2, p.12-46).
➢ Undertake a literature review on the use of arthropod as bioindicators (Chapter 3, p.47-110)

➢ Develop and test an environmental indicator system making use of arthropod assemblages to evaluate biotope quality so as to be able to develop a Biotope Quality Index (*BQI*) (Chapter 4, p.111-131).

➢ Test the concept of the *BQI*, by comparing correlations between the *BQI* and various currently used biodiversity indices to an non-parametric scale of environmental integrity and level of disturbance (Chapter 5, p.132-143).

➢ Compare various arthropod guilds to determine whether there are similar trends among the guilds when using the *BQI* (Chapter 6, p.144-175).

➢ Identify making use of the *IndVal* evaluation, the best possible taxa that are suitable for the use in the *BQI* (Chapter 6, p.144-175).

➢ Test the effect of season on the performance of the *BQI* (Chapter 7, p.188-210).

➢ Illustrate the use of the *BQI* by conducting a case study as an example (at Jonkershoek, Western Cape, South Africa) (Chapter 8, p.188-210).

1.3 PROGRESSION AND JUSTIFICATION OF STEPS WITHIN THE STUDY

1. A literature review was done on the concept of environmental health and relative studies.

2. Due to the extreme confusion and variation within the terminology used to define environment health, I came up with a working definition taking into consideration all previous literature around the concept.

3. Jonkershoek Valley was then selected for its range of available and accessible sites from expected high quality within the conservation areas, to the poor quality pine plantations.

4. Thirty sites across various biotypes were selected and evaluated by categorisation to the level of disturbance that gave an inverse reference to the integrity and quality of a biotope. Various physical measures and co-ordinates were also taken at this stage.

5. Pitfall sampling was used to sample the ground dwelling invertebrate assemblages over all four seasons (Spring, Summer, Autumn, Winter) of 2006. All specimens were sorted and counted to morphospecies level.

6. A literature review of the arthropods used as bioindicators within the last decade was done to identify possible known taxa that work well as bioindicators to narrow down the data set.

7. From step 5, Formicidae and Collembola were identified as good indicators due to there abundance and availability and I proceeded to develop the *BQI*

8. From a statistical perspective, I considered six possible variations of the basic calculation concept derived from my definition.

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9. The Formicidae and Collembola data were used to compare and select the best calculation according to how these calculations correlated to the level of disturbance.

10. The final selected method of calculation was then used to compare if the *BQI* does show better correlation with the level of disturbance from the various heterogeneity indices currently used as surrogates for environmental health.

11. After success with the results in step 10, the rest of the ±97000 arthropod specimens was sorted and identified to morphospecies

12. During the literature review on bioindicators, the best representatives of each feeding guild were identified and compared to see if guild selection is optional to better the *BQI*.

13. IndVal evaluation was done on all morphospecies and compared to species identified through *BQI* species identification

14. I also had a concern that the season in which sampling was done would have an effect on the results of the *BQI*, and therefore I tested the variation within the various data from each season and the combination of all seasons.

15. Taken in consideration all the results of the previous steps, I finalised the *BQI* and use the full data set to identify only specimens with positive correlation to quality and using these morphospecies in a form of a case study to illustrate how the *BQI* can be used.

[This thesis was written in this order as independent articles and not after every step was completed.]

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Chapter 2

ENVIRONMENTAL HEALTH: THE NEED TO PUT PHILOSOPHY INTO ACTION

Procedures are needed to either protect or restore natural resources for the future. It is imperative therefore that we firstly develop a means for evaluating ecosystem quality and integrity. Currently such an evaluation system exists in the medical field, one that evaluates human health. A possible step forward is now to adapt this system for use in ecosystem health monitoring. This chapter reviews definitions and concepts of ecosystem health and summarises approaches for evaluating and monitoring ecosystem health, as well as how to interpret these data. This chapter points out the extreme confusion in literature around this topic by reporting various opinions and definitions of relative and similar concepts. I conclude with a final definition taking in consideration previous definitions in literature: a healthy ecosystem is one that is healthy when it can sustain an optimal number of species with optimal population size and sustain their ecological processes, so providing an optimal heterogeneous sustainable system with sufficient resources, and has adequate resistance when under perturbational stress while still allowing natural succession to take place.

KEYWORDS: Diversity indices; Disturbance; Environmental health; Integrity; Network ascendancy

2.1 INTRODUCTION

Consciousness has enabled us, as humans, to think of the future and, theoretically at least, to improve chances of our long-term future. In essence, consciousness has led to us becoming a numerically superior keystone species. However, instead of establishing a homeostasis with our neighbouring species, we have over-used natural resources for our own benefits and to the detriment of other species. Our actions have changed the world too rapidly for natural change to keep pace, and as a result, nature will only be able to re-establish some pre-eminence if we can consciously construct a new and more adaptive set of terms and concepts for our own survival and for the wellbeing of our surroundings (Nortan & Steinemann, 2001). Maturana and Verela (1980), as well as Di Paolo (2005), point out that any structural changes that a living system may undergo while maintaining its identity must take place in a manner determined by, and subordinate to, its defining autopoiesis. An autopoietic system is a system organised as a network of processes of production of components that produces the components which, through their interactions and transformations continuously regenerate and comprehend the network of processes that produces them and constitute it as a concrete unity in the space in which they exist by specifying the topological domain of its realization as such a network. Therefore, a living system that losses its autopoiesis becomes disintegrated as a unity and looses its identity, and ultimately its existence.

A middle ground now needs to be found, where humans take, but also provide and protect. For us to protect ecological integrity and therefore the integrity of human society, we must first understand and appreciate the requirements of earth's biota while at the same time the desires of human society. These needs will not always be convergent but at least they must be interdependent (Raudsepp-Hearne *et al*., 2010). Furthermore, this compromise will require management and constant research, so to maintain as much of the land's original status as is compatible with human land-use and should be modified as gently and as little as possible (Callicott, 2000), a state that Leopold (1949) called harmony between man and land (Euliss *et al*., 2008).

Norton (2009) suggested a dynamic contextualist paradigm that focuses on the goal of protecting biological complexity, which must be the focal point of environmental management. He furthermore relates this complexity to self-organization, with these characteristics then being the essence of ecosystem health and integrity.

With this basic concept, we now need methods to evaluate the status of our existing surroundings. Leopold (1941) identified two entities in which the unconscious automatic processes of self-renewal have been supplemented by conscious interference and control. One of these is human (medicine and public health) and the other is land (agriculture and

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conservation) (Callicott, 1992), enabling us to borrow the metaphor for physical health from human beings and then use it as a tool to conceptualize environmental health as a method of quality evaluation (Rapport & Singh, 2006).

Use of the ecosystem health metaphor to support sustainable development and assist the public's general understanding of the functioning of ecosystems has been strongly articulated over the years (Rapport, 1992, 1995a, 1995b; Rapport *et al*., 2001a, 2001b, 2003, 2009; Rapport & Singh, 2006). The goal of this dynamic process is to protect the autonomous, self-integrative processes of nature as an essential component in an ethic of sustainability (Haskell *et al*., 1992).

This chapter summarises the concepts of, and the views on, environmental health, as well as exploring methods of measurement and interpretation of these notions. Most methods are used incorrectly by various studies, and clarity to its use and interpretation is needed. Similarly, there is confusion around the terminology and definitions associated with the concept of environmental health. This chapter thus aims to combine the various concepts and definitions into one final practical definition that can be used to develop a way of quantifying ecosystem health, that can be used in conservation and restoration ecology.

2.2 METHODS (WAYS OF MEASURING ENVIRONMENTAL HEALTH)

In medical practice, the following sequence of procedures is followed during the evaluation of human health. Firstly, symptoms are identified and vital signs measured. From these observations, provisional diagnoses are made which are then verified through a series of tests, before a final prognosis is made. Finally, treatment plans can be prescribed, executed, and monitored (Haskell *et al.,* 1992). This same concept can be used during the evaluation of environmental health.

Prognosis is supposed to precede treatment, and treatment can only occur in the context of theoretical knowledge present, but when such knowledge is limited, diagnosis takes priority over treatment on the grounds that treatment will misrepresent symptoms, thereby restraining the accumulation of knowledge about the medical condition and delaying the development of a cure according to the theory of historical therapeutic nihilism (Hargrove, 1992). With this theory, the aim is to rather leave the state of illness untreated, and let it go its own way, so that there is either unaided recovery, or there is adaption or there is extinction. This uncertainty can be prevented when better knowledge exists on the present state of being. Therefore, regular monitoring of ecosystems and regulation of sustainability is needed. The diagnosis process has seen many diagnostic tools to enable these types of evaluations. Such tools include the measurement of species richness, relative abundance, evenness, diversity indices,

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integrity, network ascendancy (or more specifically, connectance), vigor, interaction strength, stability and variability.

2.2.1 Identification of symptoms

The first step is to evaluate the state of being, where there is determination of how far the current state is from the original state at a given location with a characteristic physical appearance (i.e. biotope). This is done through recording of potential negative factors present and compares these factors to previous evaluations in the literature (if available). This process is defined by Samways (2005) and known in conservation ecology as triage (Figure 2.2.1), a system that correlates ecological integrity with the disturbance level and evaluates the necessity and feasibility of restoration.

2.2.2 Measurement of vital signs

The level of natural heterogeneity is often used as the cornerstone of environmental evaluation, where the number of species and their relative abundance, are important factors and a baseline in the evaluation of environmental health (Lu & Li, 2003). This fundamental assessment enables us to quantify the current situation using for example, diversity indices, to better compare between sites. Diversity indices measure a combination of evenness and richness with a particular weighting for each. Such diversity indices include: Simpson-Yule index; Shannon-Wiener function; Berger-Parker dominance index; McIntosh diversity measure; and the Brillouin index (Henderson, 2003).

For comparisons between unequal sized samples, it is recommended to use rarefaction, by calculating the number of species expected from each sample if all the samples were reduced to a standard size (such as 1000 individuals) (Chao et al., 2005). Assessment of species richness requires use of a species accumulation curve, by plotting the accumulative recorded number of species through the addition of new samples (Henderson, 2003). Estimated species richness can also be calculated by making use of nonparametric estimators like the Chao index, $\hat{S}_{max} = S_{obs} + (a^2 / 2b)$, where a and b are the number of species represented by one and two individuals respectively and S_{obs} is the actual number of species observed. The formula can still be used with presence-absence data by defining a as the number of species in one sample only and b as the number of species in two samples (Henderson, 2003). Relative abundance is the simple count of individuals within each taxon, and is used to compare samples that have been retrieved in the same way (Schneider, 2009).

ECOSYSTEM RESTORATION TRIAGE

Ecological integrity irretrievably lost	Ecological integrity restorable	Ecological integrity intact
Do nothing – restoration not possible only regreening, rehabilitation or ecological landscaping	Do something – the route where restoration is feasible	Do nothing – restoration not necessary

Intense and frequent; mostly anthropogenic

Intense and very infrequent or mild at any time

Disturbance level

Urbanization

Suburbanization

Agroforestry

Recreational areas

Greenways and habitat webs

Reserves with management

Reserves left alone

The original state

Rehabilitation

Regreening

Ecological landscaping

Restoration

Low   **Ecological integrity**   **High**

Compositional, structural and functional, biodiversity radically different from the original state. Ecological succession and evolutionary development on a changed, anthropogenically determined course

Compositional, structural and functional, biodiversity in its pre-anthropogenic original state, but nevertheless dynamic. Ecological succession and evolutionary development following a course uninfluenced by humans

Figure 2.2.1 A conceptual model of ecosystem restoration triage according to Samways (2005)

Estimated richness is calculated using the formula $E(S) = \sum_{i=1}^{S} \left\{ 1 - \left[\binom{N-N_i}{n} \middle/ \binom{N}{n} \right] \right\}$,

where $E(S)$ is the expected number of species in the rarefied sample size, N is the total number of individuals in the sample to be rarefied, and N_i is the number of individuals in the ith species in the sample to be rarefied, summed over all species counted. The term $\binom{N}{n}$ is a combination that is calculated as $\binom{N}{n} = \frac{N!}{n!(N-n)!}$ (Stilling, 1999). Shortcuts for this calculation exist by making use of richness indices that are just basic ratios of number of individuals per species. These include the Margalef richness index $R_1 = (S-1)/\ln N$ (Margalef, 1969), or the Menhinick richness index $R_2 = S/\sqrt{N}$ (Menhinick, 1964), or the richness index calculated by $R_3 = S/\log N$, where S is the number of species and N the number of individuals (Magurran, 2004).

Equitability, or evenness, is the pattern of distribution of the individuals between the species (Henderson, 2003) and can be best displayed by plotting the log number of individuals (relative abundance) of each species against the rank, where the rank is simply a number that gives the position of the species in a table of abundance, so the most abundant has a rank of one, the second most abundant two, and so on (Henderson, 2003).

The Simpson-Yule index (D) describes the probability that a second individual drawn from a population would be of the same species as the first (Simpson, 1949). The index is given as $D = \frac{1}{C}$ where $C = \sum_{i}^{s_{obs}} p_i^2$ and $p_i^2 = \frac{N_i(N_i-1)}{N_T(N_T-1)}$, where N_i is the number of individuals in the ith species and N_T the total individuals in the sample (Simpson, 1949). The Shannon-Wiener function (H), also known as the Information-theory index, determines the amount of information in a code and calculated by: $H = -\sum_{i=1}^{S_{obs}} p_i \log_e . p$ (Henderson, 2003), where p_i equals the proportion of individuals in the ith species.

The Berger-Parker dominance index (d) is calculated from the ratio of the number of individuals in the sample belonging to the most abundant species (N_{max}), divided by the total number of individuals caught (N_T): $d = \frac{N_{max}}{N_T}$ (Henderson, 2003).

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McIntosh (1967) created the dominance index called the McIntosh diversity measure (D) calculated by $D = \dfrac{N-U}{N-\sqrt{N}}$ and $U = \sqrt{\sum n_i^2}$, where N is the total number of individuals in the sample and n_i is the number of individuals belonging to the ith species (Henderson, 2003).

When the randomness of a given sample cannot be guaranteed the Brillouin index (H_B) can be used $H_B = \dfrac{\ln N! - \sum \ln n_i!}{N}$, where N is the total number of individuals and n_i the number of individuals in the ith species (Stilling, 1999).

2.2.3 Provisional diagnosis

At this stage, the observed factors, aided by the triage concept will give a preliminary assessment of the level of ecological integrity and level of disturbance, and indicate whether it is financially beneficial and practical to conduct a further, more detailed evaluation or whether the preferable option is to accept the route of historical therapeutic nihilism (Hargrove, 1992). Depending on your viewpoint, such research might be directed towards either detecting symptoms of pathology or to detect the ecosystem's ability to repair itself when subjected to controlled stress (Rapport, 1992; Samways *et al.*, 2010).

2.2.4 Verification tests

The test for network ascendancy is a test for quality, and is defined by Costanza (1992) as an information-theory measure, combining average mutual information (a measure of connectedness) and the system's total throughput as a scaling factor (Scharler, 2009). Ulanowicz (1992) describes it as the product of two factors, one estimating the level of system activity, and the other as capturing the degree of trophic organization. This network ascendancy concept was derived by Odum (1969) on the synopsis of the trends apparent in ecological succession (Scharler, 2009). Odum's list of twenty-four attributes of mature systems is focused at various levels of the hierarchy from the individual organism to the whole ecosystem, and includes such indicators as biochemical diversity, gross production, community respiration quotient, niche specialization and information, as well as organism size (Odum, 1969; Scharler, 2009). Ulanowicz (1992) then grouped these properties into four categories: greater species richness, more niche specialization, more developed cycling with feedback, as well as greater overall activity (Scharler, 2009). These four categories were then integrated into a single index for measuring the network ascendancy (Ulanowicz, 1992). For a community to be healthy and sustainable, the system must uphold its metabolic activity

level in addition to its internal structure and organization (network ascendancy) and must be resilient to outside stress over a time and space frame relevant to that system (Lu & Li, 2003).

Another way of calculating ecosystem health is with the formula $HI = VOR$ (Costanza, 1992; Su *et al*., 2009), where HI is known as the System Health Index. System vigor represents V, a cardinal measure of system activity, metabolism, or primary productivity, O is for system organization, represented by diversity and connectivity, while R represents the system resilience index, a 0-1 scale of relative degree of system's resilience (Costanza, 1992). Rapport *et al*. (2001b) used interaction strength as a measure of connectedness, seen as the mean magnitude of interspecific interactions, whereby the size of the effect of one species' density on growth rate of another species is calculated by the number of interspecific interactions divided by possible interspecific interactions. One aspect of the diversity is measured by the variability, whereby the variance of population densities over time or associated measures such as standard deviation or coefficient of variation (SD/mean) (Bhujel, 2008).

Holling (1986) defined resilience as a system's ability to maintain structure and patterns of behavior in the face of disturbance (Norris *et al*., 2008). Resilience can be calculated by measuring how fast the variables return to equilibrium following perturbation. Resistance on the other hand, is measured by the degree to which a variable is changed following perturbation (Rapport *et al*., 2001b). In these scenarios, perturbation is seen as the change to a system's contribution or environments outside normal range of variation (Costanza, 1992).

2.2.5 Prognosis and prescription of treatment plan

Using the above mentioned data, analysis can be done through the use of numerical models, to predict the best scenarios for further rehabilitation to re-establish the original state, as well as to determine the most cost effective, least time consuming version of treatment that would insure statistically the best results (Anselme *et al.,* 2010).

2.2.6 Execution phase

The execution phase consists of either a restoration aspect in locations that need to be repaired to a more natural state, and/or the management of the available natural resources. The management section can be further divided into three main types of management approaches: coarse-filter, fine-filter, and general ecosystem management (Carignan & Villard, 2002). The earliest advocacy of restoration was when farmers let their former fields succeed to forest to restore fertility of the soil (Hobbs & Suding, 2009). This approach is the basis of present practices of ecological restoration which lie deep within the conservation and

preservation movements of the 19th and early 20th centuries, when ecologists viewed natural succession as leading to a climax community, which once established, was considered the 'stable and final biotic assemblage' in a particular location (Hobbs & Suding, 2009).

Coarse-filter approaches aim mainly to maintain entire communities by protecting large extents of habitat and by preserving the key ecological components and processes that sustain these communities (Caringnana & Villard, 2002). Fine-filter approaches, in contrast, focus on the protection of a certain number of elements (species), supposing that their status within the ecosystem reflects the status of other elements associated with them (Meffe *et al.*, 2002).

2.2.7 Management phase

The last of the management categories is the ecosystem management approach, a phase with explicit goals which are executable by implementation of policies, protocols, and practices. Such approaches would be made sustainable by monitoring and research based on the best perception of the ecological interactions and processes essential to sustain ecosystem composition, structure, and function (Christensen *et al.*, 1996; Bush & Trexler, 2003; Spellerberg, 2005; Comin, 2010).

2.3 RESULTS (WAYS OF INTERPRETING THE DATA)

2.3.1. General concept

Assessing health in a complex system is seen as a difficult task entailing a good measure of judgment, precaution and modesty, but also much systems analysis and modeling so as to put all the individual pieces together into a coherent representation (Costanza & Mageau, 1999). By definition, all complex systems are made up of a number of interacting parts, components of which vary in their type, structure, and function within the whole system. How a system reacts cannot be calculated simply by adding up the behavior of the individual parts (Costanza, 1992). Furthermore, ecologists still do not yet have at their disposal a compendium of known diseases or stresses with associated symptoms and signs, as is the case with medical conditions (Rapport *et al.*, 2001a). However, some early attempts at measuring ecosystem health were based on agricultural models or climax community models which emphasized production and stability (Su *et al.*, 2010).

Evaluating environmental health by making use of the medical health concept has to undertake an upgrade in scale from the individual to that of an entire community. Furthermore, the size of a system is seen to be only marginally connected with the system's health, and consideration must be given to the fact that symptoms present on a small scale might not show on a larger scale, and vice versa (Rapport, 1995b). Although size may allow

a larger system to shift a smaller arrangement, it does not immediately follow that the larger is inevitably healthier than the smaller (Jordán, 2000). Furthermore, Mindell (2009) stipulated that on a temporal scale, a healthy ecosystem will slow the rate of evolution down over the longer time scale. In practice, the scientific principles for the preference of scales and development of practical assessments may only progress in those cases in which a consensus of public goals can be developed (Haskell *et al.*, 1992).

2.3.2 Identification of symptoms

Ecological integrity, one aspect of the triage concept (Samways, 2005), is the capacity of an ecosystem to support and maintain a balanced, integrated, adaptive community of organisms, having a species composition, diversity, and functional organization comparable to that of similar, undisturbed ecosystems in the region (Carignan & Villard, 2002). Karr (2006) also pointed out that ecological integrity implies an unimpaired condition of quality and a state of being complete or undivided. In short, it is the sum total of the chemical, physical and biological integrity, and lies on a scale between compositional, structural and functional biodiversity. Dynamic and ecological succession, as well as evolutionary development, follows a natural but not necessarily deterministic course, unless influenced by humans. Disturbance level, in comparison falls on a scale between intense and very infrequent, or mild at any time, to intense and frequent, as well as mostly anthropogenic (Samways, 2005).

2.3.3 Measurement of vital signs

Species richness is an estimate of the total number of species present in a defined physical area, as determined by some sampling method. In reality some species may be present but recorded, being absent at the time of sampling, or in a dormant stage or one that cannot be sampled. Furthermore, most sampling and monitoring methods are focused on specific taxa/ guilds/ life styles and will not give a realistic representation of all the species present at a given locality. Species accumulation curves are used to interpret, if enough samples or repetitions were taken to evaluate species richness, but this is never actually a finite number (Samways, 2005). It is further recommended to randomize samples to smooth out the species accumulation curve, as the order of input can drastically manipulate the shape of the curve and potentially indicate a false asymptote (Magurran, 2004). Making use of nonparametric estimators of total species richness will give both a measure of completeness of the inventory and also allow comparison with the species richness of other localities (Henderson, 2003).

As with species richness, it is important when evaluating relative abundance data, that it is only a representative of the population present in nature and more based on the effectiveness of the trapping or catching method towards certain species. It is also important to consider social species in an evaluation, as their numbers will be significantly higher than singleton species within the traps. Furthermore, predators are usually found in single or small numbers, while herbivores tend to be present in high numbers (Speight *et al.,* 2009).

Equitability or evenness is an essential component of the description of a community and is important in ecological monitoring. Highly stressed environments show low levels of equitability as the system becomes dominated by disturbances or pollution tolerant species (Henderson, 2003). Both the equitability and species richness may be summarized with a single value, a diversity index (as mentioned above) but such a summary may not always have desirable properties. For instance, the index will tend to be biased in favor of equitability or species richness and may simply reflect a change in one of these values which greatly exceeds the other. Diversity indices can be useful in comparing localities or for sampling the same locality through time but are of little use for comparison of different studies as they inevitably reflect the effort and techniques used by the researchers (Hederson, 2003). Nevertheless, diversity indices can be classified as dominance indices since being weighted towards abundance of the most common species, thus the addition of a few rare species with one individual will fail to change the index (Myers & Bazely, 2003). The Simpson-Yule and Burger-Parker indices are two such indices. The Simpson-Yule index (D) calculates a value in the range between 1 and S_{obs}, and the higher the value the greater the equitability (Henderson, 2003). While in the Berger-Parker index the reciprocal form is usually adopted, but its discriminating ability is less than the Simpson-Yule index (Southwood & Henderson, 2000). Newton (2007) mentions that the Berger-Parker index is one of the preferred diversity measures available, based partly on its results and partly on its ease of computation and because it is not that greatly influenced by the observed number of species.

Indices like the Shannon-Wiener function and Brillouin index are based on the underlying principle that diversity in a natural system can be calculated in a way that is similar to the information contained in a code or message. The analogy is that if the uncertainty of the next letter in a coded message can be known, then there is a potential to also know the uncertainty of the next species to be found in a community (Stilling, 1999). The Shannon-Wiener function tends to increase with the number of species in the sample so it often gives little more insight than species richness, and it is recommended to be plotted against the species richness to test for correlation (Henderson, 2003). Furthermore, the calculation assumes that all species are represented in the sample and are randomly sampled. The most common error

comes from the failure to include all species from the community in a sample, but this error decreases as the proportion of species represented in the sample increases and is minimal as it approaches the total actual number of species in the community (Good & Hardin, 2009). Values of the Shannon-Wiener function for real communities are often found to fall in the range between 1.5 and 3.5 (Fuller *et al.,* 2008).

Values generated by use of the Brillouin index are usually lower than those given by the Shannon-Wiener index for the same data set. One difference is that the Brillouin index does change, providing the number of species and their proportional abundance remain constant, while the Shannon-Wiener function does not (Stilling, 1999). The second variation in the two indices is that the use of factorials in the Brillouin equation quickly produces huge numbers that are computationally difficult (Lilburne & Tarantola, 2009).

In general, it is recommended to make use of more than one of these indices to give a more perceptive picture, but there are scenarios where the answers can be conflicting in rank (Tóthméresz, 1995). Henderson (2003) termed these samples as non-comparable. Such inconsistencies are an inevitable result of summarizing both relative abundance and species richness using a single value (Ricotta, 2003).

2.3.4 Provisional diagnosis

Interpretation of combined data available will lead to a decision on the course of action to be followed, and therefore it is important to take a few ecological community concepts into consideration when interpreting data before simply letting therapeutic nihilism take its course.

Khemakhem *et al*. (2010) described the concept of succession in ecology, a concept terminating in production of a 'climax community' which will perpetuate itself generation after generation until altered by some disturbance factor. In this concept, pioneer ecosystems colonizing a newly opened locality, progress through a roughly predictable series of states, concluding in what approximates a climax community, where after changes at the level of the entire community become insignificant (Jordán, 2000). An ecosystem with a trajectory toward the climax is relatively unimpeded and whose configuration is homeostatic is seen as healthy (Walker *et al*., 2007). However, in perturbation-dependent ecosystems, there are often many signs of ecosystem pathology, which mimic the natural signs of early succession behavior (Rapport & Ragier, 1992; Rapport, 1992). Odum (1969) views evolution as a form of succession at a very large scale and falls in with Holling's (1986) theory, that the destruction of the climax community by some surprise agent is one link in the necessary cycle of growth, destruction, and renewal, without which evolution could not occur (Müller *et al*.,

2010). It is thus important to distinguish between natural disturbances and human inflicted disturbances.

Eutrophication is an example of either a natural disturbance or human inflicted disturbance, whereby the biotope is enrichment with nutrients, primarily in the form of phosphorus and nitrogen. This usually leads to enhanced growth within a system. These changes may occur as a result of anthropogenic changes, or as a result of succession (Conley *et al*., 2009). Naturally eutrophic systems have developed higher diversity and organization along with higher metabolism and are therefore healthier, while artificially eutrophic systems tend toward lower species diversity, shorter food chains, and lower resilience (Rapport *et al*., 2001b). In general, eutrophication not only reduces species diversity but also favors increases in the abundance of certain species, particularly those with high tolerance for nearly anoxic conditions (Jørgensen *et al*., 2010).

A community can be considered to be stable when no change can be detected in the population sizes and number of species over a given time period (Beeby & Brennan, 2008). Decocq (2006) stated that there are a few multiple stable states within natural circumstances. Holling (1986) theorized, and then, De Lafontaine (2011) found evidence, that assemblages occupying an area before and after a disturbance are interpreted as alternative stable states when in fact their physical variables at the site have been changed. Furthermore, in most studies, data are not taken for more than a generation or a year, so to establish long term stability either before or after a disturbance is not possible (Li, 2004). This implies that a disturbance can alter a biotope, so that a totally different assemblage of species present can form an alternative equilibrium thereafter. Human influence through artificial manipulation, for example breeding programs, may also indicate multiple equilibria that differ from what the original equilibrium would have been.

Lévêque (2003) reviewed the theoretical mechanisms that have been used to explain the existence of equilibrium and non-equilibrium communities, and summarized three equilibrium theories of community organization: classical competition theory, predation extensions, and spatial variation extension. Furthermore, Lévêque (2003) summarized four non-equilibrium theories: fluctuating environment, density independence, changing environmental mean, and slow competitive displacement.

In classical competition theory, the process controlling community structure is competition where limited resources are essential for coexistence of all species (Pickett *et al*., 2007). Environmental influences are unimportant, and population growth rates can be established by deterministic equations. The environment is seen as spatially homogeneous, and there is little dispersal of organisms. The predation extension adds predation to the classical competition

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theory, whereby a new equilibrium model could be developed that allows all species to coexist on fewer resources than needed in any other system (Pickett *et al*., 2007). Whereas in the spatial variation extension, species compete for a single resource but the environment favors different species in different areas, thus it is possible for all species to coexist in a system of variant patches. This means that each area has a stable equilibrium (Picket *et al*., 2007), and the resulting model is similar to a metapopulation model (Kritzer & Sale, 2006).

Of the non-equilibrium theories, the fluctuating environment theory states that the environment changes seasonally or irregularly so that competitive rankings of species change temporally too, and no one species can establish dominance (Lahav, 2004). In the density independence theory, the environmental transformations and populations densities are not always high enough for competition or for other biotic processes, such as significant predation or parasitism (Price, 2003). Instead of an environment fluctuating around a mean, the mean actually changes, as it might if there were global climate change in the changing environmental mean theory. The slow competitive displacement theory assumes that competitive abilities are similar between species, and although competition is important, random variation and chance play a large part in determining the dominant species (Krebs, 2008).

From these theories, we can establish that communities are in constant flux tending to an equilibrium that most likely will never be reached. It is therefore extremely difficult to determine a normal state or health level for communities whose parameters are often in a condition of flux because of natural disturbances (Hinojosa *et al*., 2010). These disturbances transpire both on a spatial and on a temporal scale. Temporal disturbance exists on time scales measured in hours, days, months, years and millennia, depending, respectively, on the agents of disturbance (Jørgensen *et al.,* 2010). The fact is that in reality we generally do not know the normal range for any environmental variable, at least not for any time period greater than a few years. Schindeler (1987) stated that even with the development of a well-designed monitoring program, it would be years before we could confidently distinguish between natural variation and low-level effects of perturbations on a given ecosystem. Two decades later, this statement is still true, indicating no or very little progress has been made to develop a monitoring system (Longhurst, 2010). According to Holling (1986), most ecosystems might be described by arrhythmia, explaining the scenario of flux as dynamics that are highly irregular and punctuated by stochastic events.

These disturbances can also offset the system completely, resulting in an irreversible process of system decline leading to total destruction. This concept is known as the Ecosystem Distress Syndrome, characterized by reduced primary productivity, loss of nutrient

capital, loss of species diversity, dominance by short-lived, opportunistic, and often exotic species, increased fluctuations in key populations, retrogression in biotic structure and increased incidence of disease (Jørgensen *et al.*, 2010).

2.3.5 Verification tests

Ulanowicz (2004) pointed out that there should be a correlation between health and network ascendancy, but the association is not exact, or at least not fully understood. Even if the level of system activity and the degree of trophic organization relate to different concepts, the increase in network ascendancy can generally be attributed to the influence of a unitary mechanism in the form of autocatalysis, or indirect mutualism. This means that growth usually does not take place without development, and vice versa. Ulanowicz (2004) suggested consideration of autocatalysis as a mechanism, because it shows evidence of properties that cannot be linked to the separate actions of its components.

Odenbaugh (2005) characterized an ecosystem as stable if, and only if, the variables all return to the initial equilibrium after being disturbed. Thus, stability is evaluated relative to the original point (if known), which similar to the concept of integrity in the triage concept. Callicott (1992) stated that stability or health was related or caused by the system's diversity and complexity, but emphasized a causal relation would imply that the mechanisms are understood. Callicott (2000) further emphasized that the circumstantial evidence of the stability and diversity in the world's present communities were associated for approximately 20 000 years of coexistence without any major disturbances. However, both stability and diversity are now partly lost and greatly altered by human activity. This illustrates that the greater the losses and alterations, the greater the risk of impairment and disorganization of a system. However, 'stability' is not the stability implied in the Clementsian view of the community (Spieles, 2010), rather it is compatible with the prevalent recognition that biological systems can be considered meta-stable at best (Sit'ko, 2007). All non-successional or climax ecosystems are sustainable, but not necessarily stable (Wright, 2007).

2.3.6 Prognosis and prescription of treatment plan

The use of predictive numerical models or probabilistic models must either be based on an accurate knowledge of the structure and function of an ecosystem in its original state before the restoration attempt, or it must be formulated on a general analysis of the structure and function of the desired ecosystem type (Suding *et al.*, 2004). It must furthermore be reminiscent that the future contains ample unknown factors which might offset the resulting predictions.

2.3.7 Execution phase

Restoration is defined by the recreation of both the structural and functional attributes of a damaged ecosystem, and so it is a procedure and not a data collecting phase (Newton 2007). However, Westman (1991) discussed probable ecological criteria for evaluation of the success of restoration by means of a common inherent objective of ecological restoration to restore a self-sustaining, homeostatic system whose component species have a long co-evolutionary history (Troop, 2000). Furthermore, Troop (2000) points out that function and structure cannot always be restored simultaneously. He suggests evaluating structure by comparing species composition with the original species composition or with regional baseline levels, while measuring function by using probabilistic models to determine the likelihood of transition from one state to the next and monitoring the resilience of the ecosystem. A system is healthy if it can recover easily from perturbations and show minimal deviation from the natural succession processes (Hobbs & Suding, 2009).

Currently, restoration efforts are evaluated using community structure or ecosystem function models that favor biotic diversity, maintenance of natural processes, presence of rare or unique species, as well as system complexity. Furthermore, naturalness, as verified by the self-maintenance of the system or by the re-establishment of species present before disturbance, is also commonly used as an indicator of system health (Suding *et al*., 2004). Koehler (2005) pointed out that most models are inadequately equipped to deal with semi-natural landscapes and highly fragmented biotic communities, as well as those communities which have lost keystone species and those which cannot be completely restored.

The main hypothesis underlying the coarse filter approach is that the status of any given species is correlated with habitat availability. Nevertheless, some sites selected for conservation might be demographic sinks as presence of a species at a given site does not necessarily correlate with reproductive success and probability of persistence (Carignan & Villard, 2002). Coarse-filter approaches can be used to correctly identify the habitat features and processes required for a set of species to be present in a landscape, but this information might be inadequate for establishing the quantity and configuration of those features (Caringnan & Villard, 2002).

The fine-filter approach, has been shown to be problematic in its efficiency regarding protection of non-target species (Meffe, 2002), but indicator species, such as large vertebrates, might signify good indicators for species that require large, continuous areas of habitat (Nemésio & Silveira, 2010). These species might be unsuccessful for protecting species such as certain invertebrates that fare better in a naturally fragmented landscape (Carignan &

Villard, 2002). It is therefore recommended that components of both fine-filter and coarse-filter approaches should be integrated in conservation planning (Carignan & Villard, 2002; Noon *et al*., 2009, Samways *et al*., 2010).

2.3.8 Management phase

It is assumed that the management procedure is being implemented effectively when it results in preservation of the integrity, stability and beauty of the biotic community (Leopold, 1949; Priest & Gass, 2005). In turn a management procedure is failing when it tends to do otherwise (Karr, 1996). Leopold fundamentally established an ethic for management where consideration must be given to both the value and function of all parts of an ecosystem (Luthardt, 2010). This model has been influential in modern restoration, where productivity is less important than other values, such as naturalness and recovery of biotic diversity (Suding *et al*., 2004). The maintenance of land health as a result is not inevitably the same as maintenance of existing community structures with their historical complement of species (Callicott, 2000).

Carignana and Villard (2002) identified the five most often cited goals for ecosystem management to be: maintaining viable populations of all native species, protecting representative examples of all native ecosystem types across their natural range of variation, maintaining evolutionary and ecological processes to manage landscapes and species, responsiveness to both short-term and long-term environmental change, and accommodation of human activities within these constraints. In national land management, there has been a shift from managing for optimization of the most productive, consumable or remarkable elements of the system toward managing for protection of the most limited, unique or fragile elements (Suding *et al*., 2004). Furthermore, some believe that in contrast to the manipulative and productivity-oriented mode of ecosystem management, the best way to restore native biotic communities is to keep people and their activities out of the system (Hobbs & Suding, 2009).

2.4 DISCUSSION (SO WHAT DOES THIS ALL MEAN?)

The need to define the concept of ecosystem health arises from the failure of current economic paradigms to protect the natural environment. Nevertheless, ecosystem health cannot be defined or understood purely in biological, ethical, aesthetic or historical terms (Jørgensen *et al*., 2010). Health is a notion that is difficult to define, yet can be considered a normative concept (Ditlevsen & Friis-Hansen, 2007), better defined from what it is not, namely the occurrence of disease, trauma, or dysfunction (Ulanowicz, 1992). A healthy

ecosystem signifies a desired endpoint of environmental management, but the concept has further been difficult to use because of the complex, hierarchical nature of ecosystems (Lu & Li, 2003). It is for this reason that I, like others before, look to the analogy with the medical concept of health to better express and understand ecosystem health (Costanza & Mageau, 1999).

Correlation with the health metaphor can be drawn from the fact that human individuals and ecosystems are both complex systems composed of interacting parts in a complex balance of interdependent function (Costanza, 1992; Ambrosino, *et al.*, 2010). Although the difference is that humans are warm blooded, homeostatic systems and for the concept of human health, a compendium of known diseases, a wide body of reference data on the standard human and many diagnostic tools exists (Rodrick & Karwowski, 2006). In comparison, none of these aids is available for the ecosystem health concept, and therefore it is necessary to use ecosystem stability and resilience as indicators instead (Costanza, 2010).

Costanza (2010) indicated that an adequate definition of ecosystem health should be a collective evaluation of system resilience, balance, organization (diversity), and vigor (metabolism). Furthermore, the definition should be a comprehensive portrayal of the ecosystem. Costanza (1992), as well as Jørgensen (2010), suggested that the definition requires the use of weighting factors to contrast and aggregate different components in the ecosystem. These weights must be linked to the functional dependence of the system's sustainability on its components, and further should be able to fluctuate as the system changes to account for balance. Lastly, the definition should be hierarchical to account for the interdependence of diverse temporal and spatial scales (Rapport *et al.*, 2001b).

Leopold (1941) was the first to define land (ecosystem) health as nature's capacity for self-renewal, while Callicott (1992) replaced 'self renewal' with expressions such as 'self-maintenance' and 'self-regeneration'. Nonetheless, it was Karr (1992) who gave a more relative definition: that a biological system, whether individual or ecological, can be considered healthy when its inherent potential is realised, its condition is stable, its capacity for self-repair when disturbed, is preserved, and minimal external support for management is needed.

Norton (1991a), as well as Matlock & Morgan (2011), has suggested five axioms of ecological management to characterise ecosystem health:

1. The Axiom of Dynamism: Nature is more profoundly a set of processes than a collection of objects; all is in flux. Ecosystems develop and age over time.

2. The Axiom of Relatedness: All processes are related to all other processes.

3. The Axiom of Hierarchy: Processes are not related equally but unfold in systems within systems, which differ mainly regarding the temporal and spatial scale on which they are organized.

4. The Axiom of Creativity: The autonomous processes of nature are creative and represent the basis for all biologically based productivity. The vehicle of that creativity is energy flowing through systems which in turn finds stable contexts in larger systems, which provide sufficient stability to allow self-organization within them through repetition and duplication.

5. The Axiom of Differential Fragility: Ecological systems, which the context of all human activities, vary in the extent to which they can absorb and equilibrate human-caused disruptions in their autonomous processes.

It is Costanza's (1992) definition: 'An ecosystem being healthy and distress syndrome free when the ecosystem is stable and sustainable, as long as the ecosystem is active and preserves its organization and self-sufficiency over a temporal period as well as being resilient to stress', that stands out as most conclusive. Ecosystem health is thus closely associated with sustainability, which is seen to be an all-inclusive, multi-scaled, dynamic measure of system resilience, organization and vigor. This definition is valid for all complex systems from cells to ecosystems and allows for the fact that systems may be expanding and developing as a result of natural succession (Costanza, 1992). Furthermore, a diseased system is one which is not sustainable and will eventually cease to exist, and that individual organisms are not sustainable indefinitely. However, the populations and ecosystems of which they are part may be sustainable indefinitely in a healthy system (Rapport *et al.,* 2001b). The distress syndrome, mentioned above, is seen as the irreversible process of system breakdown leading to collapse (Chatalos, 2006).

Taking into consideration the definitions above, Costanza (1992) summarized health according to five specific viewpoints: health as homeostasis, health as the lack of disease, health as diversity or complexity, health as stability and resilience, and health as vigor and a scope of growth. The simplest and most popular view is that health is homeostasis, whereby any and all changes in the system represent a decrease in health, also defined as the maintenance of a steady state in living organisms by use of feedback control processes (Damiani, 2002). This perspective works reasonably well for organisms, especially warm-blooded vertebrates, since they are homeostatic and there is a large population from which to determine a normality or mean, but for ecosystems, with small populations, it works poorly or not at all (Rapport *et al*., 2001b). Homeostasis is thus an idealistic goal and restoration efforts should assume some alterations in species composition and abiotic factors over a temporal

scale (Bratton, 1992). If circumstances change adequately, there will be changes in the characteristics of an ecosystem, thus creating a 'new' ecosystem. This signifies fundamental change for the original ecosystem, and unless one can evaluate the relative value of the ecosystem before and after succession, then all forms of change must be considered harmful (Pyšek & Richardson, 2010).

The viewpoint of health as a lack of disease, considers a diseased system as one which is not sustainable and will eventually cease to exist (Botzler & Armstrong, 1998). Disease can then be seen as a form of stress to the system, a perturbation with certain negative effects. It is possible to catalogue and eliminate all anthropogenic stresses on an ecosystem, but without an independent definition of health, it is impossible to know if there is a single stress factor that caused each specific problem and to what degree (Rapport *et al*., 2001b). There are rather a multitude of stressors, many of them interactive, which can result in ecosystem degradation. This is especially the case in locations where moderate to intense human settlement or industrial activity are present (Zaccarelli *et al*., 2010). Under these extreme stress scenarios, a one way degradation process becomes inevitable and the ecosystem distress syndrome commonly occurs (Kasperson & Dow, 2005). Due to such inconsistency, different ecosystems often exhibit different aetiologies with respect to the same stress factors (Kasperson & Dow, 2005), and in certain scenarios, ecosystems which might exhibit much in the way of erratic behaviour in the absence of stress may also be resistant to change in the presence of stress (Schindler *et al*., 1985). Stress testing ecosystems is not a new concept. There have been many studies of recuperation times for ecosystems under natural stress (Rapport & Whitford, 1999; Laio *et al*., 2001, Wittebolle *et al*., 2009), as well as induced stress (Gray, 1989; Ditchkoff *et al*., 2006; Le Bagousse-Pinguet *et al*., 2011). As with human health, it is as inadequate to conclude that the health of a natural system by exposure to stress, and disease, for it is not only exposure, but also the natural resistance or vulnerability of the individual or ecosystem which determines the state of health (Jørgensen, 2010). Ecosystem health may further relate more to the ability of systems to use stress to their advantage, rather than to their ability to resist it completely (Rapport & Whitford, 1999).

A third possible view of ecosystem health is linked to an ecosystem's diversity or complexity as indicators of stability or resilience. However because diversity is relatively easy to measure in ecosystems, it has become a primary indicator of health (Brown & Ulgiati, 2010). Stability, and the related concept of resilience, refers to a healthy organisms' ability to withstand disease. A healthy organism is resilient and recovers quickly after a perturbation. Thus, this view implies that an ecosystem is able to preserve its structure and patterns of behaviour in the face of disturbance (Leslie & Kinzig, 2009). An ecosystem is healthy if it

can absorb stress and use it creatively rather than simply resisting it and maintaining its original configurations (Costanza & Mageau, 1999).   A problem with this viewpoint is that it excludes a system's operating level or degree of organization. A system that is dead is more stable than a live system because it is more resistant to change, but this does not make it healthier (Costanza, 1992).

Odum (1969) hypothesized that a system's ability to recover from stress is related to its overall metabolism, or energy flow, or to its capacity for growth (Costanza & Mageau, 1999), a concept that Costanza (1992) explained further where he referred to ecosystem health as a form of vigor, or activity-weighted resilience. Another view of health is that a healthy system is one that maintains the proper balance between system components, but this is a difficult prospect to monitor or evaluate, due to the lack of indicators to compare the balance, as well as due to the fact that each scenario fluctuates (Costanza & Mageau, 1999).

In the final analysis, it seems that 'health' is a concept that goes beyond scientific definition, and although it incorporates aspects that are not amenable to scientific methods of exploration, they are no less important or necessary (Rapport *et al.*, 2003). Norton (1991b) indicated that although the ecosystem health paradigm uses a broad medical model, the correlation between medicine and environmental protection does not always hold true (Rapport *et al*., 2001a). Another aspect to take into consideration is that an ecosystem has numerous functions and processes, not all of them strongly related to each other (Jørgensen, 2010). Additionally, a universal word such as 'health' can end up with all kinds of narrative definitions, and can lose some of its original meaning if applied too rigorously to examples of specific communities. This is due to the fact that communities vary greatly, ranging from a hypothetical state of equilibrium to the furthest point at the non-equilibrium end.   The cohesive concept of health can vanish if redefined for each scenario (Ehrenfeld, 1992).

Ecosystem health is a bridging notion that connects two ideas and provides a concept that can be helpful in communicating with nonscientists and as a tool to conceptualize the quality state of an ecosystem (Rapport *et al*., 2003). It also answers the plea of scientists like Karr (1992; 1996; 2006) and Norton (1991a; 1991b; 1992; 2009) who highlighted the need to find creative ways to assess the condition of the diverse dimensions of life support systems from local to global scales. For this reason, many dimensions of ecological systems are appreciated by society, and therefore, the key component of an attempt to evaluate an 'ecosystem's health' and must be a pluralistic foundation in ecological principles and theories (Mugerauer & Manzo, 2008). Miller (2000) also argues for the use of 'ecosystem health' because it provides a value that indicates the quality of life, which exemplifies non-human and even non-conscious systems independently of human utilities.

2.5 CONCLUSION

Society's conceptualization of the consequences of biotic impoverishment, and of the extent to which the human species depends on healthy biological systems, is rudimentary at best (Hilgenkamp, 2005). Therefore, it is essential that this generation implements the basic formulation for conservation, restoration, monitoring and management to optimize the period of current status between nature and the human species that favour humanity. A good starting point is to conceptualize the notion of environmental health and from this foundation, implementation can begin to take place. In conclusion I thus take into consideration all of above mentioned definitions to define ecosystem health as: "An ecosystem is healthy if it can sustain an optimal number of species with optimal population sizes and their ecological processes, thus providing an optimal heterogeneous sustainable system with sufficient resources, and indicate adequate resistance when under perturbational stress, but still allowing natural succession to take place."

It is now for ecologists to further develop a more meticulous taxonomy of ecosystem perturbations or ills, which can provide a basis for calculating statistics on the success of various efforts at rehabilitation (Jørgensen *et al*., 2010). Humanity's ability to conserve ecological health will be determined by our ability to identify the roots of the biodiversity crisis, the proliferation of disease and many forms of human injustice (Boyd, 2010). It is therefore, that we as human beings for the sake of our future, need to redefine Gifford Pinchot's consumption-oriented resource conservation ethic, which calls for harvesting our surroundings, to provide the greatest good for the greatest number of people for the longest time (Meffe, 2000), into providing the greatest good for the greatest number of species, for the longest time.

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Chapter 3

LITERATURE REVIEW ON ARTHROPODA AS BIOINDICATORS

ABSTRACT

Bioindicators are a species (or group of species) that readily: 1) reflects the abiotic or biotic state of an environment, 2) represents the impact of environmental change on a habitat, community or ecosystem, 3) or is indicative of the diversity of a subset of taxa or of wholesale diversity within a biotope. Arthropoda fit these requirements as they are small and usually highly sensitive to higly local conditions, while most, being mobile, are reactive to changing environmental conditions. Arthropods have generally fast breeding rates and short generation times, and are often highly responsive numerically to environmental changes. They show a diversity of growth rates, life history styles, body sizes, food preferences, and ecological preferences, and can, at the species level, often be linked to specific environmental variables.   Ephemeroptera, Plecoptera, Trichoptera and Diptera are useful taxa as bioindicators of the aquatic environment, while in the marine environment Crustacea, especially the orders Isopoda, Amphipoda and Decapoda, are the most useful as bioindicators. Odonata make good indicators of the riparian environment.In the terrestrial environment, Araneae in the predator guild, Orthoptera in the herbivore guild, Collembola in the decomposer guild, Nymphalidae in the pollinator guild, and Formicidae as scavengers, make good bioindicators. Using grouped data on the whole order of Coleoptera, makes for good bioindicator species due to their great diversity and wide geographical distribution.

3.1   INTRODUCTION

McGeoch (2007) defines bioindicators as those species (or groups of species) that: reflect the abiotic or biotic state of an environment; represents the impact of environmental change on a habitat community or ecosystem; or is indicative of the diversity of a subset of taxa or of wholesale diversity within a biotope. This definition is similar to that in the Oxford Dictionary of Zoology (Allaby, 1992) where an indicator species is a species that is of narrow amplitude with respect to one or more environmental factors and that is, when present, therefore indicative of a particular environmental condition or set of conditions. The original use of the term "biological indicator" was in aquatic systems, where it refers to the detection and monitoring of change in biota to reflect changes in the environment (Wilhm & Dorris, 1968).

Making use of bioindicators over time is known as biomonitoring, which is the repeated application of bioindicator taxa to provide information on the environmental conditions, or effects thereof, to which they were initially identified as correlating, sensitive and for which baseline standards, thresholds or relationships have already been determined (McGeoch, 2007). Hellawell (1991) makes the following distinctions between monitoring and other related activities. Monitoring is the intermittent surveillance carried out in order to ascertain the extent of compliance with a predetermined standard or the degree of deviation from an expected norm. In contrast, a survey is seen as an exercise in which a set of qualitative or quantitative observations are made, usually by means of standardized procedure and within a restricted period of time, but without any preconception of what the findings ought to be. In turn, surveillance is an extended programme of surveys undertaken to provide a time series, to ascertain the variability and or range of states or values which might be encountered over time (Hellawell, 1991).

Jenkins (1971), and later Spellerberg (1991) divided bioindicators into categories. They defined sentinels as sensitive organisms introduced into the environment, e.g. as early-warning devices or to restrict the effect of an effluent (Gerhardt, 1996). Detectors are species occurring naturally in the study area and which may show a measurable response to environmental change, e.g. changes in behaviour, mortality and age class structure (Nash *et al.,* 1998). Exploiters are species whose presence indicates the probability of disturbance or pollution, e.g. the Argentine ant (Roura-Pascual *et al.,* 2008).

Accumulators are organisms which take up and accumulate chemicals in measureable quantities, e.g. heavy metals in Carabidae (Butovsky, 2011). Bioassay organisms are selected organisms used as laboratory reagents to detect the presence and/or concentration of pollutants, or to rank pollutants in order of toxicity, e.g. in sediment pollution (De Lange, *et al*., 2004).

McGeoch, (2007) then re-divided bioindicators into three types, environmental, ecological and biodiversity indicators, but emphasised the distinction of bioindicators from abiotic indicators, such as soil quality, temperature and landscape structure. An environmental indicator is a species or group of species that responds predictably, in ways that are readily observed and quantified to environmental disturbance or to a change in environmental state (McGeoch, 2007). An ecological indicator is a species or a group of species that demonstrate the effects of environmental change (such as habitat alteration, fragmentation and climate change) on biota or biotic systems. These species represent the response of at least a subset of other organisms to such stresses (McGeoch, 2007). A biodiversity indicator is a group of taxa or functional group, the diversity of which reflects some measure of the diversity (e.g. character richness, species richness, level of endemism) of other higher taxa in a habitat or set of habitats (McGeoch, 2007). Biodiversity can thus be evaluated at a number of levels of organisation including genetic, species or ecosystems levels (Noss, 1990).

Hammond (1994) also used the term 'biodiversity indicators', and divided them according to their application to particular types of extrapolations from which ratio-based extrapolations can be made. For example, the biodiversity indicator may be a reference, key, focal or target group. A reference group is used as a basis for extrapolation to a group for which incomplete data are available, while a key group's principal role is to provide a focus for efforts made to document and estimate species richness in a given arena. A focal group, performs a reference role but represents a subset of a larger group of interest selected specifically for its qualities as a predictor set, similar to what Kitching (1996) called a target group, but different from Hammond's target group in that they are merely a group that is under investigation or the object of attention.

Invertebrates have long been flagged as capable bioindicators, with Bisvac and Majer (1999) suggesting that trends in species richness and community composition are better represented by certain invertebrates than by vertebrates, and that invertebrates can be highly

cost-effective indicators. Kremen *et al*. (1993) even went so far as to suggest that terrestrial arthropods could be used for virtually any monitoring programme, so long as the goals are well defined.

Invertebrates in general have some attributes which lend themselves to biomonitoring and biodiversity assessments (Samways *et al.,* 2010). They are small and therefore highly sensitive to local conditions, most being mobile and reactive to changing conditions, they can thus respond by moving away from, or moving towards, adverse or optimal conditions respectively. Invertebrates have short breeding rates and short generation times, and therefore often highly responsive numerically to changes. Generally they show a diversity of growth rates, life history styles, body sizes, food preferences, and ecological preferences, and at the species level can often be linked to specific environmental variables. The ability of their abundance to fluctuate provides an extra sensitivity layer, over that of species richness, for indicating subtle changes in environmental conditions. Furthermore, most invertebrates are relative easy and safe to survey.

Similar studies that attempted to identify bioindicators or put in place a procedure for the use of invertebrate bioindicators include Lenhard & Witter (1977), Majer (1983), Holloway & Stork (1991), Rosenberg & Resh (1993), Chessman (1995), Luff & Woiwod (1995), McGeoch (1998), Caro & O'Doherty (1999), Hilty & Merenlender (2000), Lu & Samways (2002), Duelli & Obrist (2003), Kati *et al*. (2004), Niemi & McDonald (2004), Ruano *et al*. (2004), Balvanera *et al.* (2005), McGeoch (2007), Lovell *et al*. (2007), Uehara-Prado *et al*. (2009), Aydin & Kazak (2010), Finch & Loffler (2010), Heink & Kowarik (2010), and Vandewalle *et al*. (2010).

3.1.1 *Aims of this chapter*

The aim here is to review the literature (especially over the last decade) on the use of various arthropod taxa as bioindicators, and to summarize in which way they can be used as bioindicators. As well as to identify taxa to represent the various guilds in later analysis of the Biotope Quality Index, being investigated in this thesis.

3.2   APPROACH

Literature was gathered using various search methods (Web of Science; Google Scolar; Academic Publications eJournal; BioOne; World Wide Science) on the basis that the focal studies were about testing, or the practical use, of any arthropod group as a bioindicator.

Due to the many positive results, focus was placed on the last decade (2000-2010), and the way that the field is evolving.

3.3   RESULTS

3.3.1   *Crustacea as bioindicators*

Crustaceans, which are most abundant in the marine environment, show great potential as indicators.   In the class Malacostraca (superorder Peracarida), the order Amphipoda shows most potential, and has been used as indicators of water quality to habitat quality, to detect water and sediment pollution, to determine levels of disturbances, as well as in environmental management success (Table 3.3.1.1a). The order Mysidacea, Cumacea and Isopoda have also been used as indicators (Table 3.3.1.1b).

In the order Decapoda (Eucarida, Malacostraca), the infraorder Brachyura has been the most often used indicator group. Brachyura have been used as indicators in various environmental status evaluations, water pollution studies as well as to determine level of disturbance, e.g., temperature stress, temperature change, radiation levels and changes in salinity (Table 3.3.1.2). Infraorder Anomura has also been used as an indicator in multi-taxa studies to indicate the presence and level of infestation of an intruding underwater pest plant, *Caulerpa taxifolia,* along costal areas (Francour *et al*., 2009).   I found no reference to the superorders Syncarida and Hoplocarida (class Malacostraca) being used as bioindicators. Other crustacean classes that show potential as bioindicators are Brachipoda, Copepoda, Cirripedia and Ostracoda, as habitat and water quality indicators, as well as for indicating water pollution (Table 3.3.1.3). The classes Remipedia, Cephalocarida, Mystacocarida, Branchiura and Tantulocarida have not yet been shown to be of value as bioindicators.

3.3.2   *Chelicerata as bioindicators*

None of the aquatic Chelicerates (Pycnogonida & Merostomata) have been used as bioindicators, but in the terrestrial environment species in the class Arachnida have shown great potential (Table 3.3.2 a & b).   Orders Araneae and Acari had been used in many applications, from environmental status, pollution to disturbance level studies, as well as in both environmental and agricultural management practices. The order Pseudoscorpiones has been used in evaluation of soil quality (Barros *et al*., 2010), while orders Opiliones and Palpigradi has been used to monitor post fire recovery (Broza *et al*., 1993; Pryke & Samways,

Table 3.3.1a Literature review of superorder Peracarida (Malacostraca; Crustaceae) as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Amphipoda	Environmental status	
	• Water quality	Muenz *et al.* (2006); Kunz *et al.*(2010)
	• Arctic environment	Ikko & Lyubine (2010)
	• Afrotemperate forests	Kotze & Lawes (2008)
	• Afromontane forest	Lawes *et al.* (2005)
	• River conditions	Yildiz *et al.* (2010)
	Water pollution levels	Beach & Pascoe (1998)
	• Heavy metal accumulation	Borgmann *et al.*(1993); Krupa & Guidolin (2003); Shuhaimi-Othman & Pascoe (2007)
	▪ Cadmium	Issartel *et al.*(2010)
	• Trace metal accumulation	Ungherese *et al.* (2010)
	• Toxicity levels	Anderson *et al.*(2008)
	• Alpha- & betha-endosylfan	Wan *et al.* (2005)
	• Tributyltin	Bartlett *et al.*(2004)
	• Pentachlorophenol	Graney & Giesy (1986)
	• Pesticides	Cothran *et al.* (2010)
	• Effluence levels	Maltby & Crane (1994)
	Sediment pollution levels	Obermuller *et al.* (2007)
	• Radioactivity	Alves *et al.* (2008)
	Level of disturbance	
	• Urbanization	Veloso *et al.* (2010)
	• Oil spills	Nikitik & Robinson (2003)
	• Mine tailing	Burd (2002)
	• Seagrass removal	Carvalho *et al.* (2006)
	Environmental management success	
	• River restoration	Arvai *et al.* (2002)
	• Sewage outfall restoration	Dedecker *et al.* (2006)

Table 3.3.1.1b Literature review of superorder Peracarida (Malacostraca; Crustaceae) as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Mysidacea	Physiochemical characteristics of water	
	• Water mass	Xu & Shen (2007)
	Sediment pollution levels	Chung *et al.*(2007)
	Level of disturbance	
	• Invasive alien plant *Caulerpa taxifolia*	Francour *et al.*(2009)
Cumacea	Water pollution levels	
	• Heavy metal accumulation	
	▪ Copper, Cadmium, Lead and Zinc	Swaileh & Adelung (1995)
Isopoda	Environmental status	
	• Food quality	Zidar *et al*. (2003)
	Water pollution levels	
	• Toxicity levels	Van Straalen *et al*. (2005)
	• Heavy metal accumulation	DeNicola *et al*. (1996); Akyuz *et al*. (2001); Nahmani & Rossi (2003)
	• Lead and Cadmuim	Blanusa *et al*. (2002)
	• Copper and Nickel	Alikhan (1993)
	Forensic entomology	Grassberger & Frank (2004)
	Enviromental management success	
	• Wet meadow restoration	Riggins *et al*. (2009)
	• Landscape restoration	Pryke & Samways (2009)

Table 3.3.1.2 Literature review of order Decapoda (Malacostraca; Crustaceae) as bioindicators and for what they are indicators

Infraorder	Bioindicator of	Reference
Anomura	Level of disturbance	
	• Invasive alien plant *Caulerpa taxifolia*	Francour *et al.* (2009)
Brachyura	Environmental status	
	• Habitat quality	
	• Tropical coastal marine environments	Van Oosterom *et al.* (2010)
	• Salt marshes	Morgan *et al.* (2006)
	• Estuarine health	George *et al.* (2001)
	• Water quality	Anderson *et al.* (2006); Stewart *et al.* (2010); Wong *et al.* (2010)
	Water pollution levels	
	• Toxins	Bretz *et al.* (2002)
	• Polycyclic aromatic hydrocarbons	Dissanayake & Bamber (2010)
	• Phycotoxin domoic acid	Ferdin *et al.* (2002)
	• Domoic acid	Powell *et al.* (2002)
	• Organochlorine pesticides	Eisenberg & Topping (1984)
	• Polychlorinated biphenyls	Cumbee *et al.* (2008)
	• Sodium nitrate, Potassium nitrate, Potassium chloride	Romano & Zeng (2007)
	• Heavy metal accumulation	
	• Iron, Manganese	MacFarlane (2000); Khoury *et al.* (2009)
	• Mercury	Sanders *et al.* (1998)
	• Effluents	Cumbee *et al.* (2008)
	Level of disturbance	Penha-Lopes *et al.* (2009)
	• Temperature stress	
	• Temperature change	Frederich *et al.* (2009)
	• Ultraviolet radiation	Webb *et al.* (2006)
	• Changes in salinity	de Oliveira *et al.* (2002)
		Chapelle & Zwingelsteing (1984)

Table 3.3.1.3 Literature review of class, Branchiopoda, Ostracoda, Copepoda and Cirripedia (Crustaceae) as bioindicators and for what they are indicators

Taxon	Bioindicator of	Reference
Branchiopoda (Class)	Environmental status	
	• Habitat quality	
	o Oligotrophic shallow ponds	De Los Rios *et al*. (2008)
Anostraca (Order)	• Water quality	Ruck (1998)
	Water pollution levels	
	• Toxins	
	• Peracetic acid	Antonelli *et al.* (2009)
	• Lead acetate	Kutlu *et al.*(2008)
Ostracoda (Class)	Water pollution levels	Bednarski & Morales-Ramirez (2004)
	• Heavy metal accumulation	Bergin *et al.* (2006); Martins *et al.* (2010)
	• Toxins	Martinez-Colon *et al.* (2009)
	Level of disturbance	
	• Tsunami	Hussain *et al.* (2010)
Copepoda (Class)	Environmental status	
	• Habitat quality	
	o Oligotrophic shallow ponds	Los Rios *et al*. (2008)
	• Water quality	Landa *et al.* (2007)
	Water pollution levels	
	• Heavy metal accumulation	
	o Copper	Gutierrez *et al*. (2010)
	o Nickel	Wang & Wang (2010)
	o Cadmium	Wang & Wang (2009)
Cirripedia (Class)	Environmental status	
	• Habitat quality	
	o Biomass of plankton resources	Achituv *et al.* (1997)
	Water pollution levels	Royogelabert & Yule (1994)
	• Effluents	Farrapeira *et al.* (2010)
	• Heavy metal accumulation	Rainbow & Phillips (1993); Ho *et al*. (2009)
	Level of disturbance	
	• Scouring by sea ice	Belt *et al*. (2009)
	• Shore line and water level fluctuation	Ho *et al.* (2009)

2008). No indication of use as bioindicators was found for orders, Solifugidae, Schizomida, Uropygi, Ricinulei and Amblypygi.

3.3.3 *Myriapoda as bioindicators*

Members of the order Diplopoda are used in environmental status, level of disturbance as well as environmental management studies as bioindicators, while order Chilopoda was mentioned in Kappes *et al.,* (2009), as an indicator of habitat health of temperate deciduous forests (Table 3.3.3).

3.3.4 *Apterygota as bioindicators*

In the subclass Apterygota, only the order Collembola has been used in bioindicator studies (Table 3.3.4a), and there is enough knowledge on their ecology and identification for some species of Collembola to be used at species level as bioindicators (Table 3.3.4b).

3.3.5   *Paleoptera as bioindicators*

The orders Ephemeroptera (Table 3.3.5a) and Odonata (Table 3.3.5b) are well known in the use as bioindicators of water quality as well as habitat quality, owing to their acute sensitivity to environmental change.

3.3.6 *Primitive exopterygota as bioindicators*

The order Orthoptera has been widely used in bioindication studies, usually in environmental status evaluations, to measure the level of disturbance and in environmental management (Table 3.3.6a). The orders Blattodea and Mantodea have only been used in studies to measure level of disturbance (Table 3.3.6a). It is the order Plecoptera (Table 3.3.6b) that shows the most potential as bioindicators, especially in water quality evaluations.

3.3.7 *Hemipteroids and Thysanoptera as bioindicators*

Hemipteroid species are mainly herbivores and have been only rarely used as bioindicators. However, there are a few examples where the predatory species have been used as bioindicators of water pollution levels and in environmental impact studies of mines (Table 3.3.7). The order Thysanoptera has been used in the management of olive orchards (Ruano *et al.,* 2004).

Table 3.3.2a Literature review of class Arachnida (Chelicerata) bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Araneae	Environmental status	
	• Biodiversity	Gollan *et al.* (2010)
	• Habitat structure	Buchholz (2010)
	• Microclimatic continentality	Rezac *et al.* (2007)
	• Soil compactness	Rezac *et al.* (2007)
	• Landscape and habitat features	Jeanneret *et al.* (2003)
	Environmental pollution levels	Seyyar *et al.* (2010)
	• Heavy metal pollution	Jung *et al.* (2008)
	Level of disturbance	
	• Urbanization	Magura *et al.* (2010)
	• Overgrazing	Horvath *et al.* (2009)
	• Rainforest fragmentation	Kapoor (2008)
	• Tillage	Rodriguez *et al.* (2006)
	• Mowing of wet meadows	Cattin *et al.* (2003)
	Environmental management success	Cardoso *et al.* (2004)
	• Habitat restoration	Gollan *et al.* (2010)
	• Conservation value for peat	Scott (2006)
	• Habitat transformation	
	o Arable land into grassland	Perner & Malt (2003)
	o Grassland management	Pozzi *et al.* (1998)
	Agricultural management success	
	• Push-pull habitat management	Midega *et al.*(2008)
	• Herbicide-tolerant testing	Haughton *et al.*(2003)

Table 3.3.2b Literature review of class Arachnida (Chelicerata) bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Acari	Environmental status	
	• Soil biodiversity	Gulvik (2007)
	• Physiochemical soil factors	Iturrondobeitia & Salona (1991)
	Environmental pollution levels	Eeva & Penttinen (2009)
	• Heavy metal pollution	Jung *et al.* (2008)
	• Air pollution	Steiner (1995)
	Level of disturbance	
	• Fire	Jung *et al.* (2010)
	• Prescribed fire	Camann *et al.* (2008)
	• Tilling and nitrogen fertilisation	Tabaglio *et al.* (2009)
	• Land use	Gulvik (2007)
	Environmental management success	
	• Ecosystem monitoring	O'Neill *et al.* (2010)
	• soil management	O'Neill *et al.* (2010)
	• Host age in wild boars	Foata *et al.* (2006)
	Agricultural management success	
	• Conventional management regimes	Yeates *et al.* (1997)
	• Organic management regimes	Yeates *et al.* (1997)
	Forensic Entomology	Grassberger & Frank (2004)
	• Animal and human decomposition	Perotti & Braig (2009)
Opiliones	Environmental management success	
	• Post fire recovery	Pryke & Samways (2008)
Pseudoscorpiones	Environmental status	
	• Soil quality	Barros *et al.* (2010)
Palpigradi	Environmental management success	
	• Post fire recovery	Broza *et al.* (1993)

Table 3.3.3 Literature review of superorder Myripoda as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Diplopoda	Environmental status	
	• Biodiversity	
	o Afrotemperate forest	Uys *et al.* (2010)
	• Habitat quality	
	o Temperate deciduous forest	Kappes *et al.* (2009)
	• Dispersal	Enghoff (1993)
	Level of disturbance	
	• Green-tree retention	Halaj *et al.* (2009)
	Environmental management success	
	• Ecological restoration	Snyder & Hendrix (2008)
Chilopoda	Environmental status	
	• Habitat quality	
	o Temperate deciduous forest	Kappes *et al.* (2009)

Table 3.3.4a Literature review of Apterygota as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Collembola	Environmental status	
	• Ecological status	Greenslade (2007)
	• Aquifer low-permeability	Sullivan *et al.* (2009)
	• Sustainability	King & Hutchinson (2007)
	• Habitat quality	
	o Restinga environments	Fernandes *et al.* (2009)
	Level of disturbance	Barbercheck *et al.* (2009)
	• Nitrogen deposition	Xu (2009a & b)
	• Liming and fertilization	Geissen & Kampichler (2004)
	• Experimental liming	Chagnon *et al.* (2001)
	• Tillage and nitrogen fertilisation	Tabaglio *et al.*(2009)
	• Green-tree retention	Halaj *et al.*(2009)
	• Forest conversion	Salamon *et al.* (2008)
	• Landscape stress	Greenslade (2007)
	Environmental pollution levels	Fiera (2009)
	• Herbicides	
	o Glyphosat, 2-4-D Atrazine e Nicosulguron	Lins *et al.* (2007)
	Physiochemical characteristics of soil	
	• Acidity	Van Straalen & Verhoef (1997)
	• Quality	Baretta *et al.* (2008); Barros *et al.* (2010)
	• Properties	Natal-da-Luz *et al.* (2008)
	Soil pollution levels	Souza & Fontanetti (2011)
	• Insecticides	Pramanik *et al.* (1998)
	• Hydrocarburans	Uribe-Hernandez *et al.* (2010)
	Environmental management success	
	• Ecosystem monitoring	O'Neill *et al.* (2010)
	• Soil management	O'Neill *et al.* (2010)
	• Forest restoration	Majer *et al.* (2007)
	Agricultural management success	Gardi *et al.* (2008)
	• Bentonite application	Szeder *et al.* (2008)

Table 3.3.4b Literature review of Collembola species used as bioindicators and for what they are indicators

Species	Bioindicator of	Reference
Aporrectodea caliginosa	Physiochemical characteristics of soil	
	• Salinity levels	Owojori *et al.* (2009)
Eisenia fetida	Physiochemical characteristics of soil	
	• Salinity levels	Owojori *et al.* (2009)
Enchytraeus doerjesi	Physiochemical characteristics of soil	
	• Salinity levels	Owojori *et al.* (2009)
Folsomia candida	Environmental pollution levels	
	• Toxins	
	○ Phenanthrene	Nota *et al.* (2009)
	Physiochemical characteristics of soil	
	• Salinity levels	Owojori *et al.* (2009)
	Environmental status	
	• Habitat quality	
	○ Beech forest ecosystem	Kopeszki (1992)
Heteromurus nitidus	Environmental status	
	• Habitat quality	
	○ Beech forest ecosystem	Kopeszki (1992)
Paronychiurus kimi	Environmental pollution levels	
	• Heavy metal accumulation	
	○ Cadmium, Mercury and Lead	Son *et al.*(2007)
Sinella curviseta	Environmental pollution levels	
	• Heavy metal accumulation	Xu *et al.* (2009b)

Table 3.3.5a Literature review of the subclass Paleoptera as bioindicators and for what they are indicators

Taxon	Bioindicator of	Reference
Ephemeroptera (Order)	Environmental status	
	• Biotic integrity	Bauernfeind & Moog (2000) Arimoro & Ikomi (2009) Masese *et al.* (2009a)
	• Ecological change	Spellerberg (2005)
	• Ecological quality	Garcia-Criado *et al.* (2005)
	• Environmental health	Gamboa *et al.* (2008)
	o Wetlands	Sharma & Rawat (2009)
	• Water quality	Azrina *et al.* (2006)
		Walsh (2006)
		Masese *et al.* (2009b)
		Paparisto *et al.* (2010)
	Physiochemical characteristics of water	
	• Eutrophication	Menetrey *et al*. (2008)
	Level of disturbance	
	• Timber harvest	Gravelle *et al.* (2009)
	• Impoundment	Bredenhand & Samways (2009)
	• Acid mine drainage	Gerhardt *et al.* (2004)
	• Nutrient enrichment in wetlands	Lemly & King (2000)
	• Effluents	Coimbra *et al.* (1996)
Baetidae (Family)	Environmental status	
	• Environmental degradation	Buss & Salles (2007)
Isonychiidae (Family)	Water pollution levels	
Isonychia bicolor	• Heavy metal accumulation	
	o Chronic mercury pollution	Snyder & Hendricks (1997)

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Table 3.3.5b Literature review of Paleoptera as bioindicators and what they are indicators

Taxon	Bioindicator of	Reference
Odonata (Order)	Environmental status	
	• Ecological integrity	Chovanec & Waringer (2001)
		Trigal *et al*. (2009)
		Silva *et al*. (2010)
	• Environmental health	Gamboa *et al*. (2008)
	• Environmental quality	Bulankova (1997)
	○ Arid tropical environments	Suhling *et al*. (2006)
	• Riparian quality	Smith *et al*. (2007)
	• Water quality	Azrina *et al*. (2006)
		Paparisto *et al.* (2010)
	Physiochemical characteristics of water	
	• Trophic state of ponds	Menetrey *et al.* (2005)
	Water pollution levels	Hardersen (2000)
	• Heavy metal accumulation	Nummelin *et al.* (2007)
	Level of disturbance	
	• Impoundment	Bredenhand & Samways (2009)
	• Grazing	Foote & Hornung (2005)
	Environmental management success	
	• Sewage lagoons	Catling (2005)
	• Riparian recovery	Samways & Sharratt (2010)
Pseudostigmatidae (Family)	Environmental management success	
	• Forest conversion	Fincke & Hedstrom (2008)
Platycnemididae (Family)	Physiochemical characteristics of water	
Copera annulata	• Temperature stress	Chang *et al.* (2007)
	Water pollution levels	
	• Pesticides	Chang *et al.* (2007)
Coenagrionidae (Family)		
Xanthocnemis zealandica	Water pollution levels	
	• Toxins	
	○ Carbaryl	Hardersen *et al*. (1999)
		Hardersen & Wratten (1998)

Table 3.3.6a Literature review of the superorder Orthopteroidea as bioindicators and for what they are indicators

Taxa	Bioindicator of	Reference
Orthoptera (Order)	Environmental status	
	• Biodiversity	Jana *et al.* (2006)
	• Ecological activity	Quinn *et al.* (1993)
	• Ecological change	
	○ Savanna	Samways *et al.* (2009)
	• Environmental health	
	○ Tropical savannas	Andersen *et al.* (2001)
	• Habitat quality	Bazelet & Samways (2011)
	Level of disturbance	
	• Human population size	Steck & Pautasso (2008)
	• Moisture levels	
	• Dieback disease	Newell (1997)
	• Grazing	Gibson *et al.* (1992)
	• Exotic conifer patches	Samways & Moore (1991)
	Environmental management success	
	• Land management	Jonas *et al.* (2002)
Gryllidae (Family)	Level of disturbance	
	• Mining emissions	Hoffmann *et al.* (2002)
Acrididae (Family)	Environmental status	
	• Disturbance level	
	○ Deciduous forest	Saha & Haldar (2009)
Melanoplus frigidus	Climate change	Finch *et al.* (2008)
Oxya chinensis	• Environmental pollution levels	
	• Heavy metal accumulation	
	○ Cadmium	Li *et al.* (2005)
Blattodea (Order)	Level of disturbance	
	• Dieback disease	Newell (1997)
Mantodea (Order)	Level of disturbance	
	• Moisture levels	Gavlas *et al.* (2007)

Table 3.3.6b Literature review of Orthopteroidea as bioindicators and for what they are indicators

Taxon	Bioindicator of	Reference
Plecoptera (Order)	Environmental status	
	• Climate change	Lawrence *et al.* (2010)
	• Ecological impairment	Petty *et al.* (2010)
	• Ecological integrity	Arimoro & Ikomi (2009)
	• Instream biotic integrity	Mattsson & Cooper (2006)
	• Integrity of eutrophic streams	Lecerf *et al.* (2006)
	• Ecological status	Sanchez-Montoya *et al.* (2010)
	• Ecosystem health	Young & Collier (2009)
	• Stream health	Walsh (2006)
		Collier (2008)
	• Water quality	Boonsoong *et al.* (2009)
		Hutchens *et al.* (2009)
		Masese *et al.* (2009b)
		Gabriels *et al.* (2010)
		Shiels (2010)
	Physiochemical characteristics of water	
	• Hydromorphological degradation	Korte (2010)
	• Stream condition	Moya *et al.* (2007)
	• Stream flow characteristics	Konrad *et al.* (2008)
	Water pollution levels	Imoobe & Ohiozebau (2010)
	• Heavy metal accumulation	Girgin *et al.* (2010)
	Level of disturbance	Azrina *et al.* (2006)
		Maloney & Feminella (2006)
		Waite *et al.* (2010)
	• Anthropogenic perturbations	Azrina *et al.* (2006)
		Kasangaki *et al.* (2006)
		Pecher *et al.* (2010)
	• Clear-cut logging	Banks *et al.* (2007)
	• Acid mine drainage	Ross *et al.* (2008)
	• Human population pressure	Pautasso & Fontaneto (2008)
	• Human activities	Krno *et al.* (2007)
	Environmental management success	
	• Stream types	Dohet *et al.* (2008)
	• Stream restoration	Selvakumar *et al.* (2010)
	• Stream buffer effectiveness	Muenz *et al.*(2006)
	• Wastewater treatment	Pinto *et al.* (2010)
	Agricultural management success	
	• Land use	Song *et al.* (2009)
Perlidae (Family)	Environmental status	
Anacroneuria (Genus)	• Water quality	Tomanova & Tedesco (2007)

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Table 3.3.7 Literature review of the superorder Hemipteroidea as bioindicators and for what they are indicators

Taxon	Bioindicator of	Reference
Hemiptera (Order)	Level of disturbance	
	• Acid mine drainage	Gerhardt *et al.* (2004)
	• Terrestrialisation of floodplain habitats	Skern *et al.* (2010)
	Water pollution levels	
	• Heavy metal accumulation	Nummelin *et al.* (2007) Pettigrove & Hoffman (2005)
	Sediment pollution levels	
	Environmental management success	
	• Mine restoration	Orabi *et al.* (2010)
Miridae (Family)	Water pollution levels	
Nabis kinbergii	• Insecticides	Cole *et al.* (2010)
Thysanoptera (Order)	Agricultural management success	
	• Olive-orchard management regimes	Ruano *et al.* (2004)

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3.3.8   *Diptera as bioindicators*

Diptera are well known as water quality and forensic bioindicators, and have been used to test the level of disturbance as well as in palaeontology to estimate climatic variables (Table 3.3.8). It is the family Chironomidae that stands out as a bioindicator in fresh water studies.

3.3.9   *Coleoptera as bioindicators*

Coleoptera species often make good indicators, and can be used across the spectrum, from bioindicators of biodiversity, ecological integrity to water quality. Coleoptera have been used in water quality analysis, forensic investigations, as well as in various environmental and agricultural management systems, to detect disturbances (Table 3.3.9). The families Carabidae, Scarabeaidae and Staphylinidae have been particularly significant in these activities.

3.3.10 *Lepidoptera as bioindicators*

Lepidoptera, more specifically the family Nymphalidae, are an European favoured bioindicator of biodiversity and has been used for measuring levels of disturbances, particularly in the forest environment (Table 3.3.10).

3.3.11   *Hymenoptera as bioindicators*

Hymenoptera as bioindicators (Table 3.3.11) include large ants but also honeybees particularly as environmental indicators of pollutant levels (Samways *et al*., 2009). Ants have been strongly promoted as bioindicators (Alonso, 2000; Andersen *et al*., 2002), while parasitoids, have been proposed as indicators of sustainability (Thomson *et al*., 2007).

Table 3.3.8 Literature review of the order Diptera used as bioindicators and for what they are indicators

Taxa	Bioindicator of	Reference
Diptera (Order)	Environmental status	
	• Climate change	Ciamporova-Zat`ovicova *et al.* (2010)
	Physiochemical characteristics of water	
	• Hydromorphological degradation	Korte (2010)
	• Hydrological change	Luoto (2010)
	• Water chemistry	Parsons *et al.* (2010)
	Water pollution levels	Imoobe & Ohiozebau (2010)
	Palaeontology	
	• Quantitative temperature	Velle *et al.* (2010)
	Forensic entomology	
	• Colonization time	Pohjoismaki *et al.* (2010)
	• Decomposition and dipteran succession	Horenstein *et al.* (2010)
	• Buried bodies	Szpila *et al.* (2010)
Chironomidae (Family)	Environmental status	
	• Water quality	Al-Shami *et al.* (2010a)
	Physiochemical characteristics of water	
	• Changes in pH	Rees & Cwynar (2010)
	• Hypolimnetic oxygen	Luoto & Salonen (2010)
	Water pollution levels	Samways *et al.* (2009)
	Sediment pollution levels	Di Veroli *et al.* (2010)
	Palaeontology	
	• Paleotemperature	Eggermont *et al.* (2010)
Chironomus (Genus)	Level of disturbance	
	• Anthropogenic stress	Al-Shami *et al.* (2010b)
Chironomus riparius	Water pollution levels	Langer-Jaesrich *et al.* (2010b)
	• Insecticide - Thiacloprid	Langer-Jaesrich *et al.* (2010a)
	Sediment pollution levels	Langer-Jaesrich *et al.* (2010b)

Table 3.3.9 Literature review of order Coleoptera used as bioindicators and for what they are indicators

Taxa	Bioindicator of	Reference
Coleoptera (Order)	Environmental status	
	• Biodiversity	Sanchez-Fernandez *et al.* (2006)
	o Ponds	Indermuehle *et al.* (2010)
	• Ecological integrity	Kaiser *et al.* (2009)
	• Water quality	Benetii & Garrido (2010)
	o Standing pools	Picazo *et al.* (2010)
	Physiochemical characteristics of water	
	• Floodplain characteristics	Schipper *et al.* (2010)
	• Hydromorphological degradation	Korte (2010)
	Water pollution levels	Imoobe & Ohiozebau (2010)
	Level of disturbance	
	• Human population	Della-Bella & Mancini (2009)
	• High-altitude ski pistes	Negro *et al.* (2010)
	Forensic entomology	
	• Decomposition rate	Voss *et al.* (2009)
		Shi *et al.* (2009)
	Environmental management success	
	• Restoration	Paoletti *et al.* (2010)
		Babin-Fenske & Anand (2010)
	• Spruce forests	Bishop *et al.* (2009)
	• Conservation planning	Gomez (2010)
	Agricultural management success	
	• Land use	Song *et al.* (2009)
	Environmental status	Niemelä *et al.* (2000)
	Palaeontology	Ponel *et al.* (2003)
Carabidae (Family)	Environmental pollution levels	
	• Insecticides	Ito *et al.* (2010)
	Environmental status	Pearson & Cassola (2005; 2007)
Cicindelidae/-inae (Family/Subfamily)	Environmental status	
Scarabaeidae (Family)	• Habitat quality	Samways *et al.* (2009)
	Level of disturbance	Samways *et al.* (2009)
	• Farming	Jacobs *et al.* (2010)
	Environmental status	
Staphylinidae (Family)	• Habitat quality	
	o Sub-Andean rural landscape	Vasquez-Velez *et al.* (2010)
	Environmental pollution levels	
Atheta coriaria	• Toxins	
	o Cry1Ab	Garcia *et al.* (2010)

Table 3.3.10 Literature review of the order Lepidoptera used as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Lepidoptera	Environmental status	
	• Biodiversity	Zografou *et al.* (2009)
	Level of disturbance	
	• Logging	Samways *et al.* (2010)
	• Timber harvest	Summerville *et al.* (2009)
	• Forest disturbance	Hayes *et al.* (2009)
	• Rainforest fragmentation	Uehara-Prado & Freitas (2009)
	• Land use	Kadlec *et al.* (2009)
		Haughton *et al.* (2009)
	Environmental management success	
	• Biogeographical analysis	Romo & Garcia-Barros (2010)
	Agricultural management success	
	• Herbicide resistant	Hilbeck *et al.* (2008)

Table 3.3.11 Literature review of order Hymenoptera used as bioindicators and for what they are indicators

Taxa	Bioindicator of	Reference
Hymenoptera (Order)	Environmental status	Alonso (2000) Kasperi & Majer (2000) Andersen *et al.* (2002) Thomson *et al.* (2007)
	• Sustainability Level of disturbance	
	• Invasive species	Yemshanov *et al.* (2011)
	• Complete clearance	Paolucci *et al.* (2010)
	Environmental pollution levels	Samways *et al.* (2010)
	• Insecticides	Pereira *et al.* (2010)
	Forensic entomology	Voss *et al.* (2009)
	Environmental management success	
	• Wayana Amerindian land use	Delabie *et al.* (2009)
	• Restoration	Majer (1983)
Formicidae (Family)	Environmental management success	
	• Restoration	Coelho *et al*. (2009) Dekoninck *et al.* (2008)
Vespidae (Family)	Environmental management success	
	• Riparian forests	De Souza *et al.* (2010)
Apidae (Family)	Environmental pollution levels	
Apis mellifera	• Toxins	
	o Chlorfluazuron, Oxymatrine, Spinosad	Rabea *et al*. (2010)

3.3.12 *Other Endopterygota as bioindicators*

Of the remainder of the superorder Endopterygota, it is order Trichoptera and its use in water quality evaluations that is prominent as bioindicators (Table 3.3.12a), while orders Neuroptera and Megaloptera also feature in the literature in the use of agricultural management and water analysis respectively. Orders Mecoptera and Siphanoptera are not mentioned as bioindicators. Although, there have been some studies that indicate that ectoparasites, like Siphanoptera, can be indicators of stress in small mammals (Matthee & Krasnov, 2008).

3.4 DISCUSSION

Bioindicators can indicate the changes in the abiotic or biotic state of the environment and how affected the environment is by perturbational stress caused by humans (McGeoch, 2007). Bioindicators can also help us to interpret and understand the effect of our actions on a habitat, community or ecosystem. Bioindicators can additionally be suggestive of the diversity within a biotope. For these approaches to be effective, these bioinidicators need to be in abundance, and sampling them should not put the habitat, community or ecosystem under any stress. As arthropods occur in most biotopes, ecosystems and environments, in large numbers, they are among the first choice in bioindication studies.

When deciding on a potential bioindicator taxon, it is important to make a decision that is appropriate for the bioindication objective (Duelli & Obrist, 2003). The scale of the study must be clearly defined (Debuse *et al.,* 2007), and use of a standardized methodology is recommended (Neville & Yen, 2007). Samways *et al*. (2010) suggested using of a range of different taxa which may represent different functional groups as surrogates to lessen the significance of the study. Samways *et al.* (2010) provide a table of useful characteristics for selecting potential bioindicators prior to testing (Table 3.4.1). According to these criteria, arthropods could, as Kremen *et al*. (1993) pointed out, be used for virtually any monitoring programme, so long as the goals are well defined.

The results of this literature review suggest that terrestrial and aquatic invertebrates have been widely used as a management tool for monitoring change in ecosystems. Their suitability and value for assessing a range of environmental problems from pollution impacts, through habitat evaluation for conservation to the long-term degradation and recovery of ecosystems is clear (Hodkinson & Jackson, 2005).

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Table 3.3.12a Literature review of the superorder Endopterygota used as bioindicators and for what they are indicators

Order	Bioindicator of	Reference
Trichoptera	Environmental status	
	• Biotic integrity	Masese *et al.* (2009a)
	• Ecological integrity	Arimoro & Ikomi (2009)
	• Climate change	Lawrence *et al.* (2010)
	• Ecological impairment	Petty *et al.* (2010)
	• Ecological status	Sanchez-Montoya *et al.* (2010)
	• Ecosystem health	Young & Collier (2009)
	• Environmental health	Gamboa *et al.* (2008)
	• Environmental change	Luoto (2009)
	• Habitat quality	Hall *et al.* (2009)
		Sullivan & Watzin (2008)
	• Stream health	Collier (2008)
	• Water quality	Paparisto *et al.* (2010)
		Gabriels *et al.* (2010)
		Masese *et al.* (2009b)
		Purcell *et al.* (2009)
	Physiochemical characteristics of water	
	• Hydromorphological degradation	Korte (2010)
	• Morphological degradation	Chakona *et al.* (2009)
	• Stream types	Dohet *et al.* (2008)
	• Stream flow characteristics	Konrad (2008)
	• Environmental characteristics	Takao *et al.* (2008)
	Water pollution levels	Aizawa *et al.* (2009)
	• Heavy metal accumulation	Girgin *et al.* (2010)
	Level of disturbance	
	• Acid mine drainage	Ross *et al.* (2008)
	• Impoundments	Bredenhand & Samways (2009)
		Rybak & Sadlek (2010)
	• Timber harvest	Gravelle *et al.* (2009)
	• Human population size	Pautasso & Fontaneto (2008)
	• Human activities	Della-Bella & Mancini (2009)
	Environmental management success	
	• Stream restoration	Selvakumar *et al.* (2010)
	Agricultural management success	
	• Land use	Song *et al.* (2009)

Table 3.3.12b Literature review of superorder Endopterygota used as bioindicators and for what they are indicators

Taxon	Bioindicator of	Reference
Megaloptera (Order)	Physiochemical characteristics of water	
	• Trophic state of ponds	Menetrey *et al.* (2005)
Neuroptera (Order)	Agricultural management success	New (2007)
	• Olive-orchard management	Ruano *et al.* (2004)
	Environmental pollution levels	Zeleny (1978)
	• Pesticides	Booth *et al.* (2003)
Chrysopa perla	Environmental pollution levels	Clarke (1993)

Table 3.4.1 Suggested useful characteristics for selecting potential bioindicators prior to testing (Samways *et al.*, 2010)

- Cost efficient and effective (time, funds, personnel)
- Sample and sort easily
- Adequate representation in samples (abundance)
- Ease and reliability of storage
- Taxonomically well known, readily identified, taxonomic expertise available
- Sample individuals expendable
- Spatial and temporal distribution predictable to ensure long-term continuity
- Relatively independent of sample size
- Baseline data on biology and population dynamics available
- Low genetic and functional variability
- Sufficiently sensitive to provide early warning
- Able to differentiate between natural cycles and trends and those produced by anthropogenic stress factor
- Representative of critical components, functions and processes
- Show a well defined response, i.e. either die or decrease, change or mutate, replace or be replaced by other species
- Readily accumulate pollutants
- Easily cultured in the laboratory
- Capable of providing continuous assessment over a wide range of stress
- Recognized importance of agriculture, environment. etc.
- Economic importance as a resource or pest
- Representative of all trophic levels and major functional guilds
- Representative of related and unrelated taxa
- Include a broad range of body sizes and growth forms
- tend to be distributed over a range of habitats or environments
- Representative for low, medium and high diversity groups
- Wide range of host specificities

It is in the aquatic environment that invertebrates have been widely used for bioindication. Aquatic arthropods are used for evaluating factors that affect aquatic insect distribution, like oxygen availability (Verbruggen *et al.,* 2011), temperature (Hester & Doyle, 2011), sediment and substrate type (Wellnitz *et al.,* 2010) and presence of pollutants such as pesticides, acidic materials and heavy metals (Abbasi & Abbasi, 2011).   In the aquatic systems, crustaceans have been the main bioindicators in the marine environment (Rinderhagen *et al.,* 2000), while insects make useful bioindicators for evaluating freshwater systems (Adham *et al*., 2009). Crustaceans in the orders Isopoda, Amphipoda and Decapoda have mostly been used as indicators, mainly because they are abundant within marine environments. The lesser known and rarer Crustacean classes might have great indicator potential; their rarity might be the cause of their sensitivity to change within an ecosystem, but have not been properly been researched.

Ephemeroptera, Plecoptera and Trichoptera (EPT) are known as the trio that makes a good collective indicator of freshwater quality owing to their sensitivity to changes in environmental conditions (Zhou *et al.,* 2009).   Other predator orders like Odonata, Hemiptera, Coleoptera, and Megaloptera are also useful in detection of heavy metal accumulation (Nummerlin *et al.,* 2006). Most dipteran presence is associated with polluted waters, although some families are extremely sensitive to toxicity (Bae *et al.,* 2005). Members of the order Odonata have also been used as indicators of riparian vegetation quality (Simaika & Samways, 2009).

A range of typical responses to aquatic insect communities following disturbance have already been identified, e.g. dissolved oxygen reduction causing haemoglobin-possessing bloodworms (Chironomidae) to increase in abundance (Luoto & Salonen, 2010); pesticide runoff leading to substantial reduction in species diversity (Gullan & Cranston, 2010); increase in nutrient levels due to fertilizer run off causing the abundance of a few species to increase dramatically, while the general species diversity declines (Hansen *et al*., 2003); Plecoptera nymphs decline when temperature increases; increase of particulate material including sediment causing certain species of mayflies in the family Caenidae and Caddisflies in family Hydropsychidae to increase in abundance (Roberts, 2005). Resh & Jackson (1993) provide many additional examples of how aquatic insects and other freshwater macroinvertebrates are used as bioindicators.

In terrestrial environments, arthropods have been extensively used as bioindicators, traditionally to test soil fertility and to determine the level of pollution. Recently, arthropods have become more popular due to the fact that bioindication is more financially viable than the more expensive chemical and physical analyses. The orders Araneae and Acari, are the main taxa used as bioindicators in the subclass Chelicerata. Spiders, as micropredators, are regularly used as indicators, owing to their high position in foodwebs. Heavy metal pollution within the environment can accumulate in their bodies via the food that they eat. Furthermore, spider abundance is dependent on the numbers of their prey, when the environment is disturbed and their prey numbers drop, their numbers will also decline. Other Arachnida in the orders Scorpiones, Pseudoscorpiones, Opiliones, Solphugidae, Schizomida, Uropygi, Ricinulei and Amblypygi are all predators, and thus should give similar trends than Araneae, but have not been studied in detail.

O'Neill *et al.* (2010) identified Collembola species as the most useful bioindicators in soil management. Collembola are frequently being used to evaluate the quality of soil (Barros *et al.,* 2010), pollution levels within soil (Fiera, 2009) and to determine soil properties like moisture content (Natal-da-Luz, 2008). Ground dwelling insects that are easy to sample via pitfalls make good bioindicators. Within the order Coleoptera, the families Carabidae, Cincinellidae(-inae), Staphylinidae and Scarabaeidae feature in the literature mainly for evaluating habitat quality (Niemelä *et al*., 2000; Pearson & Cassola, 2007; Samways *et al*., 2010; Vasques-Velez *et al*., 2010). However, it is the ants that have been usedthe most as bioindicators, owing to their great abundance, ease of sampling and wide range of habitat preferences (Coelho *et al.,* 2009). Bisvac and Majer, (1999) found that ants and beetles were the taxa with the highest correlation with the overall community composition.

In grass and shrub environments, it is the Lepidoptera, Neuroptera, Orthoptera and Hymenoptera that stand out, mainly as indicators of heterogeneity. Nymphalid butterflies dominate the European literature for being a good indicator group, mainly because the biased attention Lepidoptera studies get within Europe, owing to their ecology being well studied and their presence easily monitored (Romo & Garcia-Barros, 2010). The chrysopid larvae have been used for agricultural monitoring, to assess the effects that pesticides have on the natural fauna (New, 2007). Honey, a product of honey bees, can easily be tested for toxins and pollutants within the environment (Rabea *et al.,* 2010). Grasshoppers have been used in

environmental management to evaluate the state of a biotope, or to evaluate the success rate of restoration (Saha & Halder, 2009).

From an environmental and agricultural management perspective, scientists have also identified a range of reaction towards certain disturbances within bioindicator taxa: ants, ground beetles and spiders respond negatively to forest thinning (Pearce & Venier, 2006); Scarab beetles respond negatively to habitat alterations caused by forest fragmentation (Rainio & Niemelä, 1998). The nymphalid butterflies and longhorn beetles respond highly positively to the presence of herbaceous plants and understory trees and can be used to infer the integrity of thinning treatments (Maleque *et al.*, 2009).

Although single species can be used as bioindicators in various situations, questions have been asked about the efficiency of using only a single taxon group instead of multiple taxa (Prendergast, 1997; Pryke & Samways, 2012). Lawton *et al.* (1998), suggested the use of several indicator taxa together, called predictor sets. Predictor sets would ideally encompass different ecological functional groups or guilds.

There are a few aspects to take in consideration when using arthropods as bioindicators. It is important to analyse species specific biomonitoring when the species are present within the system. Various artropods use the egg, larval or pupal phases in their life cycle to overcome unfavourable conditions (Regoli *et al.*, 2002), and these may not be amenable to sampling or apparent at the time of sampling. Clear boundaries need to be set at all levels of spatial scales (McGeoch, 1998). The scale or size of the study area, as well as at what taxonomic level the study will be executed is an important consideration if results are to be used in comparison studies.

More knowledge of bioindicators in different scenarios is needed so as to properly interpret the results, i.e. a pollutant can cause an effect to a bioindicator that is expected, but if that pollutant is present in conjunction with another, it might cause a reaction that causes the bioindicator species to react differently (Bierkens, 2000). When using indicators for monitoring for specific conditions, caution should be used for any given time, as a new or previous dormant factor can come into play that might have a similar effect on the bioindicator species e.g. an invasive species that enters a system and out competes other species. For example, Rainio & Niemelä (2003) used bioindicators to test the effect of pesticides on native fauna. They found that the effect of the invasive species within the

system also caused a reduction in abundance of indicator species, and cannot thus be correlated to the pesticides alone, but also to the invasive species (Rainio & Niemelä, 2003).

Nature is constantly in flux, and to pinpoint a value to a biotope, for comparison, might not indicate the best overall condition, but the best condition at that specific time. Natural disasters are also known to greatly change the status quo. During, and for a period after a disaster, bioindication will not be valuable, with the exception of when the disaster itself is under study (Speight *et al.,* 1999).

Indicator species are not usually interchangeable. Just because a particular taxon has been shown to be useful in one environment for one particular bioindication objective, does not mean that it will necessarily be suitable for any other environment or bioindication objective and must be tested first (McGeoch, 2007). Generally the biggest problem with the use of bioindicators is the lack of knowledge about the species and their ecologies within the biotope. This is especially true for the highest risk areas where monitoring is most needed.

Furthermore, only data on indicator species which indicate a positive correlation with a target variable are usually published that can leed to scientists wasting resources and time on repeating bioindicator testing.

3.5   CONCLUSION

Biological monitoring using arthropods has many advantages. Arthropods, being mobile and reactive to changing conditions, often respond by moving away from, or towards, adverse or optimal conditions respectively. They are widely distributed and present in all terrestrial environments, except the most extreme. A bioindicator species proved to be successful in one part of the world, can steer future studies to find similar bioindicator species from the same taxa, to be used in similar environments. Most arthropods are small and show a diversity of growth rates, life history styles, body sizes, food preferences, and ecological preferences, and can, at the species level, often be linked to specific environmental variables. This means that the smallest environmental change can affect their population levels and even their precence or absence. These characteristics enable researchers to choose a specific taxon according to the needed task and its required resolution. Arthropods have fast breeding rates and short generation times, and therefore are often highly responsive numerically to environmental change. Some environmental changes may be subtle and a result of complex interactions between abiotic and biotic components that cannot be measured directly, but can

be seen in the behaviour or local survival of the species. Sometimes, the biotic community continues to change long after physical or chemical traces of the impact are no longer measurable, yet the effects can be seen in the arthropod community. Arthropods are mostly easy and safe to sample. No elaborate equipment is usually needed, making the use of arthropods as bioindicators more cost effective compared to that of other taxa and methods (Markert *et al.*, 2003).

My literature review identified Ephemeroptera, Plecoptera, Trichoptera and Diptera as widely used bioindicators in the aquatic environment. In the marine environment, crustaceans, especially the isopods, amphipods and decapods, have been most used as bioindicators. Odonata make good indicators of the riparian environment.

In the terrestrial environment, spiders in the predator guild, grasshoppers in the herbivore guild, springtails in the decomposer guild, nymphalid butterflies in the pollinator guild and ants as scavengers, make good bioindicators under certain conditions. The order Coleoptera make for good bioindicator species due to their high diversity and wide distribution.

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Chapter 4

DEVELOPMENT OF THE BIOTOPE QUALITY INDEX AS A MEASURE OF BIOTOPE QUALITY

ABSTRACT

A healthy ecosystem can be viewed as one that can sustain an optimal number of species with optimal abundance and sustain their ecological processes. This provides an optimal heterogeneous, sustainable system with adequate resources for maintenance, and has adequate resistance when under perturbational stress, yet enables natural succession still to take place. I introduce here a methodology for determining the health of a range of biotopes by making use of a mean species specific assemblage. Species with above average abundance (logged), at all sites, are identified for each site. The number of standard deviation units above the mean abundance of each morphospecies is combined to calculate an Biotope Quality Index value for each biotope [$BQI = \sum_{i=1}^{n} (\frac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i})$ when $\frac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i} > 0$]. This method can be used for management of reserves as well as for evaluating the effect of environmental impacts.

KEYWORDS: Biotope assessment; Biotope quality; Biotope Quality Index; Environmental health

4.1   INTRODUCTION

Increased human pressure on ecosystems across the world is causing habitat loss and ecosystem health decline (Dietz *et al*., 2007). These perturbation pressures, such as pollution, deforestation, alien invasion, exploitation, erosion and global climate change are all impacts resulting from little regard for the sustainability of the global ecosystem (Haskell *et al*., 1992; Mooney *et al*., 2005).

There has also been concern over the potential extinction of thousands of species worldwide (Mooney *et al*., 2005; Hockey & Curtis, 2009), with the mitigation response aimed principally at individual threatened species (UNEP, 2011), especially large vertebrates, certain plants and conspicuous invertebrates. Such single species approaches do not include networks of interacting species (Memmott, 2010), nor any relationship with their abiotic surroundings (Kemp, 1998), despite the fact that loss of a single keystone species can result in a major change in local environmental integrity (Tews *et al*., 2004). Butchart *et al.* (2010) indicated that the mitigation process focusing on the decline in biodiversity from 2002 to 2010 had no significant effect on reductions in rate of decline.

No comprehensive standard has been developed to assess the biological condition or quality of a biotope, or to define the nature and extent of degradation (Karr, 2004). Currently, ecosystems are being evaluated for heterogeneity, making use of various species surrogates such as species richness, relative abundance, diversity indices and phylogenetic indices, as well as environmental surrogates at the landscape level (Faith & Walker, 1996; Clarke & Warwick, 1998; Reyers *et al*., 2002). These values are then used in management of ecosystems, where systems with high heterogeneity are considered more important than systems with low heterogeneity. Yet, quality of the biotopes at the landscape level is rarely taken in consideration (Dennis *et al*., 2007). The triage approach (Samways, 2005) considers ecological integrity as well as disturbance levels for predicting an ecosystem's ability to recover, and whether complete restoration is feasible. Yet monitoring these systems is rarely put in place to evaluate the recovery process.

Evaluating the health of ecosystems involves a pluralistic approach and an array of indicators of system status (Rodrick & Karwowski, 2006). Richardson *et al.* (1998) suggested that possibilities for effective management can be enhanced by the availability of various databases, legislation and management tools for planning, zoning and locating areas of high

conservation status, but most important is the need to evaluate the quality of these systems as well as their ascendancy (UNEP, 2011). Ulanowicz (1997) defined network ascendency as the product of the aggregate amount of material or energy being transferred in an ecosystem multiplied by the coherency with which the outputs from the members of the system relate to the set of inputs to the same components (Scharler, 2009).

A first step is to define what is meant by biotope quality. A clearer conception of the quality and health of natural environments will assist in bringing our regulatory mandates in line with legislative procedures (Haskell *et al*., 1992).   A biotope is defined as the smallest geographical unit of the biosphere or a habitat that can be delimited by convenient boundaries and is characterized by its biota (Allaby, 2010). In turn, a healthy ecosystem can be seen as one that can sustain an optimal number of species with an optimal population size and sustain their ecological processes, thus providing an optimal heterogeneous, sustainable system with sufficient resources, and has adequate resistance when under perturbational stress, yet enable natural succession to still take place (Chapter 2, p.12-46). Thus, corresponding to Bratton's (1992) climax theory, it implies that the climax community in the successional sequence has maximal biomass, most complex nutrient cycles, greatest productivity, greatest species diversity, and be able to maintain itself indefinitely when freed from major disturbances (Su *et al*., 2010).

Historically, the health of a biotope has been measured using indices of particular species or components (Haskell *et al*., 1992).   These measures include species richness, Simpson-Yule index, Shannon-Wiener function, Berger-Parker dominance index, Brillouin index and McIntosh diversity index (Simpson, 1949; McIntosh, 1967; Stilling, 1999; Henderson, 2003; Schneider, 2009).   Furthermore, Ulanowicz's (1980) index of network ascendancy integrates four characteristics of ecosystems: species richness, niche specialization, developed cycling and feedback, and overall activity (Scharler, 2009).   Network analysis allows for quantitative, integrated, hierarchical treatment of all complex ecosystems. Ascendancy assesses not only species richness in an ecosystem but also the way in which these species are organized (Scharler, 2009).

Other specialized indices have been developed for measuring quality of water and aquatic environments such as the index of biotic integrity (Karr, 1981), which computes the sum of scores for 12 characteristics of a fish community (Conti, 2008). This index has been adapted

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for specific localities according to the species present in the area. Invertebrate presence is also used in water quality indices (Schaeffer & Cox, 1992; Dickens & Graham, 2002, Simaika & Samways, 2009). Likewise, birds are used in the Bird Community Index (O'Connell *et al*., 2000) and also to evaluate forest conditions (Canterbury, 2000). However, Bradford *et al*. (1996) pointed out that the sensitivity of bird matrices appears to be greatest at the high impact end of the spectrum, which suggests they may have limited utility in distinguishing between sites experiencing light and moderate impact (O'Connell *et al*., 2007). O'Connell *et al*. (2000) reported a similar problem from medium to poor ecological conditions, with a shift in land cover composition from forested to non-forested biotopes. Niemi *et al*. (1997) suggest focusing on comprehensive techniques that improve habitat classifications as well as combining monitoring of trends in biotope types with bird assemblages within those biotopes, rather than focusing on a few representative species.

Physical properties of the abiotic components of the environment involve calculable evaluations, and is commonly substituted for a measurement of environmental health. Examples of abiotic indices include the universal soil loss equation and the predicted index of abiotic integrity (Schaeffer & Cox, 1992; Winchell *et al*., 2008). These indices, with the exception of network ascendancy, are also used to calculate other environmental properties and then simply surrogated for measuring biotope quality. However, they are inadequate because they do not reflect the complexity of the biotope or ecosystem (Haskell *et al.,* 1992). Furthermore, network ascendancy requires absolute knowledge of all species present, as well as the interactions among each other and their environment.

I develop here and test an Biotope Quality Index (*BQI*) using invertebrate assemblages to evaluate biotope quality in a similar way to the development of indices for habitat quality (Powell & Powell, 1986; Moilanen, 1998; Canterbury *et al*., 2000;Knight, 2003; Simaika & Samways, 2009; Kennedy *et al*., 2011), water quality (Karr 1981, 1987, 1990, 1991; Karr *et al*., 1986; Haskell *et al*., 1992; Chainho *et al*., 2008; Fishar & Williams, 2008; Novotny *et al*., 2009), ecosystem health and integrity (Schaeffer *et al.,*1988; Novak & Schaeffer, 1990; Norton, 1992; Bradford *et al.,* 1998; Kane *et al*., 2009) and habitat availability (Saura, 2007).

4.2   METHODS

4.2.1   *Study area and sampling methods*

Thirty sites in the Jonkershoek Valley, in the Cape Floristic Region, South Africa (38°58' S; 18°55' E), were selected to represent six main biotope types, and an *a priori* range of disturbance levels from fully natural to disturbed. Sites were categorized into three natural fynbos vegetation biotope types: <0.5 m-high, Ericaceae-dominant fynbos (five sites), ±1m-high Restionaceae-dominant fynbos (four sites) and >1.5 m-high Proteaceae-dominant Fynbos (five sites). Additional sites included grass-invaded fynbos (five sites) as well as exotic pine plantations categories <10 yr old pine stands (six sites) and >10 yr old pine stands (five sites) (Figure 4.2.1).
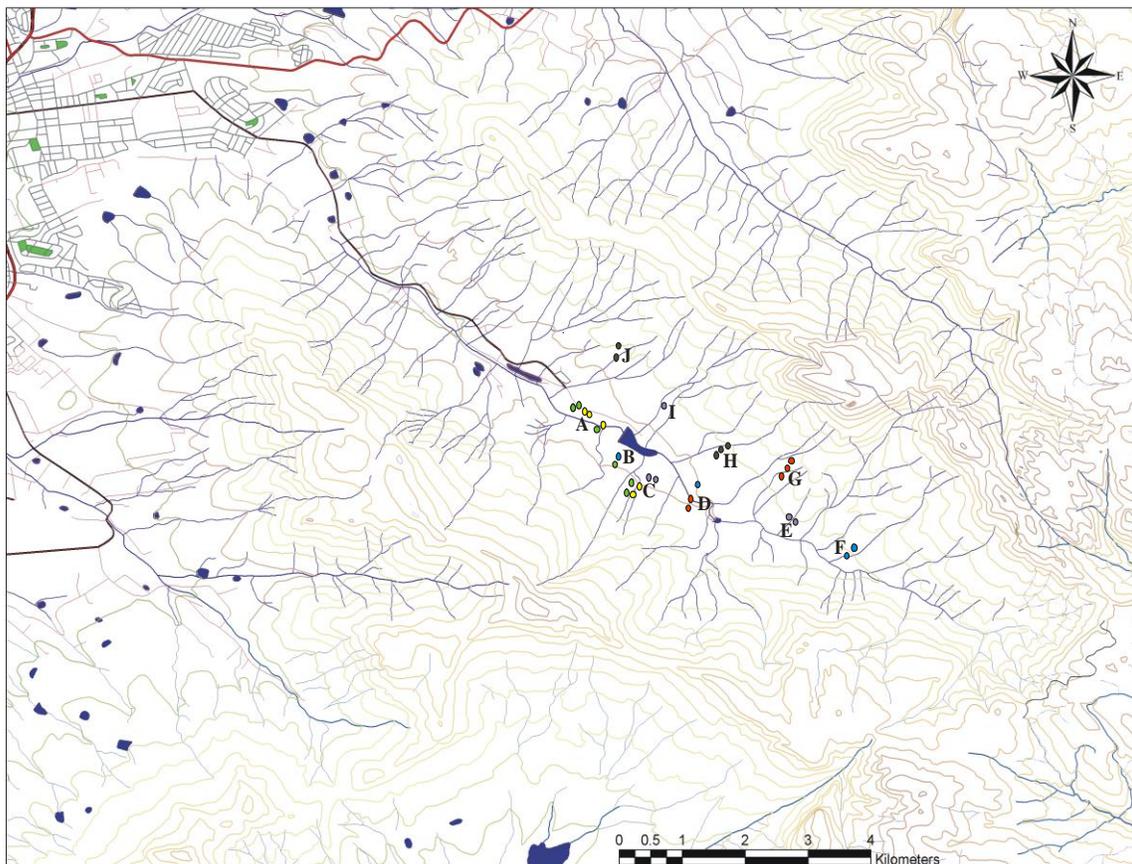
Figure 4.2.1   Location of 30 sampling sites at locations (A-J), in the Jonkershoek Valley, Western Cape, South Africa

Each site was evaluated according to integrity and disturbance levels, and rated from near-natural to least-natural, using mean values from the opinions of a group of Masters & PhD research students within Conservation Ecology and Entomology department of Stellenbosch University. This was achieved by making use of a rubric scoring table system. The disturbance level was then calculated taking in consideration, pollution (0 = no visual signs of pollution : 10 = rubbish dump), invasive aliens(0 = no invasive plant species present : 10 = 100% alien plants), distance from nearest disturbance (0 = more than 1000 m from nearest road ; 10 = less than 5 m from nearest road), human activity (0 = no visual signs of any human activity : 10 = building with constant human presence) and biotope structure (0 = signs of new growth and no unhealthy plants : 10 = diseased plants and no sign of natural new growth). This was a subjective approximation of data. This non-parametric method gave data against which I could compare the *BQI* to identify a positive trend, seeing that there is no current practical quantitive calculation to measure environmental health, integrity or disturbance as a whole. Co-ordinates and elevation was derived from a GPS, while slope (α) was calculated making use of the formula $\tan \alpha = \frac{\Delta h}{d}$, where Δh is the difference in altitude between two points and d equal the distance between two points.

Using the method described by Woodcock (2000), ten 350 ml pitfall traps (Woodcock, 2000), were placed at random at each of the 30 sites for a period of seven days in each of March, June, September and December 2006, representing the four seasons. Ants (Formicidae) and Springtails (Collembola) were selected as the focal taxa for this study, being species rich and abundant. All individuals were sorted to morphospecies.

4.2.2   *Comparison of variations in the Biotope Quality Index*

The *BQI* concept was derived from the definition of a healthy ecosystem being viewed as one that can sustain an optimal number of species with optimal abundance and sustain their ecological processes. I identified only populations of invertebrates that have more individuals than the mean population for that morphospecies. Additionally, a biotope with more morphospecies will thus have more values to contribute to the final value. The question then arose to what statistical measurement will show best correlation to disturbance. Six versions of

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the *BQI* were evaluated to determine which of these most closely matched the level of disturbance, making use of regression analysis. These included 'above average count quality index' (BQI_1), 'standard deviation above average quality index' (BQI_2), 'above average count on log data quality index' (BQI_3), 'standard deviation above average count of log data quality index' (BQI_4), 'distance from average count quality index' (BQI_5) and 'distance from average count on log data quality index' (BQI_6).

The above average count quality index was the original idea behind the index, where I count the number of morphospecies with above average abundances within each biotope. This was calculated by $BQI_1 = \sum_{i=1}^{n}(a_i = 1)$ when $a_i = x_{ij} - \bar{x}_i > 0$ where x_{ij} indicates the relative abundance for morphospecies i at site j, $\bar{x}_i$ indicates the mean relative abundance of the ith morphospecies, n the number of morphospecies and a the result if the condition set between the differences of x_{ij} and $\bar{x}_i$ are larger than zero.

I had concerns around giving the same score to populations that are significantly higher than those slightly above average. Therefore, I also decided to count the number of standard deviation points from the average to see if it does not fit the correlation better. The standard deviation above average quality index was thus calculated by $BQI_2 = \sum_{i=1}^{n}(\frac{x_{ij} - \bar{x}_i}{\sigma})$ if $\frac{x_{ij} - \bar{x}_i}{\sigma} > 0$, with similar conditions as in BQI_1 with the addition of standard deviation (σ), calculated $\sigma = \sqrt{\frac{\sum(x_{ij} - \mu_i)^2}{N}}$, where μ_i indicates the population mean of the ith morphospecies and N the number of morphospecies sampled.

A further concern was that the abundance data would not follow a natural distribution curve, and therefore I included a variable of each type, by first logging all abundance data. Above average count on log data were calculated as BQI_1 with the exception that data were normalized by taking the log value of counted data, giving the formula $BQI_3 = \sum_{i=1}^{n}(a_i = 1)$ where $a = \log x_{ij} - \overline{\log x_i} > 0$.

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As with BQI_3, the standard deviation above average count on log data, must first see the

data normalized before calculating the index, giving the formula: $BQI_4 = \sum_{i=1}^{n}(\dfrac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i})$

when $\dfrac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i} > 0$.

I then thought that what if the number of specimens above the average might be a better indication, as it's a smaller unit than standard deviation. The distance from average count was calculated by $BQI_5 = \sum_{i=1}^{n}(x_{ij} - \overline{x_i})$ where $(x_{ij} - \overline{x_i}) > 0$. I then also created a similar version on data that was first logged. The distance from average count on log data must also first be normalized as $BQI_6 = \sum_{i=1}^{n}\log x_{ij} - \overline{\log x_i}$, where $\log x_{ij} - \overline{\log x_i} > 0$.

The best fit calculation is termed here the Biotope Quality Index (BQI) for further comparison and discussion.

4.3   RESULTS

A total of 18 649 ant individuals in 41 species, and 14 646 springtail individuals in 33 species was used for evaluation and determining the BQI. Coordinates, aspect, slope and results of the integrity and disturbance level analysis are given in Table 4.3.1.

The best fit of BQI_{1-6} was 'Standard deviation above average count of log data quality index' (BQI_4) with the strongest correlation to the disturbance level ($R^2 = 0.833$), followed closely by 'Above average count on log data quality index' (BQI_3; $R^2 = 0.826$), 'Standard deviation above average quality index' (BQI_2; $R^2 = 0.705$), 'Above average count quality index' (BQI_1; $R^2 = 0.704$), 'Distance from average count on log data quality index' (BQI_6; $R^2 = 0.670$) and finally 'Distance from average count quality index '(BQI_5) indicating weak correlation to disturbance levels ($R^2 = 0.299$) (Figure 4.3.1 a & b).

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Table 4.3.1 Site attributes and disturbance levels for the sampling sites at Jonkershoek, South Africa. The mean level of disturbance for each biotope category is given in brackets.

Site	South	East	Aspect	Slope (°)	Disturbance level
Fynbos - Ericaceae dominant (<0.5m - high)					(2.32)
C.v	33°59'11"	18°57'13"	NE	8	4.34
C.vi	33°59'12"	18°57'14"	NE	11	4.43
E.i	33°59'25"	18°58'11"	ESE	18	1.61
E.ii	33°59'26"	18°58'12"	SE	27	1.02
I.i	33°58'36"	18°56'51"	W	14	0.22
Fynbos - Restionanceae dominant (±1m - high)					(2.75)
B.i	33°59'03"	18°56'38"	N	28	3.60
D.iii	33°59'15"	18°57'20"	SE	7	2.50
F.i	33°59'40"	18°58'32"	E	46	1.85
F.ii	33°59'42"	18°58'29"	E	65	3.04
Fynbos - Proteaceae dominant (>1.5m - high)					(3.47)
D.i	33°59'18"	18°57'19"	ESE	9	4.50
D.ii	33°59'22"	18°57'18"	ENE	8	5.20
G.i	33°59'19"	18°58'08"	SW	37	1.25
G.ii	33°59'18"	18°58'08"	SW	37	3.22
G.iii	33°59'17"	18°58'09"	SSW	40	3.16
Poaceae Invaded Fynbos					(4.05)
A.iii	33°58'31"	18°56'19"	NE	12	5.99
A.iv	33°58'31"	18°56'21"	N	10	5.47
A.v	33°58'39"	18°56'36"	NE	13	2.89
C.iii	33°59'13"	18°57'11"	NNE	40	3.60
C.iv	33°59'14"	18°57'10"	NE	54	2.30
Pine plantation (< 10 year old)					(5.05)
A.i	33°58'13"	18°56'38"	NE	5	6.60
A.ii	33°58'14"	18°56'34"	NE	8	6.60
A.vi	33°58'41"	18°56'36"	NNE	15	3.30
B.ii	33°59'04"	18°56'57"	E	38	6.40
C.i	33°59'12"	18°57'11"	NE	39	4.20
C.ii	33°59'13"	18°57'09"	ENE	42	3.20
Pine plantation (> 10 year old)					(7.21)
H.i	33°59'07"	18°57'28"	SW	11	8.00
H.ii	33°59'07"	18°57'27"	W	11	6.90
H.iii	33°59'08"	18°57'26"	W	8	8.17
J.i	33°58'21"	18°56'33"	W	17	7.80
J.ii	33°58'18"	18°56'34"	W	14	5.20

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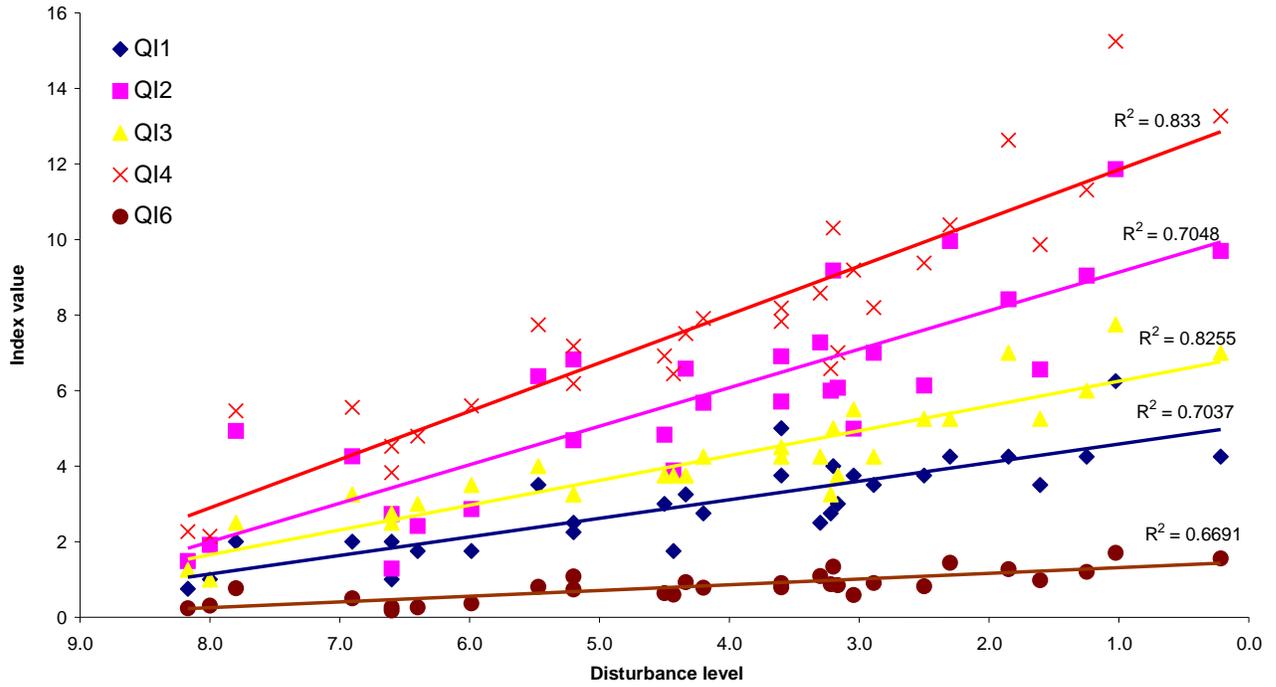
Figure 4.3.1a Correlation between variations of the Biotope Quality Index and disturbance levels for the 30 sites .
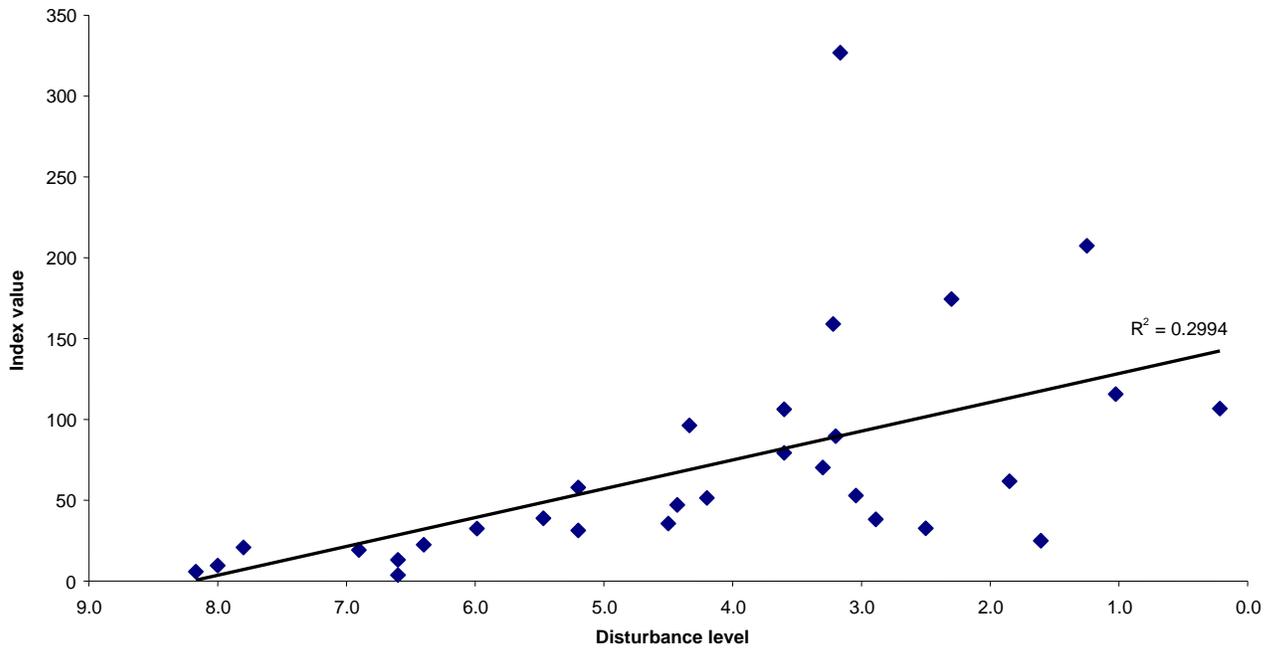
Figure 4.3.1b Correlation between the *QI $_5$* version of Biotope Quality Index and disturbance levels for the 30 sites

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4.4   DISCUSSION

This *BQI* (explained step by step in Box 1) is based on the assumption that any biotope that is able to sustain above average populations, must be in a healthy state i.e. one that can sustain an optimal number of species and their ecological processes. In other words, it must provide an optimal heterogeneous sustainable system with sufficient resources, and have adequate resistance when under perturbational stress, while still allowing natural succession to take place (Chapter 2, p.12-46).

Dale *et al.* (2000) recognized that the responses of the land to changes in use and management depend on expressions of five fundamental principles in nature to be taken in consideration when evaluating biotope quality. Firstly, ecological processes take place within a particular temporal scale and also vary over time. Secondly, the shape, size and spatial relationships of biotope patches on the landscape scale will affect the structure and functioning of communities within the biotope. Additionally, each biotope or site has a distinctive set of species and abiotic conditions affecting and constraining ecological processes. While the species, in turn, are interactive and will have differential effects on those ecological processes. And lastly, perturbation pressures are important and ubiquitous ecological events whose effects may influence ecosystem, community and population dynamics.

Maximizing diversity to achieve sustainability and systemic health is done by avoiding the abnormal or the pathological (Fowler, 2008). Therefore, the *BQI* takes heterogeneity into consideration, and with a high number of species present, this will generate a higher quality value. Here, it is recommended to use as many morphospecies as possible to give the clearest picture.

The *BQI* compares only the relative abundance of one species with the information obtained for that particular evaluated species from the range of values. This means that it only compares similarities and not values across different species.

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Box 4.4.1 *Example of how to calculate the Biotope Quality Index (BQI)*

1. Create a table consisting of sample sites or repetitions as columns, each species as a row, and enter relative abundance data (Table 4.4.1).

Table 4.4.1 Example of data matrix indicating relative abundance of each species per site

	Site A	Site B	Site C	Site D	Site E
Morphospecies 1		23	56	5	109
Morphospecies 2	4	3	2	8	5
Morphospecies 3	2		8		
Morphospecies 4	1	1	1	2	
Morphospecies 5	1753		2367	1876	2354

2. Calculate the logarithm for each of the entries on the data matrix (Table 4.4.2) (Entries equalling zero are ignored {Log 0 does not exist} or adapt all entries by adding one before calculating the logarithm: $Log(x+1)$).

Table 4.4.2 Example of data matrix indicating the logarithm of the relative abundance of each species per site

	Site A	Site B	Site C	Site D	Site E
Morphospecies 1		1.362	1.748	0.699	2.037
Morphospecies 2	0.602	0.477	0.301	0.903	0.699
Morphospecies 3	0.301		0.903		
Morphospecies 4	0	0	0	0.301	
Morphospecies 5	3.244		3.374	3.273	3.372

3. Calculate the mean and standard deviation for each morphospecies (Table 4.4.3).

Table 4.4.3 Table indicating the mean and standard deviation calculated for each morphospecies

	Mean	StDev
Morphospecies 1	1.46	0.58
Morphospecies 2	0.60	0.23
Morphospecies 3	0.60	0.43
Morphospecies 4	0.08	0.15
Morphospecies 5	3.32	0.07

4. Identify all entries that are above its relative morphospecies mean. Delete all entries that are less (Table 4.4.4).

Table 4.4.4 Example of data matrix indicating which entries are above the relative mean

	Site A	Site B	Site C	Site D	Site E	Mean
Morphospecies 1		~~1.362~~	1.748	~~0.699~~	2.037	1.46
Morphospecies 2	0.602	~~0.477~~	~~0.301~~	0.903	0.699	0.60
Morphospecies 3	~~0.301~~		0.903			0.60
Morphospecies 4	~~0~~	~~0~~	~~0~~	0.301		0.08
Morphospecies 5	~~3.244~~		3.374	~~3.273~~	3.372	3.32

Box 4.4.1 (Continued) Example of how to calculate the Biotope Quality Index (*BQI*)

5. Take the difference between remaining values and their relative morphospecies mean and then divide by the standard deviation of each relative morphospecies. Thus, counting the number of standard deviation points from their relative morphospecies' mean (Table 4.4.5).

Table 4.4.5 Example of data matrix indicating calculation of the number of standard deviation points away from the mean of each entry

	Site A	Site B	Site C	Site D	Site E	Mean	StDev
Morphospecies 1			(1.748-1.46)/0.58 = 0.497		(2.037-1.46)/0.58 = 0.995	1.46	0.58
Morphospecies 2	(0.602-0.6)/0.23 = 0.009			(0.903-0.6)/0.23 = 2.926	(0.699-0.6)/0.23 = 0.430	0.60	0.23
Morphospecies 3			(0.903-0.6)/0.43 = 0.705			0.60	0.43
Morphospecies 4				(0.301-0.08)/0.15 =1.473		0.08	0.15
Morphospecies 5			(3.374-3.32)/0.07 = 0.771		(3.372-3.32)/0.07 = 0.743	3.32	0.07

6. Add all entries for each site to calculate the value for the *BQI* (Table 4.4.6).

Table 4.4.6 Example of data matrix to calculating the biotope quality value

	Site A	Site B	Site C	Site D	Site E
Morphospecies 1			0.497		0.995
Morphospecies 2	0.009			2.926	0.430
Morphospecies 3			0.705		
Morphospecies 4				1.473	
Morphospecies 5			0.771		0.743
Environmental quality value	**0.009**	**0.000**	**1.973**	**4.399**	**2.168**

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Invertebrates are an ideal taxon for evaluating environmental quality due to their high diversity and global distribution (Samways, 2005). Furthermore, many species have high niche specialization and are thus sensitive to environmental change caused by human activity (Pearson & Cassola, 1992; Divictor *et al*., 2009). Most invertebrates, unlike rooted plants, have the ability to disperse to more optimal environments (Stohlgren *et al*., 1997). Owing to their size, invertebrates are easy and cost effective to trap and relatively safe to handle (Gardner *et al*., 2008). The large numbers of individuals needed for accurate statistical analysis can more easily be obtained and over a much smaller spatial scale than, for example, vertebrates.

Although we recommend the use of invertebrates, any living species can be substituted in place of the invertebrate examples used here, and even across taxonomic barriers, making this a general approach in impact assessments. However, it is important to gather data at the species level. Use of higher taxa would likely create a similar but less sensitive picture. This can be partly explained by the grouping of uneven numbers of species in the various higher taxa (some families have only one species, while others may have hundreds). Furthermore, the grouped values can be skewed by a single species contributing to the majority of counts which can produce a similar score as a grouped value with ample average values.

The exact methodology used for obtaining the data is also not prescribed. Importantly however, all data must be obtained using the same methodology, as for example, using pitfall traps for capturing ground dwelling invertebrates. Use of trapping methods that have set boundaries is recommended, rather than, for example, active searching which is subject to human concentration and effort (Ellis & Flaherty, 1992).

Futuristically, a *BQI* mean value for each species could be created when sufficient information on species population frequency is known, instead of using the mean of the species over a sampled range. Thus as knowledge of the various species increases, more specific values can be determined.

4.5   CONCLUSION

With ever-increasing pressure on natural resources, it is important to have reliable methods for evaluating the health of a specific geographical area in terms of its integrity and disturbance level. The *BQI* proposed here is not only a rapid and practical means of doing environmental evaluations, as well as an easily adaptable method across biotope types, taxa and methods of sampling.

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Chapter 5

COMPARISON OF THE BIOTOPE QUALITY INDEX WITH SURROGATED INDICES USED IN PRACTICE

ABSTRACT

Current environmental decline requires mechanisms to survey existing biological resources. Such surveys are essential for establishing legislation and other protocols for conserving biological resources. Species diversity and species richness are among the measures that have shortfalls for assessing biotope quality. Ecosystems, and their component structural biotopes, which are rich in species and microhabitats are generally healthier, in terms of resilience in the face of perturbation, than are their authropogenically disturbed and species-poor equivalents. I determine here the health of a range of biotopes, making use of a mean species specific assemblage. Species with above average abundance, derived from all sites, are identified for each site. The number of standard deviation units, above the mean average abundance of each morphospecies, is added up to calculate an Biotope Quality Index value for each biotope. The Biotope Quality Index is then compared to existing diversity indices, and found to be much more suitable for the assessment of ecosystem health.

5.1   INTRODUCTION

No comprehensive standard has been developed to assess biological condition or quality, or to define the nature and extent of biological degradation (Karr, 1992; 2004). Currently, ecosystems are being evaluated for heterogeneity, making use of various species surrogates such as species richness, relative abundance, diversity indices and phylogenetic indices, as well as environmental surrogates at the landscape level (Faith & Walker, 1996; Clarke & Warwick, 1998; Reyers *et al*., 2002; Magurran *et al*., 2010; Chapin *et al.,* 2011). These values are then used in management of ecosystems, where systems with high heterogeneity are considered more important than systems with low heterogeneity. Yet, quality of the biotopes at landscape level is rarely taken in consideration (Dennis *et al*., 2007).

Relative species abundance and species richness describe key elements of biodiversity (Hubbell, 2001). Species richness is the basic unit in which an ecosystem's homogeneity is assessed. The actual number of species present alone is largely an arbitrary number, but calculations for measuring the estimated species richness, for example the Chao index, exist and give a better idea of the actual number of species present within an biotope (Henderson, 2003).

Relative species abundance refers to how rare or common a species is relative to other species in a given biotope or community (McGill *et al*., 2007). Relative species abundance is described for a single trophic level where species will potentially compete for similar resources (Hubbell, 2001), similar to Tokeshi's (1990) theory, that approaches to evaluate species abundance distributions use available resources as the mechanism that drives relative abundance. The consistency of relative species abundance patterns suggests that some universal ecological rule or process determines the distribution of individuals among species within a trophic level (McGill *et al*., 2007). Tokeshi (1990) also points out that when species in the same trophic level consume the same limited resources, these limited resources will be divided among species to determine how many individuals of each species can exist in the community. Species with access to plentiful resources will have higher carrying capacities than those with limited access. In comparison, species diversity indices are statistics which are intended to measure the differences among individuals of a set consisting of various types of objects (Cover & Thomas, 1991).

These diversity indices put different weight on proportions of individuals, evenness, species richness or abundance. The most dominant diversity indices used in conservation ecology are Simpson-Yule index (Simpson, 1949), Shannon-Wiener function (Shannon, 1948), Berger-Parker dominance index (Henderson, 2003), McIntosh diversity measure (McIntosh, 1967) and the Brillouin index (Stilling, 1999). These forms of indices can provide estimates of the quality of the biotope by making assumptions that there is an underlying correlation between diversity and ecosystem health. I thus aim to test this concept, by comparing the correlations between the *BQI* and various established biodiversity indices relating to environmental integrity and disturbance levels.

5.2   METHODS

5.2.1   Study area and sampling methods

Thirty sites in the Jonkershoek Valley, Cape Floristic Region, South Africa (38°58' S; 18°55' E), were selected to represent six main biotope types, and an *a priori* range of disturbance levels from fully natural to disturbed. Sites were categorized in three natural fynbos vegetation biotope types: <0.5 m-high, Ericaceae-dominant fynbos (five sites), ±1 m-high Restionaceae-dominant fynbos (four sites) and >1.5 m-high Proteaceae-dominant Fynbos (five sites) were selected. In addition, a grass-invaded fynbos (five sites) and two exotic pine plantations categories: <10 yr old pine stands (six sites) and >10 yr old pine stands (five sites) were used (Figure 4.2.1).

Each site was evaluated according to integrity and disturbance levels, and rated from near-natural to least-natural, using mean values from the opinions of a group of Masters & PhD research students within Conservation Ecology and Entomology department of Stellenbosch University. This was achieved by making use of a rubric scoring table system. The disturbance level was then calculated taking in consideration, pollution (0 = no visual signs of pollution : 10 = rubbish dump), invasive aliens (0 = no invasive plant species present : 10 = 100% alien plants), distance from nearest disturbance (0 = more than 1000m from nearest road ; 10 = less than 5m from nearest road), human activity (0 = no visual signs of any human activity : 10 = building with constant human presence) and biotope structure (0 = signs of new growth and no unhealthy plants : 10 = diseased plants and no sign of natural new growth). This was a

subjective approximation of data (Table 4.3.1). This non-parametric method gave data against which I could compare the *BQI* with known biodiversity indices, in view of there being no current practical quantitive calculation for measuring ecosystem health, integrity or disturbance as a whole.

Using the method of Woodcock (2000), ten 350 ml pitfall traps were placed at random at each of the 30 sites for seven days in each of March, June, September and December 2006, to represent the four seasons. Ants (Formicidae) and springtails (Collembola) were the focal taxa for this part of the study, because they were the most species rich and abundant taxonomic groups sampled by the pitfall traps. All individuals were sorted to morphospecies.

5.2.2   Comparison of known diversity indices to the BQI

The *BQI* (identified in chapter 4, p.111-131) was compared by regression analysis, using of Microsoft Office Excel 2007 (O'Leary & O'Leary, 2008), against existing diversity indices that are commonly surrogated for evaluating ecosystem quality (Table 5.2.2.1).

5.3   RESULTS

The *BQI* correlated more strongly than any of the diversity indices to the site's level of disturbance ($R^2 = 0.833$). Species richness was the only index other than the *BQI* which showed strong positive correlation ($R^2 = 0.62$). Relative abundance ($R^2 = 0.278$), Shannon-Wiener function ($R^2 = 0.071$), McIntosh diversity index ($R^2 = 0.069$), Brillouin index ($R^2 = 0.062$), Berger-Parker dominance index ($R^2 = 0.021$), and Simpson-Yule index ($R^2 = 0.001$) showed weaker correlations than the *BQI* (Figure 5.3.1).

Table 5.2.2.1 List of existing formulas and diversity indices used as surrogates for evaluating ecosystem health

Name	Formula	References
Estimated Species richness (Chao index)	$\hat{S}_{max} = S_{obs} + (a^2 / 2b)$ S_{obs} = number of observed species a = number of species represented by 1 specimen b = number of species represented by 2 specimens	Henderson (2003) Southwood & Henderson (2000)
Relative abundance	n_i = number of individuals within each taxon	(Stilling, 1999)
Simpson-Yule index	$D = \dfrac{1}{C}$ if $C = \sum_{i}^{s_{obs}} p_i^2$ and $p_i^2 = \dfrac{N_i(N_i - 1)}{N_T(N_T - 1)}$ N_i = number of specimens in the i th species N_T = total individuals in the sample	(Simpson, 1949)
Shannon-Wiener function	$H = -\sum_{i=1}^{S_{obs}} p_i \log_e \cdot p$ p_i = proportion of sample in the i th species	Henderson, (2003) Shannon (1948) Weaver & Shannon (1949)
Berger-Parker dominance index	$d = \dfrac{N_{max}}{N_T}$ N_{max} = number of individuals in the sample belonging to the most abundant species N_T = total individuals in the sample	Henderson, (2003)
McIntosh diversity measure	$D = \dfrac{N - U}{N - \sqrt{N}}$ if $U = \sqrt{\sum n_i^2}$ N = total number of individuals in samples n_i = number of individuals in the i th sample	McIntosh (1967) Henderson, (2003)
Brillouin index	$H_B = \dfrac{\ln N! - \sum \ln n_i!}{N}$ N = total number of individuals in samples n_i = number of individuals in the i th sample	Stilling (1999)
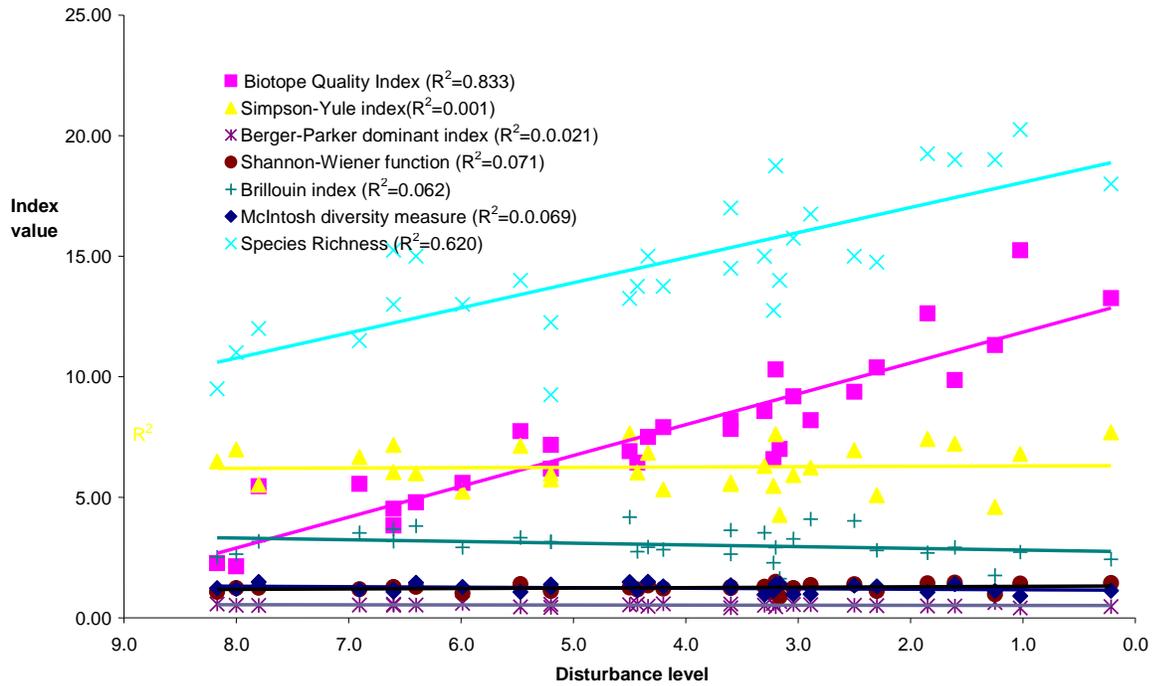
Figure 5.3.1 Comparison of the correlation between Biotope Quality Index (*BQI*) and diversity indices to the disturbance level calculated for 30 sites

5.4   DISCUSSION

The *BQI* correlated much more strongly with the level of disturbance than any of the diversity indices, mainly as the diversity indices do not measure ecosystem quality but quantitative species composition. Species richness is related to the size of the sample area, but is also affected by sampling intensity, making it relatively subjective. When a relatively small numbers of samples are taken from a given area, the number of species identified will be lower than if a greater number of samples is collected (Smith & Smith, 2008).   Furthermore, the surrogation of species richness for ecosystem health makes the assumption that in all biotopes with the best possible species richness, independent to geographical location or in which landscape they are found, the correlation will still be the same. However, this sameness has been found to be untrue, with a strong inverse correlation in many groups between species richness and latitude (Qian & Ricklefs, 2011).   Furthermore, as elevation increases, species richness decreases, indicating an effect of area, available energy, isolation and zonation (Smith & Smith, 2008).   Species richness does not take into consideration species evenness, whereas other indices do.

The Simpson-Yule index (Simpson, 1949), one of the most widely used indices, indicates a perfect homogeneous population if the index score is equal to zero, while a perfectly heterogeneous population would have an index score of one, assuming infinite categories with equal representation in each category (Henderson, 2003). Therefore, a community dominated by one or two species is considered to be less diverse than one in which several different species have a similar abundance. It is with these extreme values that Usher (1983) found discrepancies with the Simpson-Yule index, because a certain amount of confusion exists as to what the term "probability of a chance" means. A probability of zero, the smallest value that a probability can be, means that the event will never happen. In biological applications, generally there is a certain amount of rounding, and hence a probability written as zero in reality means a very small number approaching zero, and hence in words this would represent an event that was exceedingly unlikely to happen rather than an event that will never happen. Similarly, the maximum value that a probability can take is one, and indicates that an event must always happen. In practice, a probability of one actually means that a probability is approaching 1 and hence the event considered is exceeding unlikely not to happen (Usher, 1983). Furthermore,

Simpson-Yule's index has three variations, Simpson's Index **(D),** Simpson's Index of Diversity (1 – D) and Simpson's Reciprocal Index **(1 / D).** The Simpson-Yule index (D) gives an estimate of the probability that two individuals selected at random from the community will be of the same species, and as such it is easily to comprehend, but has the disadvantage of decreasing as diversity increases. To further complicate this situation there are two different versions of the formula for calculating Simpson-Yule index (D). The bigger the value of D, the lower the diversity, creating a problem in the logical interpretation, as the value is neither intuitive nor logical (Henderson, 2003).

Usher (1983) therefore suggested solving this problem by using the Simpson's index of Diversity (1-D) version because it provides the probability that two individuals drawn at random from the community will calculate to a total of one in the most diverse communities. Another way of overcoming the problem of the counter-intuitive nature of Simpson's index (D) is to take the reciprocal of the index (1/D). This index's lowest value is always one, and represents a community containing only a single species. The maximum value is the number of species in the sample (*n*) (Henderson, 2003). Another problem with the Simpson-Yule index is that it gives more weight to the more abundant species in a sample. The addition of rare species to a sample causes only small changes in the value of D (Henderson, 2003).

In comparison, the Shannon-Wiener function describes the uncertainty of predicting the species of a randomly chosen individual from the community (Weaver & Shannon, 1949). Heister (1972) pointed out that the index does not always adequately reflect problems with the number of individuals collected when they are evenly distributed and it provides no specifics on the causes of perturbation. One of the diversity indices with the most praise, because of its simplicity of calculation is the Berger-Parker dominance index. The Berger-Parker dominance index only takes into consideration the proportion of the abundance of the dominant species compared to the total abundance. It can be a good indication of invasive species, but in biotopes that naturally are dominated by a single species, it will indicate low diversity. If this is used as a surrogate for environment health, a healthy biotope can be misinterpreted as unhealthy (Henderson, 2003).

The McIntosh diversity measure as with most diversity indices is strongly influenced by sample size (Henderson, 2003). In contrast, the Brillouin index, which is useful when the randomness of a sample is not guaranteed, makes the assumption that the community is completely sampled (Henderson, 2003). In practical ecology, this will never be the case, as sampling methods always have a focal group of guilds or taxa they use as a surrogates for the complete sample.

Acar and Troutt (2008) identified three deficiencies with existing diversity indices, causing insufficient discrimination among related but dissimilar situations. Firstly, they lack normalization, which is corrected for in the BQI by each species being correlated with the mean population size of that same morphospecies. Secondly, diversity indices lack calibration, corrected for in the BQI by using the same scale in the form of the same morphospecies and methodology used at all sites sampled. Finally, most indices are non-linear while the BQI shows results on a linear scale making it suitable for comparison of sites. Furthermore, both the number of morphospecies, as well as the number of replicates of sites used, will determine the quality of the fit to the real situation, meaning that more is better (Thomson *et al*., 2007).

5.5   CONCLUSION

This study thus indicate that the theoretical *BQI,* shows that it is more suitable than any of the familiar diversity indices currently being used by environmental impact assessors to evaluate ecosystem health.

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Chapter 6

POSSIBLE WAYS TO ENHANCE THE BIOTOPE QUALITY INDEX BY CONCENTRATING THE DATA SET

ABSTRACT

A guild is defined as a group of species that exploit the same class of environmental resources in a similar way and should exhibit similar ecologies. A guild groups together species without regard to taxonomic position, yet which overlap significantly in their niche requirements. Guilds are useful for providing certain generalizations that would be missed by taxonomic studies alone. This chapter compares the Biotope Quality Index values for the guilds, dominantly herbivore (Orthoptera), predators (Araneae), omnivores/ scavengers (Formicidae) and decomposers (Collembola) with that of a broad-spectrum taxon representing all guilds (Coleoptera). I found that the omnivore/scavenger (Formicidae) and decomposer (Collembola) guilds are the best suited bioindicators in most conditions, when using the Biotope Quality Index (*BQI*). These results are confirmed with the IndVal test done on the same taxa. Lastly, I discuss the selection of taxa to create a short list of bioindicators best suited for the fynbos biotopes under study here, outside of guild preference, and to be used within the *BQI*. Similar testing of morphospecies present within the biotope being evaluated is recommended. This allows for selection of only highly positive correlating bioindicators to improve the quality of the results from the *BQI*.

6.1   INTRODUCTION

An environmental impact assessment mainly looks at the environment as a whole, the diversity, interactions between species, but mostly at the function of the environment at a biotope scale to determine the state of its biological integrity (Jørgensen & Fath, 2008). The Biotope Quality Index (*BQI*) uses this concept to look at the invertebrate assemblages, and to determine each biotope's ability to sustain an above average population. The question thus arises; will the trends seen from the *BQI* be similar throughout the various guilds?

According to Simberloff & Dayan (1991) a guild is defined as a group of species which exploits the same class of environmental resources in a similar way, and thus, according to Speight *et al*. (1999) should exhibit similar ecologies.   A guild groups together species without regard to taxonomic position, and which overlap significantly in their niche requirements (Simberloff & Dayan, 1991). Although taxonomic position is not important for defining a guild, in ecology, many guilds are composed of groups of closely related species that all arose from a common ancestor and they most likely exploit resources in similar ways as a result of their shared ancestry and evolutionary biogeography (Flannery & Thompson, 2007). Guilds are useful for providing generalizations that would be missed by taxonomic studies alone. For example how certain insects will react under unfavourable conditions (Speight *et al*., 1999).

Terrestrial arthropod ecological guilds include the herbivore, fungivore, carnivore, omnivore and decomposer guilds, depending on the type of food (plant, fungal or animal) and the status of their food (dead, recently deceased or alive) (Speight *et al*., 1999).

The herbivore guild comprises arthropods that feed on living plant material and can be sub-divided into various smaller guilds depending on the part of the plant on which they feed on or the food source itself (Peeters *et al.,* 2001).   These include for the terrestrial habitat, gall-formers (Hemiptera, Diptera, Hymenoptera), sap suckers or saprovores (Acari, Hemiptera), grazers, leaf-chewers or leaf-minners (Orthoptera, Phasmatodea, Psocoptera, Coleoptera, Lepidoptera), root- or stemfeeders (Isopoda, Diplopoda, Collembola, Hemiptera, Diptera, Coleoptera), borers (Isoptera, Coleoptera, Hymenoptera), seed feeders (Coleoptera), frugivores (Thysanoptera, Lepidoptera, Diptera), and nectar feeders (Lepidoptera, Diptera, Hymenoptera) (Speight *et al*., 1999).

Grasshoppers (Acrididae, Orthoptera) one of the most widely used herbivore bioindicators (Bazelet & Samways, 2011). They tend to respond well to changes in savanna (Prendini *et al*., 1996; Samways *et al*., 2010) especially in grassland ecosystems, and are also used in grass rangeland management (Quinn *et al*., 1993), as well as the effect of overgrazing (Gibson *et al*., 1992). Grasshoppers have also been used for impact assessment of disturbances in dry deciduous forest (Saha & Haldar, 2009) and tropical savannas (Andersen *et al*., 2001).

Members of the fungivore guild consist of species that primarily or solely feed on living fungi (mycophages) (Acari, Collembola, Coleoptera, Diptera) (Carlile *et al*., 2001). There is no work on any arthropod fungivores that feed only on fungi being used as bioindicators, although reference has been made to the potential of the Phylum Tardigrada for detecting air pollution (Ramazzotti & Maucii, 1994).

The terrestrial predatory guild comprises species which kill and eat other organisms. They can be subdivided in smaller guilds based on the way they kill and feed on their prey: constriction or crusher killers consume their prey whole or in chunks (Araneae, Solphugidae, Uropugi, Amblypygi, Scorpiones, Odonata, Mantodea, Orthoptera, Dermaptera, Coleoptera, Megaloptera, Neuroptera, Hymenoptera); sucking killers only feed on the body fluids of the prey after killing it (Hemiptera, Mecoptera, Neuroptera), ectoparasites that do not kill their prey but feed on the body fluids (Acari, Pseudoscorpiones, Phthiraptera, Siphonaptera, Diptera), poison killers (Araneae, Opliones, Chilopoda, Hymenoptera); artificial trappers, using something like a web to catch prey (Araneae, Embioptera, Neuroptera); parasitoids that imbed themselves within their prey for some part of their life cycle (Diptera, Strepsiptera, Hymenoptera) (Molles, 2002).

Araneae have been used in various indicator studies, including those on environmental management for habitat restoration (Cardoso *et al*., 2004; Gollan *et al*., 2010), evaluation of biodiversity (Gollan *et al.,* 2010), habitat transformation (Perner & Malt, 2003), microclimatic continentality and soil compactness (Rezac *et al*., 2007), conservation value of peat (Scott, 2006) and general grassland management (Pozzi *et al*., 1998). In the agricultural environment, Araneae have been used for monitoring landscape and habitat features (Jeanneret *et al*., 2003), effect of herbicides (Haughton *et al*., 2003) as well as in the management of the push-pull habitat management (Midega *et al.,* 2008). Araneae have also been used as indicators of the effect of environmental pollution (Seyyar *et al*., 2010),

specifically heavy metal pollution (Jung *et al.*, 2008), as well as rainforest fragmentation (Kapoor, 2008), urbanization (Magura *et al.*, 2010), overgrazing (Horvath *et al.*, 2009), tillage intensity (Rodriguez *et al.*, 2006) and mowing (Cattin *et al.*, 2003).

The omnivore guild consists of species that are opportunistic, generalist feeders that consume both plant and animal material as their primary food. The omnivore guild can be subdivided depending on where they consume their food: scavengers, that feed where they find their food (Isopoda, Amphipoda, Orthoptera, Blattodea, Diptera, Coleoptera); and foragers or collectors, that collect material for themselves or to collect to feed on at a later stage (Hymenoptera) (Singer & Bernays, 2003).

Ants (Formicidae, Hymenoptera) are the omnivorous foragers that have often been promoted as bioindicators (Alonso, 2000; Kasperi & Majer, 2000). They are frequently used in agroecosystems to test for contamination from pesticides or herbicides (Pereira *et al.*, 2010), as well as for environmental management in restoration studies (Majer, 1983; Dekoninck *et al.,* 2008; Delabie *et al.*, 2009; Coelho *et al.*, 2009; Paolucci *et al.*, 2010).

Decomposers or ditrivores feed on dead organic matter and are in most cases are subdivided according to the size of the particles on which they feed or the origin of the organic matter: plant material (Acari, Collembola, Thysanura, Archeognatha, Psocoptera, Diptera, Coleoptera, Hymenoptera); carrion (Coleoptera, Diptera) or both (Amphipoda, Isopoda, Diptera, Coleoptera, Hymenoptera) (Swift *et al.*, 1979).

Of the decomposers, Collembola are the best known as indicators for testing toxin contamination in the soil (Pramanik *et al.,* 1998; Chagnon *et al.*, 2001; Lins *et al.*, 2007; Szeder *et al.*, 2008; Gardi *et al.*, 2008; Souza & Fontanetti, 2011) as well as the effect of tillage (Tabaglio *et al.*, 2009). In conservation, Collembola have been used for ecosystem monitoring, especially soil management (Kopeszki, 1992; van Straalen & Verhoef, 1997; van Straalen, 1998; Geissen & Kampichler, 2004; Timmermans *et al.*, 2007; Baretta *et al.*, 2008; Owojori *et al.,* 2009; O'Neill *et al.*, 2010) and for monitoring the effect of various pollutants (Son *et al.,* 2007; Greenslade, 2007; Natal-da-Luz *et al.*, 2008; Fiera, 2009; Xu *et al.*, 2009a, 2009b; Barbercheck *et al.*, 2009; Barros *et al.*, 2010; Uribe-Hernandez *et al.*, 2010).

The Biotope Quality Index (*BQI*) (Chapter 4, p.111-131) makes use of the whole target taxon assemblage. However, the *BQI* could be simplified by making use of guilds or

smaller groups of specific taxa. The question then becomes: will the *BQI* calculated on a specific guild be a surrogate for the *BQI* calculated for the whole arthropod assemblage?

Samways *et al*. (2010) provide a stepwise decision making framework for the compilation of a set of taxonomically diverse indicator species, based on Hilty and Merelender (2000). In the first step, a decision is made as to which ecosystem attributes the indicator taxa should reflect. This involves deciding on which biotope attributes of the goal ecosystem are suitable for a variety of specialist taxa or guilds. Secondly, a list all species or taxonomic groups is made that meet the basic information criteria, meaning that the taxonomy should be clear, biology and life history well known, and the species distribution known, but mostly their tolerance levels to environmental pressures also be known, especially in relation to specific changes in environmental conditions. Samways *et al*. (2010) also mention that only fairly common and easily detectable species, which are evenly distributed in the biotope, should be used. The third step includes listing available information on niche and life history, as well as sensitivity to environmental stressors, niche and life history criteria in relation to trophic level, reaction time to environmental changes, mobility, minimum area requirements, detailed niche information, fine structural, but essential, aspects of the biotope, and sensitivity to different environmental stressors. It is then from these three steps that a final list is compiled of complementary species, taxa or guilds to satisfy every criterion (Samways *et al*., 2009).

Furthermore, McGeoch (1998) points out that many studies focus on the identification of taxa as potentially suitable indicators but very few have been formally tested as indicators. This crucial second step is the focus of this study, which aims to identify the best possible taxa or guilds that are suitable for use within the *BQI*.

6.2   METHODS

6.2.1   *Study area*

Thirty sites in the Jonkershoek Valley, in the heart of the Cape Floristic Region, South Africa (38°58' S; 18°55' E), were selected to represent six main biotope types, and an *a priori* range of disturbance levels from fully natural to disturbed. Sites were categorized into three natural fynbos vegetation biotope types: <0.5 m-high, Ericaceae-dominant fynbos (five sites), ±1 m-high Restionaceae-dominant fynbos (four sites) and >1.5 m-high

Proteaceae-dominant Fynbos (five sites) were selected. In addition, a grass-invaded fynbos (five sites) and two exotic pine plantations categories: <10 yr old pine stands (six sites) and >10 yr old pine stands (five sites) were used (Figure 4.2.1).

6.2.2   Sampling method

Each site was evaluated according to integrity and disturbance levels, and rated from near-natural to least-natural, using mean values from the opinions of a group of Masters and PhD research students within Conservation Ecology and Entomology department of Stellenbosch University. This was achieved by making use of a rubric scoring table system. The disturbance level was then calculated taking in consideration, pollution (0 = no visual signs of pollution : 10 = rubbish dump), invasive aliens (0 = no invasive plant species present : 10 = 100% alien plants), distance from nearest disturbance (0 = more than 1000m from nearest road ; 10 = less than 5m from nearest road), human activity (0 = no visual signs of any human activity : 10 = building with constant human presence) and biotope structure (0= signs of new growth and no unhealthy plants : 10 = diseased plants and no sign of natural new growth). This was a subjective approximation of data. This non-parametric method provide a spectrum whereby we can correlate the objective Biotope Quality Index (*BQI*) for each guild to the level of disturbance. Ten 350 ml pitfall traps (Woodcock, 2000) were placed at random at each of the 30 sites for seven days in each of March, June, September and December 2006, to represent the four seasons.

6.2.3   Calculation of Biotope Quality Index (BQI) with respect to arthropod guilds

Ants (Formicidae), springtails (Collembola), spiders (Araneae), grasshoppers (Orthoptera) representing the forager, decomposers, predators and herbivores guilds respectively, were selected as the focal taxa, as they were species rich and abundant groups found within the pitfalls samples, as well as being recognized in the literature as good indicator taxa (Chapter 3, p.45-108). These taxa were then compared with a broad-spectrum taxon (Coleoptera) that consists of all the guilds as a control group. Individuals were sorted to morphospecies.

The *BQI* was calculated for of each of the guilds i.e., foragers (Formicidae), decomposers (Collembola), predators (Araneae), herbivores (Orthoptera), control group

(Coleoptera), making use of the formula $BQI = \sum_{i=1}^{n}(\frac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i})$ when $\frac{\log x_{ij} - \overline{\log x_i}}{\log \sigma_i} > 0$

Furthermore, the *BQI* values were calculated for each taxon, to family or genera level. These values were then compared by regression correlation with the disturbance level done with Microsoft Excel 2007 (Kelly & Simmons, 2007) for each guild as well as for each taxon. This was done for the pooled data set, as well as for data separately from pristine fynbos sites versus the disturbed fynbos sites. For the guild comparisons, sites where further catogorized into Protea-, Restio-, or Erica-dominant fynbos sites compared to Poaceae invaded sites and Young pine plantations (trees <10 year old) and Old pine plantations (tree >10 years old). Positive as well as negative regressions were identified for each morphospecies but only those with significant correlation were tabulated.

6.2.4   *Calculation of IndVal (Indicator Value) for each morphospecies*

The taxa showing bioindicator potential were identified by calculating the *IndVal* values for each taxon. For each species *i* in each site group *j*, the product of A_{ij} was computed, which is the mean abundance of species *i* in the sites of group *j* compared to all groups in the study, by B_{ij}, which is the relative frequency of occurrence of species *i* in the sites of group *j*, as follows:

A_{ij} = Nindividuals$_{ij}$/ Nindividuals$_i$

B_{ij} = Nsites$_{ij}$/Nsites$_j$

IndVal $_{ij}$ = A_{ij} x B_{ij} x 100

where *IndVal* is the indicator value of species *i* in site cluster *j* (Dufrêne & Legendre, 1997).

6.3   RESULTS

6.3.1   *Comparison between different guilds using the BQI*

The *BQI* value calculated for ants as representative of the foraging guild showed significantly higher percentage of correlation (71.25%) than any of the other guilds for the pooled sites. The broad-spectrum beetle group as well as the predatory spider guild showed negative correlation to the level of disturbance (Figure 6.3.1.1).
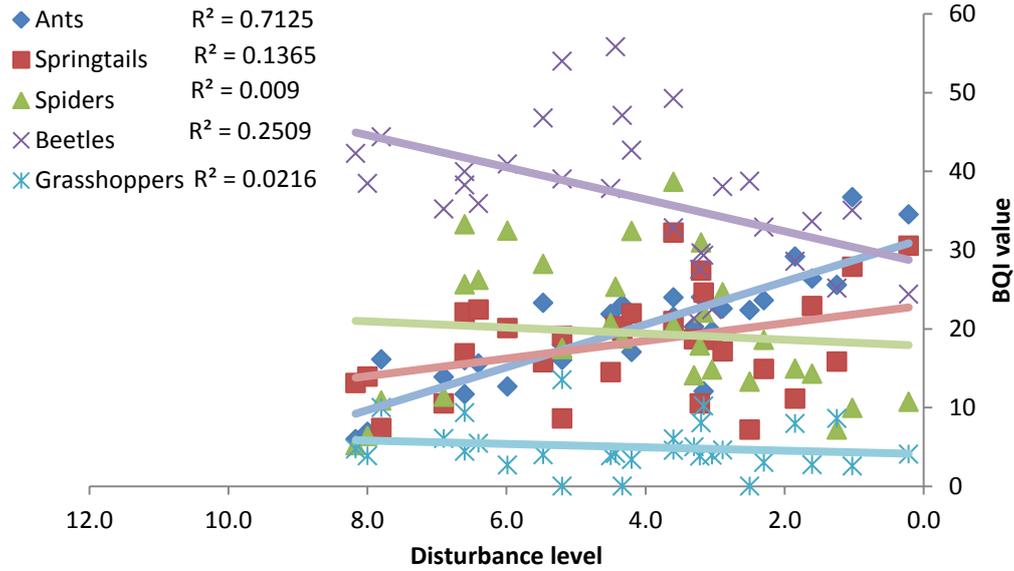
Figure 6.3.1.1 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for

each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore –

Ants; Detritivore – Springtails; Control – Beetles) for all sites sampled
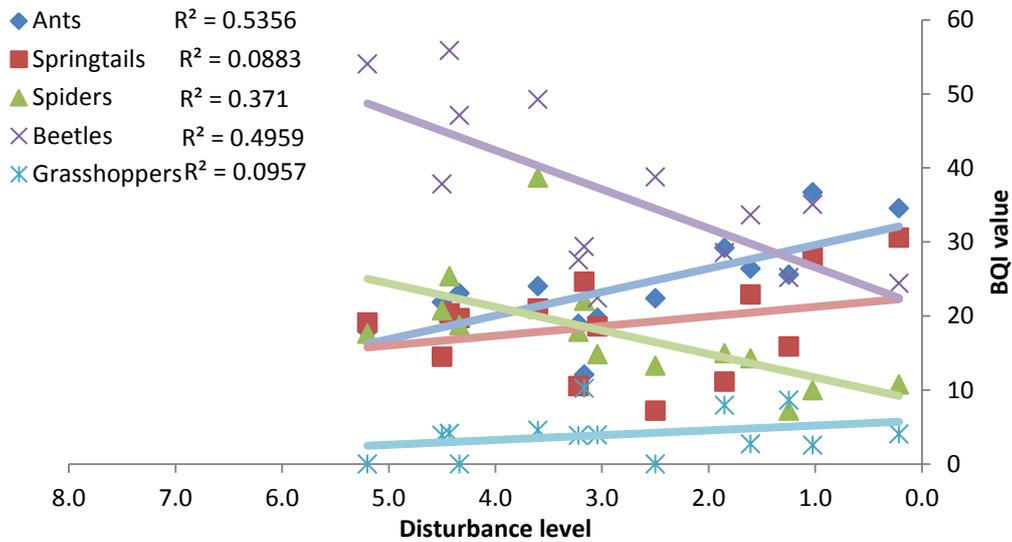
Figure 6.3.1.2 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for

each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore –

Ants; Detritivore – Springtails; Control – Beetles) for all non-disturbed fynbos sites

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A similar trend can be seen if we look only at the pristine non-disturbed fynbos sites. Ants had a correlation of 53.56%, and both beetles and spiders again showed negative correlation to the disturbance level (Figure 6.3.1.2). Ants also showed the best correlation (71.85%) when considering only the disturbed fynbos sites. Interestingly, spiders did not show a negative correlation (Figure 6.3.1.3).

When dividing the categories further, in the Protea-dominant fynbos the crickets and grasshoppers were the indicator guild that showed the strongest correlation (59.43%). Beetles were the only guild that showed negative correlation with disturbance level (Figure 6.3.1.4).

In the Restio-dominated fynbos, springtails have the highest coefficients of determination with the disturbance level (64.71%), while ants showed a negative correlation (Figure 6.3.1.5). In the Erica-dominated fynbos the strongest positive correlations were with springtails (86.07%) and ants (81.25%), while beetles (91.28%) and spiders (84.15%) showed very strong negative correlation to the disturbance level (Figure 6.3.1.6).

The Poaceae invaded fynbos sites showed a stronger correlation for the predator guild (spiders: 81.05%), while ants are the only taxon which showed a negative correlation (Figure 6.3.1.7).

In the young pine plantations (<10 year old trees), where remnants of the fynbos community were still present, there was no significant positive correlation, but both springtails and ants showed negative correlations (Figure 6.3.1.8). With age, the older pine plantations, suppressed the fynbos undergrowth, and I found strong negative correlations for the *BQI* values of both spiders (88.09%) as well as grasshoppers (62.72%) (Figure 6.3.1.9).

6.3.2   *IndVal values*

Table 6.3.2.1 shows the potential of various taxa as bioindicators in both undisturbed and disturbed fynbos. The highest mean *IndVal* value between the guilds indicated that the decomposers (springtails: 12.24) and scavengers (ants: 11.65) are the best potential bioindicators for undisturbed fynbos. Springtails were the only taxon that showed a high IndVal mean in the disturbed fynbos sites (11.44).
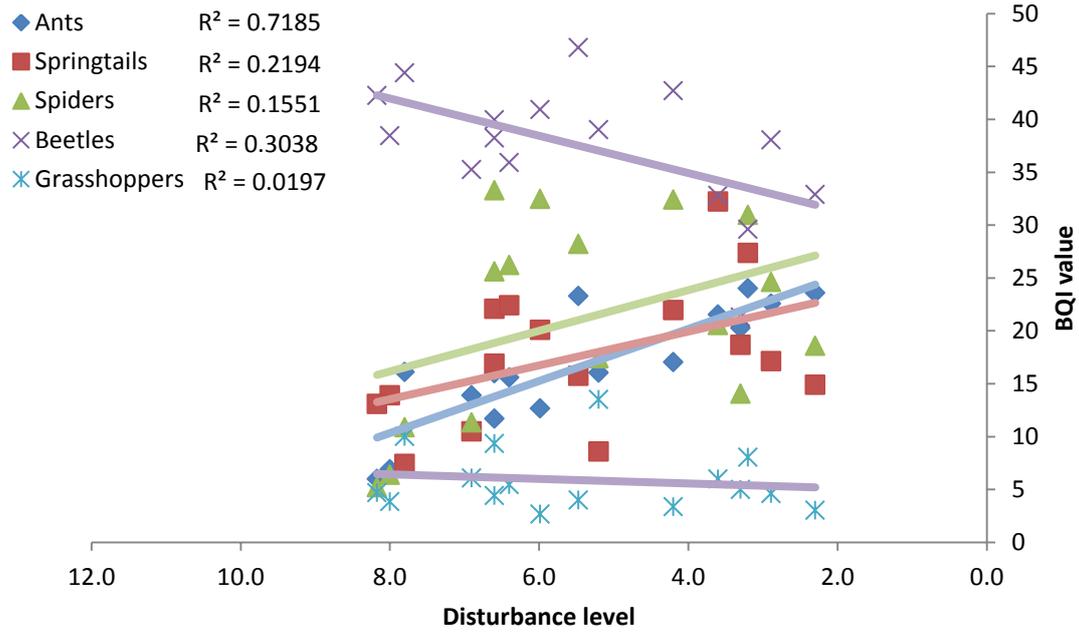
Figure 6.3.1.3 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for

each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore –

Ants; Detritivore – Springtails; Control – Beetles) for all disturbed fynbos sites
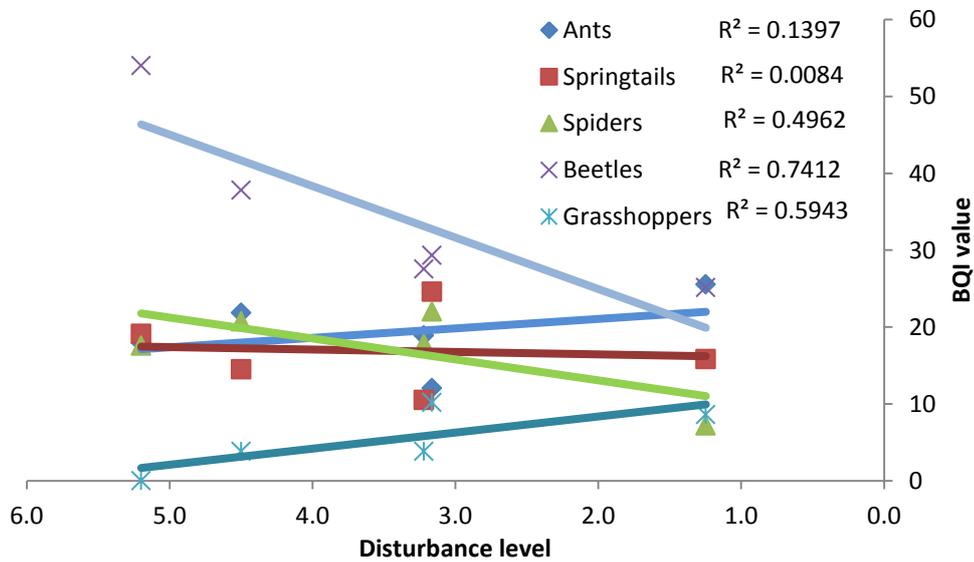
Figure 6.3.1.4 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for

each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore –

Ants; Detritivore – Springtails; Control – Beetles) for all Protea dominated fynbos sites

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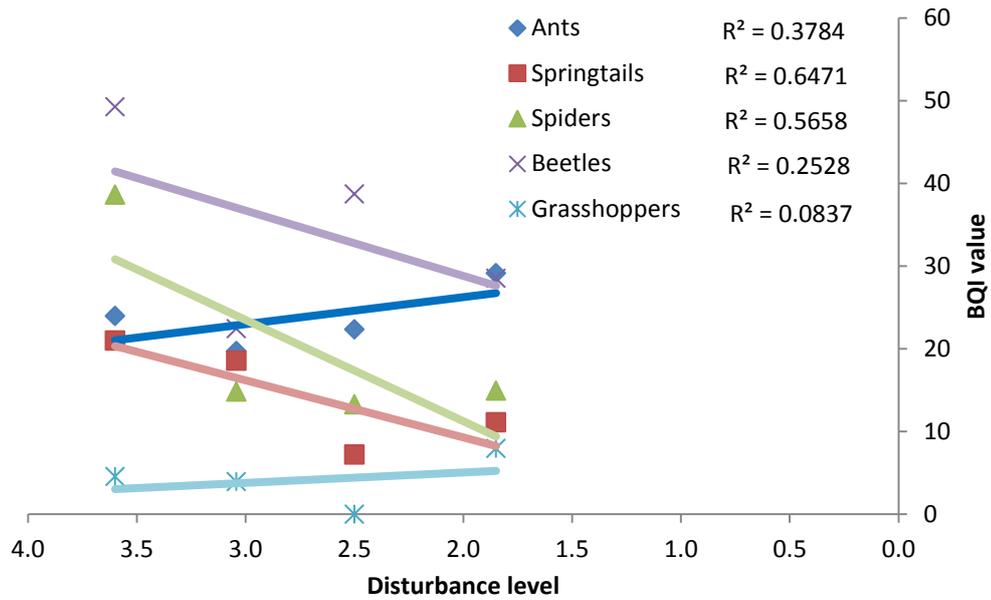
Figure 6.3.1.5 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore – Ants; Detritivore – Springtails; Control – Beetles) for all Restio dominated fynbos sites
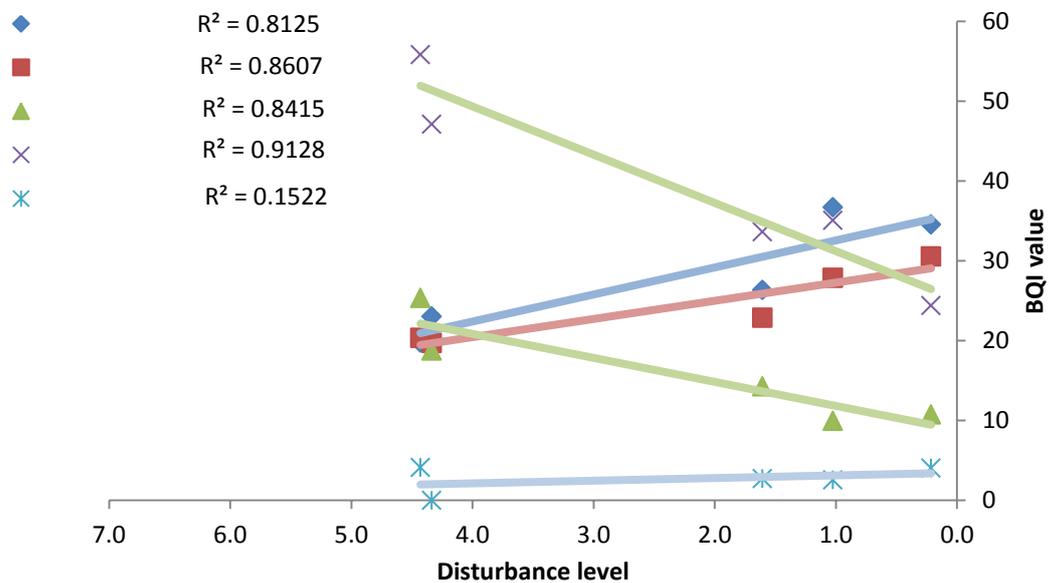
Figure 6.3.1.6 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore – Ants; Detritivore – Springtails; Control – Beetles) for all Erica dominated fynbos sites
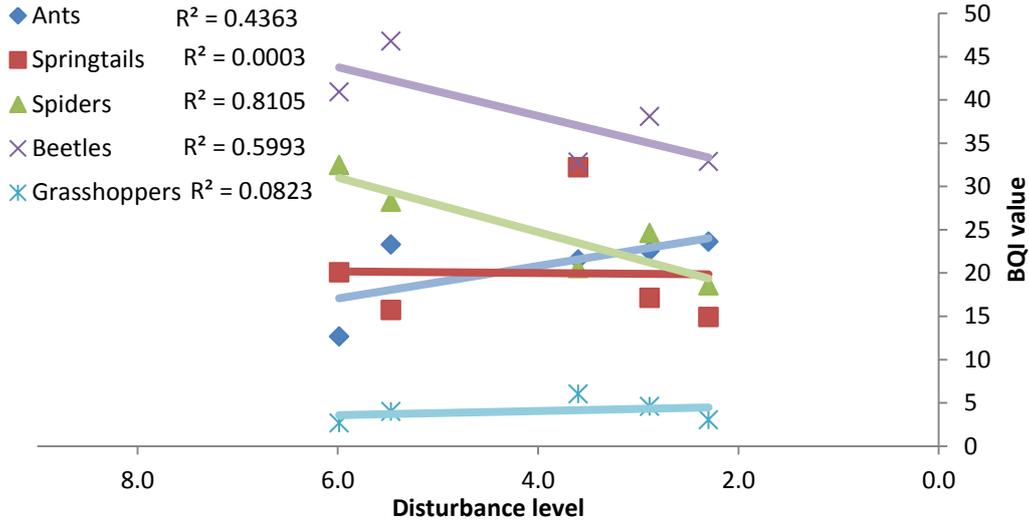
Figure 6.3.1.7 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated

separately for each representative of various feeding guilds (Carnivore – Spiders; Herbivore –

Grasshoppers; Omnivore – Ants; Detritivore – Springtails; Control – Beetles) for all Poacea invaded

fynbos sites
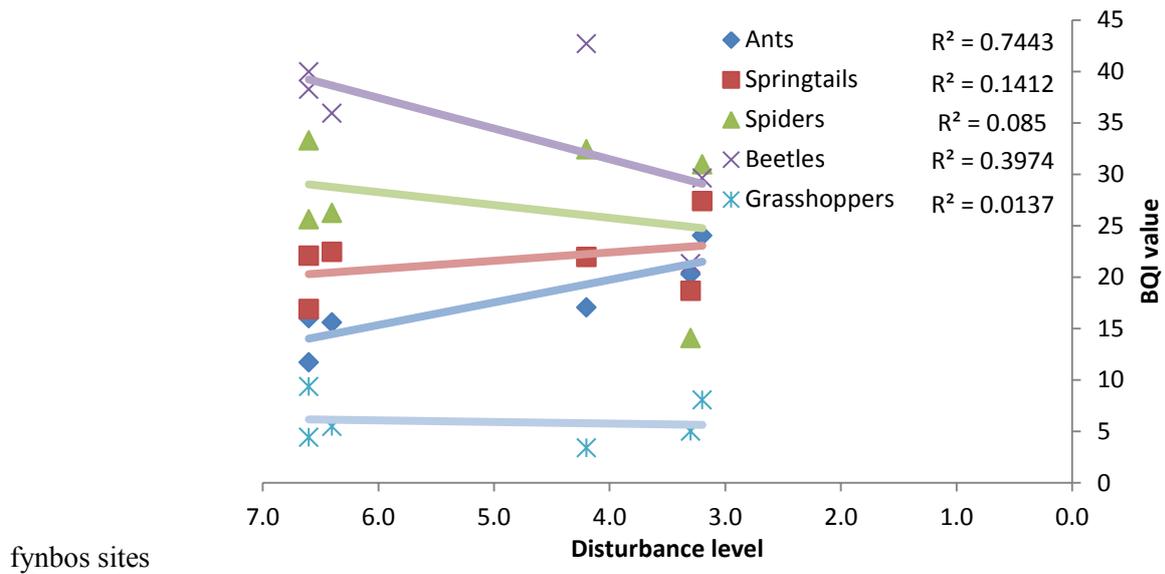
Figure 6.3.1.8 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated

separately for each representative of various feeding guilds (Carnivore – Spiders; Herbivore –

Grasshoppers; Omnivore – Ants; Detritivore – Springtails; Control – Beetles) for all young pine

forest sites

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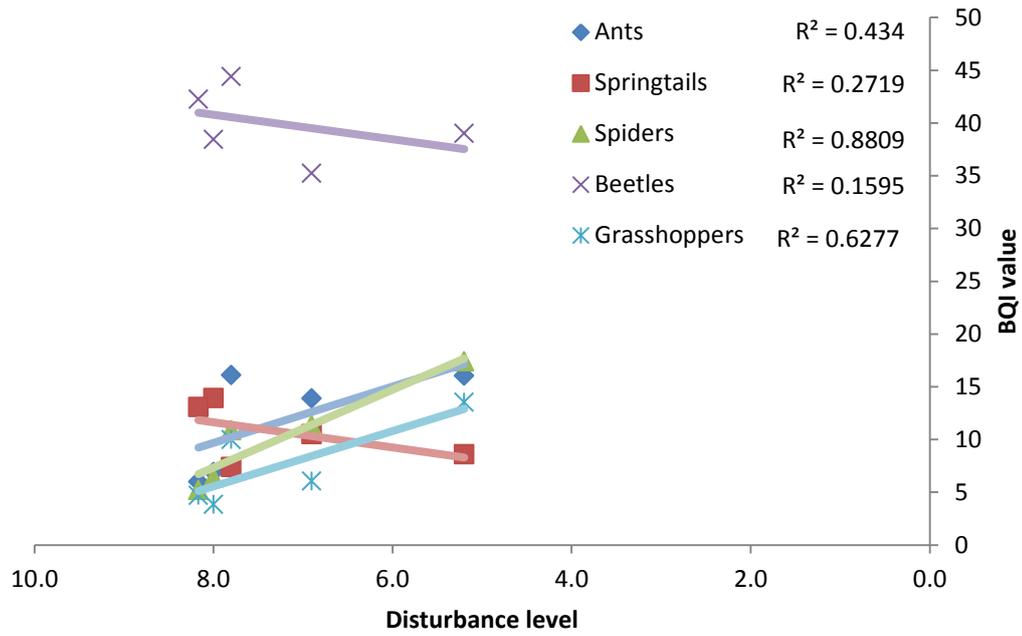
Figure 6.3.1.9 Differentiation between trends of Biotope Quality Index (*BQI*) values calculated separately for

each representative of various feeding guilds (Carnivore – Spiders; Herbivore – Grasshoppers; Omnivore –

Ants; Detritivore – Springtails; Control – Beetles) for all matured pine forest sites sampled in 2006 from

Jonkershoek, South Africa

Analysis of the various ant taxa in the undisturbed fynbos sites, showed the genera *Lepisiota* (26.8), *Solenopsis* (20.9) and *Camponotus* (20.04) as potential bioindicators. The springtail families, Poduridae (19.38) and Entomobryidae (16.56) also showed potential, while beetles when pooled were not good bioindicators, although some of the smaller rarer families, Silvanidae (19.33), Cucujidae (18.67) and Phalacridae (17.4), appeared to have potential.

In the disturbed fynbos sites, various other families and genera also showed potential as bioindicators. The ant genera *Plagiolepis* (23.24) and *Meranoplus* (21.75) showed most potential. The beetle families Corylophidae (42.86), Scraptiidae (25.56), Trogidae (21.74) and Bostricidae (21.23) also had high *IndVal* values.

6.3.3 *Taxa that indicate correlation between their* BQI *value and level of disturbance*

Table 6.3.3.1 shows that for the *BQI* values for the beetle families Curculionidae (75.8%), Cucujidae (73,5%) and Scarabaeidae (60.9%), the springtail family Onychiuiridae (60.6%), and the orthopteran family Tetrigidae (56.83%) all show strong positive correlation with disturbance level. Strong negative correlations that stand out can be seen in the ant genus *Dorylus* (100%), beetle families Chrysomelidae (100%), Ptiliidae (75%) and Histeridae (64%), and spider families Ctenizidae (78.7%) and Dictynidae (63.6%).

Table 6.3.3.2 illustrates only those taxa that show potential as bioindicators for pristine fynbos, with the orthopteran family Tetrigidae and the ant genus *Technomyrmex* both showing 100% positive correlation to disturbance level. The beetle families Chrysomelidae (100%), Cleridae (100%) and Curculionidae (81.7%) showed the highest negative correlation, with the springtail family Dictyrtomidae (85.9%) and spider family Ctenizidae (97.5%) also showing strong negative correlation.

In the disturbed fynbos and pine plantations (Table 6.3.3.3), there were positive correlations with disturbance level by the spider family Selenopidae (100%), beetle families Chrysomelidae (100%) and Curculionidae (82.8%), as well as the ant genus *Camponotus* (83.1%). The spider families Salticidae, Ctenizidae, the orthopteran family Stenopelmatidae, beetle family Anthicidae, Ptiliidae and the ant genus *Dorylus* all showed 100% negative correlation to disturbance levels within the disturbed biotopes.

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Table 6.3.2.1 Taxa having bioindicator potential through high *IndVal* values for both undisturbed and disturbed fynbos sites

Fynbos (Undisturbed)				Fynbos (Disturbed)			
	n	Mean	Maximum		n	Mean	Maximum
Araneae	191	4.19	26.49	Araneae	189	3.84	38.00
Agelenidae	43	4.88	26.49	Dictynidae	33	4.78	38.02
Dysderidae	8	7.24	24.65	Agelenidae	43	6.12	32.40
Salticidae	27	5.05	23.33	Pisauridae	19	4.00	27.58
Selenopidae	20	3.74	22.22	Salticidae	27	3.38	21.00
Scytodidae	4	9.07	16.67	Dysderidae	8	5.00	18.69
Collembola	27	12.24	34.93	Collembola	29	11.44	31.16
Entomobryidae	12	16.56	34.93	Smithuridae	5	13.72	31.16
Sminthuridae	5	10.80	20.00	Entomobryidae	12	13.41	25.59
Poduridae	1	19.38	19.38	Dicyrtomidae	5	9.91	17.89
Orthoptera	42	4.93	40.54	Orthoptera	42	2.83	23.33
Gryllidae	11	12.36	40.54	Gryllidae	11	5.38	23.33
Coleoptera	235	5.17	37.23	Coleoptera	235	5.36	42.86
Carabidae	31	5.15	37.23	Corylophidae	1	42.86	42.86
Bostrychidae	4	9.23	34.02	Scraptiidae	2	25.56	41.89
Phalacridae	3	17.40	31.31	Bostrichidae	4	21.23	39.76
Silvanidae	3	19.33	29.33	Staphylinidae	16	10.37	39.07
Mordelidae	6	6.39	23.40	Nitidulidae	8	8.83	30.64
Histeridae	9	4.66	22.52	Carabidae	31	4.13	30.31
Ceratocanthidae	7	8.63	22.47	Mordelidae	6	9.56	28.79
Scarabaeidae	27	5.36	22.07	Scydmaenidae	6	12.87	27.81
Staphylinidae	16	5.08	20.70	Ceratocanthidae	7	3.93	27.39
Cucujidae	1	18.67	18.67	Scarabaeidae	27	3.10	22.32
Apionidae	3	6.94	17.50	Silvanidae	3	10.22	22.00
Scydmaenidae	6	10.54	17.11	Trogidae	1	21.74	21.74
Chrysomelidae	30	2.44	16.67	Tenebrionidae	6	6.22	16.97
				Histeridae	9	5.98	16.72
Formicidae	41	11.65	29.94	Phalacridae	2	8.82	10.97
Camponotus	6	20.04	29.94				
Lepisiota	1	26.80	26.80				
Monomorium	3	14.60	23.55	Formicidae	41	7.50	31.16
Pheidole	2	19.28	22.17	*Linepithema*	1	17.73	17.73
Solenopsis	1	20.90	20.90	*Meranoplus*	1	21.75	21.75
Tetramorium	7	13.68	20.38	*Monomorium*	3	15.38	26.45
Linephithema	1	19.37	19.37	*Plagiolepis*	1	23.26	23.26
Crematogaster	6	17.52	17.52	*Tetramorium*	6	18.22	31.16

Table 6.3.3.1 Taxa that correlated with biotope quality from the pooled data of all 30 sites, with reference to the number of morphospecies used within the taxon (n), the mean square of Pearson's product moment correlation coefficient in percentage (R^2), as well as the maximum percentage correlation for all morphospecies

Positive correlations				Negative correlations			
Taxa	n	Mean R^2	Max	Taxa	n	Mean R^2	Max
Araneae				**Araneae**			
Selenopidae	2	50.03	100.00	Ctenizidae	2	78.67	100.00
Agelenidae	11	44.85	100.00	Dictynidae	6	63.63	100.00
				Dysderidae	3	47.93	100.00
Collembola				Salticidae	4	31.19	81.90
Onychiuridae	1	60.59	60.59	Pisauridae	3	22.39	66.90
Sminthuridae	5	36.11	98.92				
Dicyrtomidae	5	36.11	80.50	**Collembola**		(none)	
Entomobryidae	10	29.47	100.00				
				Orthoptera			
Orthoptera				Stenopelmatidae	2	50.39	99.54
Tetrigidae	2	56.83	100.00	Gryllidae	7	21.65	69.96
Coleoptera				**Coleoptera**			
Curculionidae	7	75.80	100.00	Chrysomelidae	3	100.00	100.00
Cucujidae	1	73.49	73.49	Ptiliidae	1	75.00	75.00
Scarabeaidae	6	60.90	100.00	Histeridae	3	64.07	96.23
Mordelidae	4	50.23	100.00	Carabidae	11	56.77	100.00
Scraptiidae	2	45.70	70.92	Anthicidae	2	50.88	100.00
				Bostrychidae	4	35.28	100.00
Formicidae				Silvanidae	2	31.63	62.31
Technomyrmex	2	42.97	79.18	Staphylinidae	8	18.65	100.00
Camponotus	5	41.97	66.68				
Tetramorium	7	22.99	100.00	**Formicidae**			
				Dorylus	1	100.00	100.00

Table 6.3.3.2 Taxa that correlated with biotpe quality from the pristine fynbos sites, with reference to the number of morphospecies used within the taxa (n), the mean square of Pearson's product moment correlation coefficient in percentage (R^2), as well as the maximum percentage correlation for all morphospecies

Positive correlations				Negative correlations			
Taxa	n	Mean R^2	Max	Taxa	n	Mean R^2	Max
Araneae		(none)		**Araneae**			
				Ctenizidae	1	97.47	97.47
Collembola				Pisauridae	3	79.82	89.99
Hypogasturidae	3	48.10	100.00	Dysderidae	2	79.46	100.00
Sminthuridae	5	34.70	98.92	Salticidae	3	45.98	99.32
				Agelenidae	7	32.14	100.00
Orthoptera							
Tetrigidae	1	100.00	100.00	**Collembola**			
Gryllidae	6	24.84	93.99	Dicyrtomidae	4	85.85	100.00
				Entomobryidae	8	22.82	88.36
Coleoptera							
Scarabaeidae	5	54.69	100.00	**Orthoptera**		(none)	
Formicidae				**Coleoptera**			
Technomyrmex	1	100.00	100.00	Chrysomelidae	2	100.00	100.00
				Cleridae	1	100.00	100.00
				Curculionidae	4	81.71	100.00
				Histeridae	2	75.84	100.00
				Carabidae	5	48.50	100.00
				Lyctidae	2	44.21	62.96
				Formicidae			
				Crematogaster	1	56.50	56.50
				Tetramorium	6	19.09	89.16

Table 6.3.3.3 Taxa that correlated with biotope quality from the disturbed fynbos, with reference to the number of morphospecies used within the taxa (n), the mean square of Pearson's product moment correlation coefficient in percentage (R^2), as well as the maximum percentage correlation for all morphospecies

Positive correlations				Negative correlations			
Taxa	n	Mean R^2	Max	Taxa	n	Mean R^2	Max
Araneae				**Araneae**			
Selenopidae	1	100.00	100.00	Salticidae	1	100.00	100.00
Agelenidae	8	40.56	100.00	Ctenizidae	1	90.67	90.67
				Dictynidae	4	59.98	98.37
Collembola							
Poduridae	1	65.73	65.73	**Collembola**			
Entomobryidae	10	31.11	100.00	Dicyrtomidae	5	65.69	100.00
Orthoptera				**Orthoptera**			
Tetrigidae	2	71.60	100.00	Stenopelmatidae	1	100.00	100.00
				Gryllidae	6	20.68	100.00
Coleoptera							
Chrysomelidae	1	100.00	100.00	**Coleoptera**			
Curculionidae	2	82.78	100.00	Anthicidae	2	100.00	100.00
Trogidae	1	65.57	65.57	Ptiliidae	1	100.00	100.00
Histeridae	2	55.45	94.34	Carabidae	10	74.57	100.00
Scraptiidae	2	42.03	70.92	Eucnemidae	1	58.79	58.79
Mordelidae	3	39.15	100.00	Silvanidae	2	57.35	99.85
				Bostrychidae	4	38.56	100.00
Formicidae				Staphylinidae	8	26.17	100.00
Camponotus	3	83.07	100.00				
Tetramorium	6	32.83	94.08	**Formicidae**			
				Dorylus	1	100.00	100.00

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6.4   DISCUSSION

The taxonomic *IndVal* evuation as well as the *BQI* guild evaluation identified ants and springtails as the most suitable for use as bioindicators, at least in this fynbos system. Ants as representative of the omnivores/scavenger guild showed positive correlation to both pristine and disturbed fynbos sites, with the exception of the Erica-dominated fynbos and the younger pine plantations.   Genera such as the alien *Linepithema* and the native *Pheidole* are opportunistic towards these disturbed conditions. The Erica-dominated fynbos was also highly suitable for ants, with easy excess to resources and optimal heat conditions from sunlight penetrating to the soil surface (Bazzaz, 2000). Disturbed areas tended to have more predators due to the fact that they may find it easier to catch prey (Speight *et al*., 1999). Ants are better adapted than other insects to deal with these conditions, with some safety in numbers as well as an array of defence mechanisms (Alonso *et al.,* 2000).

Springtails were consistent within *IndVal* evaluation, as well as regression analysis between *BQI* and disturbance levels. It is in the Restio-dominated fynbos that they showed most potential, but surprisingly showed negative correlation in Erica-dominated fynbos. This might be due to the family Dicyrtomidae that seems to prefer open areas with little shade, and could be associated with dieback of plants in disturbed conditions (Rusek, 1998).

Ground spiders made for good indicators but cannot be used with other taxa within the *BQI*, as their numbers increased at the two extremes of the disturbance scales. Under disturbed conditions, their numbers increased following the increase of invasive plant species that take advantage in these conditions, as well as the increased ability to find weaker prey under these conditions (Matsuoka & Seno, 2008).   Additionally at sites with better environmental quality, the expected increase in abundance of spiders followed the increase in biomass of the prey, as well as a decrease in numbers due to the time spent finding prey. In

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the low growing vegetation, Erica-dominated and Poaceae-invaded fynbos, the correlation between the *BQI* and disturbance level was positive, while in the older plantations the correlation was negative.

Here, orthopterans were not good potential bioindicators within the fynbos system, as expected from the literature review (Chapter 3, p.47-110), but never the less showed there was good correlation between the *BQI* and the disturbance level within pristine Protea-dominated fynbos. These weak results were largely due to the use of pitfall traps. Pitfall sampling are far from ideal for sampling this taxon, and a better suited sampling method that focus on grasshoppers might show more meaningful results.

All the guilds were better as bioindicators within the *BQI* than the pooled beetle assemblage as broad-spectrum control group. The pooled beetle data gave a weak result, mainly due to the balancing out factor, whereby good indicators are suppressed by bad indicators. These results indicate that a selected group of taxa can be used within the *BQI*, rather than using pooled data (as seen in beetles). Therefore Samways *et al*.'s (2010) four step selection method is recommended.

To determine the best group of bioindicators the first step, according to Samways *et al*. (2010), is to make a decision as to which ecosystem attributes the indicator taxa should reflect. This involves deciding on which biotope attributes of the focal ecosystem are suitable for variety of specialist species. This means that the multispecies group should contain species that are sensitive to ecosystem change, such as fragmentation, overall environmental change, or even microclimate change. These species should be dependent on one or more of the typical biotope attributes, and preferably all biotope attributes, i.e. they should be sensitive to changes away from the ideal conditions (Samways *et al*., 2010). The

BQI focuses mainly on the quality of a biotope, and thus is reactive to species that are sensitive to disturbances to the environment in which they live.

The second step identifies only relatively rare and yet easily detectable species, which are evenly distributed in the focal area, with high abundance for better statistical analysis (Samways *et al*., 2010). Many arthropod species are in this category. However, it is essential that the species used in this analysis are adequately sampled so that the sensitive species are recorded.

The third step in Samways *et al.*'s (2010) identification of potential indicators species, asks for information based on the ecology of each species. However, there is often very little knowledge regarding specific arthropod species, while as at family level there is a better understanding. Until better knowledge of the species level is available, most species within arthropod families have similar ecologies and therefore can be pooled as bioindicators to create better data sets. Guilds also provide an option for pooling all arthropods according to similar ecologies, or more specifically according to what they consume.

Ants often do not fit the above criteria, as there can be large differences in biology from one species to another within the family. Nevertheless ant genera can be surrogates. Particular ant genera that can be used within the *BQI* for the fynbos biome are *Camponotus* and *Technomyrmex*. *Dorylus* and *Crematogaster* showed negative correlation and thus should be avoided. *Linepithema* and *Pheidole* owing to their high abundance are also not suitable (Sanders & Suarez, 2010). *Linepithema humile* is a known invader species and has the ability to outcompete most other ant species, especially in disturbed conditions, therefore it is not a true indication of biotope quality. The genera *Tetramorium, Monomorium, Meranoplus, Ovymyrmex, Pachycondyla, Thoptromyrmes* and *Solenopsis* did not show any significant or gave mixed results, and should be tested for positive or negative correlation to

disturbance levels in specific biotopes before being short listed as indicators. Other foraging families that can be considered for use in short listing are the hymenoptera families of Apidae and Vespidae, as well as most families of Isoptera (Chapter 3, p.47-110).

The springtails, showed positive results for the families Onychiuiridae, Poduridae, Sminthuridae, Entomobryidae and Hyposgasturidae for use in the *BQI*. Only the family Dicyrtomidae should not be included. The other families, Odontellidae, Brachystomellidae, Neanuridae, Isotomidae, Paronellidae, Cyphoderidae, Tomoceridae and Neelidae should be tested for selected biotopes, as they showed poor indication of ecosystem health within the fynbos here, and even their absence within parts of the area. The literature review (Chapter 3, p.47-110) identified the genera *Porrectodea, Eisenia, Enchytraeus, Folsomia, Hetromurus, Paronychiurus* and *Sinella* possibly to be good bioindicators.

Tetrigidae (Orthoptera) showed the best potential as a bioindicator within the fynbos biome. The Gryllidae (Orthoptera) also showed promise, although this group would need more testing before selection. Stenopelmatidae showed clear negative correlation at most sites, and thus must be excluded from use within the *BQI* for the fynbos biome. Species in the Acrididae family was present only in low abundance, and therefore did not have bioindicator value, at least with this trapping method. Families Tettigoniidae, Acrididae, Gryllotalpidae, Gryllacrididae, Schizodactylidae, Rhaphidophoridae, Euschmidtiidae, Thericleidae, Pneumoridae, Charilaidae, Pamphagidae, Lathiceridae, Pyrgomorphidae and Lentulidae need more evaluation, preferably using more appropriate sampling methods specifically for these groups.

The spiders had two families, Selenopidae and Agelenidae, with potential for use in the *BQI*. The other families, Ctenizidae, Pisauridae, Dysderidae, Salticidae and Dictynidae all showed negative correlation and should not be included in the *BQI*. Other families were

not included most likely because they rarely will be trapped within pitfalls and therefore need further testing before including them within the selected bioindicators.

Although beetle species pooled together did not show strong correlation with disturbance level, individual families did show significant positive correlation. These included Curculionidae, Cucujidae, Scarabaeidae, Mordelidae, Scraptiidae and Trogidae, and could be included for use in the *BQI* for the fynbos biome. The families Eucnemidae, Ptiliidae, Cleridae, Anthicidae, Bostrychidae, Silvanidae and Lyctidae should be left out, owing to weak correlation. Chrysomelidae and Histeridae gave different results under different circumstances and thus need further testing. Carabidae and Staphylinidae, which are often recommended as bioindicators (Chapter 3, p.47-110), showed negative results here in the fynbos and so would need further evaluation. These results might be due to the fact that they are also predators like spiders, and can thus take advantage of other species in unfavourable conditions. Representatives of both families are also highly mobile and only spend a short time within disturbed habitats to feed on other species.

Pest and invasive species that can occur in extreme abundances within a biotope, due to lesser pressures from environmental conditions, may out compete the native species (Speight, 1999). In these conditions the pest or invasive species will show clear negative correlation within the correlations test, and thus removed from the calculation, as mentioned above with i.e. *Linepithema humile.* Additionally, the native species numbers will be below average due to the pressure from the pest or invasive species and therefore not attribute to the *BQI* value, giving the infected biotope a low value, sign of a problem within the biotope. Furthermore, species that are highly mobile and easily move between biotopes, also do no contribute mush to the health of a biotope and these species will be present as meanly singletons within the data set. Singletons will always be below or equal to the mean in the calculation of the *BQI* and therefore not contribute to the total.

For selection of the final set of taxa as potential indicators, and which will best suit the *BQI*, the short listed groups must first be tested under various situations and conditions, as well as making use of various different methodologies. As well as providing us with the best potential indicator, these short listings need to include those taxa which also react to various stressors. The *BQI* thus has the potential to grow into a more effective index as we gather more information and rework the process.

6.5   CONCLUSION

The *BQI*'s taxonomic division (Coleoptera as broad-spectrum control group) fared poorly relative to the various guilds. The foraging (ant) and decomposer (springtail) guilds stood out as the most promising bioindicators, while the predatory guild (spiders), showed negative correlation to the level of disturbance.

Testing of a set of bioindicators for a specific biotope, and a specific trapping method is recommended. A simple correlation test of morphospecies abundance versus a non-parametric index for the level of disturbance will indicate strong positive correlation (>33.33%), weak positive/negative correlation (0-33.33%), and strong negative correlation (>33.33%). I recommend selecting only the morphospecies that indicate strong positive correlation (>33.33%) for use in the *BQI*.

6.6   REFERENCES

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Chapter 7

THE EFFECT OF SEASONALITY ON THE BIOTOPE QUALITY INDEX (*BQI*) RESULTS

ABSTRACT

Arthropods, being poikilothermic, are highly responsive to weather and climatic conditions. Such conditions affect the ability of arthropods to find food and reproductive partners, and they have evolved their life strategies around these seasonal changes. The Biotope Quality Index (*BQI*) makes use of the relative abundance of various species as bioindicators of biotope disturbance and quality. Changes in abundance of species caused by seasonal change, might thus also affects the *BQI* values. I aim here to test whether there is a significant difference between the seasons throughout the year, and if so, then determine which time period is most favorable for sampling arthropods for *BQI* evaluation. Data from 30 sites in a protected area in the Cape Floristic Region showed that the pooled data for all seasons represented the strongest correlation to disturbance level. Of the four seasons, autumn was the best time for *BQI* sampling, at least when using pitfall trapping within the fynbos biome for comparative sampling. The favorableness of autumn is probably due to high species abundance of adult arthropods actively moving around, looking for the most suitable biotopes with the best food and reproductive partners during this comparatively mild and moist time.

KEYWORDs: Arthropoda; Autumn; Biotope Quality Index; Level of disturbance; Seasonal change; Seasonal sampling

7.1 INTRODUCTION

A season is a division of the year, marked by changes in weather, climate and daylight availability, resulting from the yearly revolution of the earth around the sun and the tilt of the earth's axis relative to the plane of revolution (Khavrus & Shelevytsky, 2010). These changes can trigger alterations in the ecologies of various animals. Arthropods, being poikilothermic animals, are more affected by these changes than larger mammals due to their small size and that they are incapable of regulating their own body heat (Waldbauer, 1996).

Seasonal changes refer to changes in temperature, rainfall, wind and humidity and thus are important drivers of arthropod ecology (Speight *et al*., 2009). Insect life stages have evolved around these changes in the seasons. Many species use egg and pupal stages as an overwintering mechanism, or even as a dormant stage during extreme heat (Leather *et al*., 1995). Moisture and daylight availability play a major role in determination of embryonic development, and are an indirect cause for survival, since without the right weather conditions, the resources that the arthropod needs will not be available (Speight *et al*., 2009). These seasonal changes also synchronise sexual development, and can initiate migration and dispersal behaviour (Drake & Gatehouse, 1995).

Mansingh (1971) divided the reaction of populations to unsuitable seasonal changes into three categories: 1) Quiescence, referring to the response of individual arthropods to a sudden unanticipated, non-cyclic and usually short duration deviation of normal weather conditions, mainly causing growth retardation. 2) Oligopause is seen in species inhabiting areas of moderate winter where there is a fixed period of dormancy in response to cyclic and rather longer-term climate change, resulting in species that collect nutritional reserves. 3) Diapause, which differs from the other two in that it is a definite preparatory phase usually initiated by a temperature-independent factor, involving metabolic changes. The insect does not feed during this period, and the return of favourable conditions does not terminate diapause immediately.

Temperature influences everything that an arthropod organism does (Clark, 2003). The higher the temperature, the faster metabolic reactions are able to take place. Higher metabolic rates, in turn, affect the development rates, as well as life expectancy of arthropods (Speight *et*

al., 2009). Adaptations by insects to local temperatures can also include changes in phenology and morphology. Peat *et al.* (2005) showed that bumble bees are larger and have longer setae in colder local conditions. The activity and foraging success of predators or parasitoids are also temperature dependent (Bourchier & Smith, 1996).

Day length provides an insect with information about the progression of the seasons, and the ability to vary growth and development rates, as well as enabling it to achieve efficient timing relative to favourable conditions (Leimar, 1996). Parasitoids, for example, only develop to adulthood when day length reaches a certain length of time, to insure that their prey is fully developed and active, so energy is not wasted looking for prey that is not present (Speight *et al*., 2009). In turn, wind can displace smaller insects (Harrison & Rasplus, 2006), and it often brings rain needed for food production (Ji *et al*., 2006). Wind can create new habitats by, for example, fallen trees (Eriksson *et al*., 2005), and can help or interfere with chemical communication (Wyatt, 2003). The effect of rainfall can be direct or indirect. Heavy rain can knock insects from their host plants, while the lack of rain can cause desiccation and death (Speight *et al.,* 2009). Rain can influence the ways in which host plants flourish and grow and provide food, in particular, the quality of food (Hale *et al*., 2003). Palmer (2010) showed that the amount and duration of rainfall has significant effects on phenology of species. Droughts can stress trees, rendering them susceptible to insect attacks (Speight *et al.,* 2009). Rainfall also affects humidity, which combines with temperature and wind to dictate local microclimatic conditions (Speight *et al*., 2009).

This information suggests that quality show a correlation with the weather patterns and seasonality. Too much rain, draught, fire at the wrong time, all these factors can affect the quality of the environment. Thus sampling at the wrong time, and not taking seasonal changes into account will have an effect on the results of *BQI* evaluation. This chapter evaluates whether there is variation within the results when sampling in different seasons. Furthermore, it looks for the best time of the year to sample for *BQI* evaluation (within the Fynbos biome).

7.2   METHODS

7.2.1   *Study area and sampling methods*

Wicht *et al.* (1969) described the climate of the Jonkershoek Valley as being a Mediterranean type, whereby the system has an average temperature of below 22°C for the warmest month. This correlates with the classification used by Purves (1995) as a Humid-mesothermal Erica-climate (Type Csb) and Climate Type IV in Walter and Leith's classification system (Werger *et al.,* 1972).

During the summer, strong anti-cyclonic winds from the south-east, enter the Jonkershoek Valley over the Dwarsberg. This wind also called the Black southeaster, originates in high pressure cells between about 25°- 35° South latitude and often reaching gale force strengths (Weather Bureau, 1960). This air is cold and as it passes over the Dwarsberg, its temperature drops further causing moisture condensation and creating the renowned south-east cloud (McDonald, 1985).

Prefrontal cyclonic winds of the westerly wind system bring moisture-laden air form the north-west into the Jonkershoek Valley in winter. Heavy orographic rain results when these winds meet the mountain barrier (McDonald, 1985). After the passage of a cold front, the wind direction usually changes from north-west to west and south-west. The resultant drop in temperature and rise in atmospheric pressure cause unstable conditions, during which storms and showers often occur (Wicht *et al*., 1969; Jackson & Tyson, 1970; Erasmus, 1981).

Eighty five percent of the rainfall in the Jonkershoek Valley occurs between April and September (Versfeld & Donald, 1991), with a mean annual rainfall, recorded over a 50 year period (1925 to 1984) at Jonkershoek, of 1096 mm/yr (Salie, 2003). Swartboschkloof weather station recorded a mean annual rainfall over a five year period of 1567.5 mm/yr, 104 days of rain/yr and a mean rainfall per event of 15.1 mm/day (Miller *et al.,* 1985).

The upper reaches of the catchments are known to have higher rainfall than the lower areas and are more prone to flooding and spate flows (Sera & Cudlin, 2001). Wicht *et al*. (1969) reported an annual mean of 1850 mm/yr in the upper reaches of Swartboschkloof, and at the top of the Dwarsberg range, the mean annual rainfall increases to 3600 mm/yr. This clearly

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demonstrates the increase of rainfall with elevation at any site within Jonkershoek. There is also an increase from north-west to south-east in annual rainfall in the Jonkershoek Valley (Wicht *et al.* 1969; van Laar 1984). Heavy rains have been recorded when a decrease in temperature occurs as air passes over the Dwarsberg, resulting in condensation and often moisture deposition at the higher altitudes. This cooling effect causes the persistent cloud cover over the valley. The air is not adiabatically heated and moisture is not reabsorbed into the atmosphere (Wicht *et al.*, 1969; Fuggle & Ashton, 1979).

The rainfall is usually cyclonic and orographic, but thunderstorms can occur (Weather Bureau, 1960). Hail and snow occasionally occur during winter, but the snow is restricted to the higher altitudes, usually above an elevation of 915 m (Wicht *et al.*, 1969).

As can be expected in the mountainous Jonkershoek Valley, temperatures vary considerably with locality. High, sustained temperatures are experienced in the Eerste River basin. The lowest winter temperatures are also found in these localities due to strong nocturnal infra-red reradiation and katabatic drainage of cold air from the mountains to the valley (van Lill, 1976). In the Jonkershoek Valley, the period from December to March is not only the driest, but also the warmest, with average daily maximum temperature of 28°C in midsummer compared to 17°C in midwinter. The average daily minimum is 15°C in summer and 6°C in winter, while a minimum of 4°C and -5°C in January and July respectively can occur at higher elevation (Weather Bureau, 1984; Heth & Donald 1978), but temperatures rarely fall below freezing point. The mean daily temperature fluctuation varies little from summer (14°C) to winter (11°C). The nights are mild to cool with frost being very rare (Wicht *et al.,* 1969; Versfeld *et al.,*1992).

Thirty sites in the Jonkershoek Valley, Cape Floristic Region, South Africa (38°58' S; 18°55' E), were selected to represent six main biotope types, and an *a priori* range of disturbance levels from fully natural to disturbed. Sites were categorized into three natural fynbos vegetation biotope types: <0.5 m-high, Ericaceae-dominant fynbos (five sites), ±1m-high Restionaceae-dominant fynbos (four sites) and >1.5m-high Proteaceae-dominant Fynbos (five sites).   In addition, a grass-invaded fynbos (five sites) and two exotic pine plantations categories: <10 yr old pine stands (six sites) and >10 yr old pine stands (five sites) were used (Figure 4.2.1, p.100). Each site was evaluated according to integrity and disturbance levels, and rated from near-natural conditions to least-natural conditions, using mean values from the

opinionated categorized survey done by a group of research students from the Stellenbosch University (n=7). The disturbance level was then calculated taking into account, pollution, invasive aliens, distance from nearest disturbance, human activity and biotope structure, among others. These subjective data provides a spectrum so that we can correlate the objective Biotope Quality Index (*BQI*) for each season.

Ten 350 ml pitfall traps (Woodcock, 2000) were placed at random at each of the 30 sites for seven days in each of March, June, September and December 2006, to represent the four seasons. Ants (Formicidae) and springtails (Collembola) were selected as the focal taxa for this study, as they were species rich and abundant groups within the pitfalls. All individuals were sorted to morphospecies.

7.2.2   *Statistical analysis*

The *BQI*, introduced in chapter 4 (p. 111-131), was used to calculate values for each of the seasons as well as for the whole year. The observed species richness, as well as relative abundance, was also calculated. These values were then tested for significant difference using Kruskal-Wallis non-parametric tests (Kruskal & Wallis, 1952) in Microsoft Office Excel 2007 (Reding & Wermers, 2008). The null hypothesis that there is no difference in the data retrieved between seasons. Regression analysis to test correlation between each season's *BQI* values against the level of disturbance was calculated in Microsoft Office Excel 2007 (Reding & Wermers, 2008) to determine which season is best suited to *BQI* sampling.

7.3   RESULTS

There was no significant difference between relative abundances over the seasons ($H=2.38$; $df=3$; $p >0.5$). However, species richness ($H=17.71$; $df=3$; $p <0.001$), as well as the *BQI* values ($H=24.19$; $df=3$; $p <0.001$), were highly significantly different between seasons. Regression analysis between the level of disturbance and the *BQI* values (Figure 7.3.1) indicated that the pooled data over the whole year showed the best correlation (55.57%). Of the various seasons, autumn (March) showed the highest correlation (47.12%), while summer (December) had the lowest correlation (14.4%). All seasonal regressions showed positive correlations.
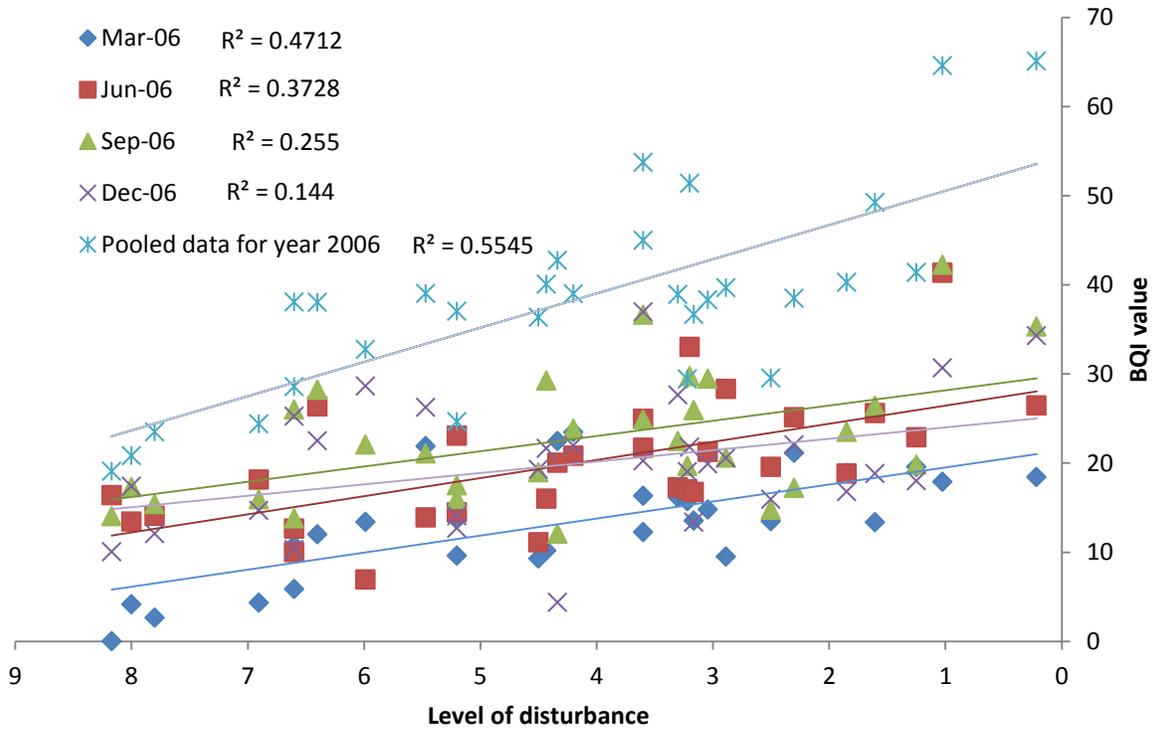
Figure 7.3.1 Differentiation between Biotope Quality Index values calculated separately for each

season, as well as the pooled data set for all sites combined

7.4   Discussion

Seasonal change seems to have an effect on the *BQI* data, probably indirectly, owing to the significant difference of the species richness between the seasons.

The fynbos biome endures winter rainfall, with relative dry summers. Insects that prefer the colder, wetter conditions flourish within the winter, while those which prefer drier weather increase in numbers during summer (southern hemisphere). The study area has high levels of rainfall during winter, and occasionally even some snowfall (Visser, 1990). This all indicates that arthropod species would have adopted a strategy of overwintering during the potential cold, rainy weather, in either egg or pupal phases (Leather *et al*., 1995).

The rainy winter season could also increase the number of species associated with the riparian environment (Bagatini *et al*. 2010), where seasonality had a significant effect on trophic status and energy content of invertebrates. Energy content, referring to the biomass and abundance of individuals, is a factor that could affect the results of the *BQI* evaluation. The summer fire season combined with a scarcity of food, due to it being the dry season, presents another seasonal period that challenges some insects (van Wilgen, 2010). These adverse times suggest that sampling for the *BQI* should take place during spring or autumn. It is during spring that most arthropods emerge from eggs, and are found in immature stages mainly on host plants or within stream biotopes. During these times, arthropods rarely actively search for optimal food sources (Stehr, 2008). Furthermore, there is little identification literature for immature stages to species level. This in combination with my results here point towards the best season for sampling for the *BQI* being autumn, when most mature adults are actively looking for mates as well as food. However, overall, my results indicated that the pooled data over all seasons showed the best correlation, making a pooled data set optimal for sampling for *BQI* evaluation. Thompson and Townsend (1999) point out that there is a significant difference within various aspects of the food webs over seasons. If this is true for food webs, it will be indirectly true for the ecology of the arthropods, driven by changes within the environment. The *BQI* looks at the quality of the environment as a whole, and that includes all seasons, because an event within one season will affect the population sizes in following seasons.

7.5   CONCLUSION

Seasonal changes affect the *BQI*. Therefore I recommend that for comparative studies, sampling must take place preferably in the same season. But as the pooled data for the whole year gave more significant correlation than the individual seasons it is best if data are collected over all the seasons and pooled. However, of the seasons, autumn (March) sampling can be recommended for pitfall sampling in the fynbos, as it gave the best correlating *BQI* results.

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7.6   REFERENCES

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Chapter 8

AN EXAMPLE OF EVALUATING BIOTOPE QUALITY USING THE BIOTOPE QUALITY INDEX (*BQI*) AND LOOKING AT POTENTIAL INVERTEBRATE BIOINDICATORS FOR THE JONKERSHOEK VALLEY, SOUTH AFRICA

ABSTRACT

This study tests the *BQI* for evaluating the biotope quality of thirty variously transformed sites. The sites were allocated to six main biotope types and an *a priori* range of disturbance levels from relatively pristine to disturbed fynbos. Pitfall sampling was used to comparatively sample arthropod morphospecies for use in the *BQI*. Hymenoptera and Collembola species were dominant in the traps. Collembola population levels were highest in the winter months, while Hymenoptera, more specifically Formicidae, levels dropped greatly during the same period. In turn, Coleoptera relative abundance was highest during spring. There were significant differences in the *BQI* values between natural fynbos and pine plantations, between pristine Ericaceae dominated fynbos and invaded Poaceae fynbos, between younger and older pine plantations, as well as between the Ericacae dominant, Restioaceae dominant and Proteacea dominant fynbos. The study found that there might be an effect on biotope quality from agricultural activities, presence of dirt road, as well as from fire management. The *BQI* is a useful management evaluation tool for comparison of environmental conditions and biotope quality.

KEYWORDS: Arthropods; Biotope quality, Disturbance level; Diversity indices; Biotope Quality Index.

8.1   INTRODUCTION

From the greater need to properly evaluate the status of our environment, due to the pressures placed on the environment by humanity (Mooney *et al.*, 2005), I developed the *BQI* (Chapter 4, p.111-131) based on the definition of ecosystem health (Chapter 2, p.12-46). A healthy ecosystem is one that can sustain an optimal number of species with an optimal number of specimens and their ecological processes, thus providing an optimal heterogeneous, sustainable system with sufficient resources, and has adequate resistance when under perturbational stress, yet enable natural succession to still take place. The *BQI* make use of arthropod species assemblages, comparing these values to each other to identify above average populations sizes. During this evaluation, the next step is to count the number of standard deviations away from the mean for each population, and then add these values for each morphospecies present. This will lead to a total which indicates the quality of the environment, based on the concept where a healthier biotope will be able to support and sustain more individuals (Chapter 4, p. 111-132).

The *BQI* is suitable for comparative studies of biotopes sampled within the same season, as well as those sampled in same method (Chapter 7, p. 176-187). Furthermore, it can be used to compare biotopes over time, as long as methodology used to obtain data are consistent (Chapter 4, p. 111-1131). Although all data of morphospecies abundance can be used in the *BQI*, we recommend the use of arthropods, due to their strong record as bioindicators, as well as their large numbers, ideal for statistical analysis (Chapter 3, p.47-110). The evaluation will most likely be better if only comparing biotopes within the same biome type.

This study tests the use of the *BQI* evaluation method for determining the impact of human activities using an example (Jonkershoek Valley, Western Cape, South Africa). These activities include: afforestation with exotic pine trees, impounding of the Eerste River by the Kleinplaas Dam, wine farming and associated agricultural processes, fire management, as well as tourism and leisure activities that include scenic drives along the a circular road through the park, mountain biking, fly-fishing and paintball war games (CapeNature, 2008).

The Jonkershoek Valley is in the Cape Floristic Region (CFR), part of the south western mountainous areas of the Mediterranean regions. The CFR is known for its cold, relatively humid winters and hot dry summers, with distinctive floral adaptations, known as fynbos (Salie, 2003).

The CFR, flora is a fire adapted, species rich sclerophyllous shrubland with a scarcity of trees and with only minor representation by grasses and evergreen succulent shrubs (Kruger, 1979). The dominant plant families are Proteaceae, Restionanceae and Ericaceae.

Compared to the floral component of the CFR, the fauna have received relatively less attention. Earlier studies indicated that the faunal diversity shows little correlation with that of the highly diverse flora (Goldblatt & Manning, 2000). More recently it has been suggested that this might be due to a lack of information on the ecology and biology of the fauna, especially the invertebrates (Procheş & Cowling, 2006).

Fire is the greatest natural pressure placed on individuals of fynbos biome (Heelemann, 2011). In Jonkershoek, management for restraining the spread of fire involves removal of large areas of fynbos to create barriers between the pine plantations and natural fynbos to prevent fire moving into the timber. Furthermore, the biomass of fynbos is not burned on a regular basis due to the fear of fires spreading into pine plantations and other agricultural areas (Working on Fire, 2011). This actions lead to the build-up of highly flammable fynbos biomass.

Afforestation is another major problem as it converts a highly diverse environment into a monoculture. Richardson and van Wilgen (1986) established that in the period from 1945 to 1984, fynbos vegetation species richness in Jonkershoek was reduced by 58% in biotopes where *Pinus radiata* had been planted. Furthermore, pine trees use much more water and nutrients than the natural fynbos (Everson *et al.*, 2011). There is also secondary affects that take place, for example, the effect that chemical seepage and leaf litter have on the soil (Gielis *et al.*, 2009). This effect kills the fynbos ground cover and in the process reduces the quality of the biotope.

The lower parts of the Jonkershoek Valley are associated with viticulture (CapeNature, 2008). The management regimes might not affect the biotope quality directly, but we can expect that the

management of pests, in the form of chemical control, will have an effect on the environment and kill numerous indigenous arthropods (Desneux *et al*., 2007), as well as allow bioaccumulation of toxic substances within certain predator species (Golsteijn *et al*., 2011).

Three main geographical areas of invasive plant species can be identified within the Jonkershoek Valley: 1) along the riparian zone with various alien trees and shrubs, 2) pine trees establishing around the edges of plantations, and 3) grass invasion in various human disturbed areas. Examples of these biotopes are in the vicinity of the dam and pipelines, areas cleared for fire management, as well as alongside the circular ecotourism road that runs through the park (van Wilgen *et al*., 2006). This road might also create an edge effect, due to noise, runoff, chemical and air pollution (Morgan *et al*., 2010).

Recent ecological research on arthropods in the Jonkerhoek Valley region includes studies on insect biomass and community structure by Donnelly (1983), as well as Schlettwin and Giliomee (1987). They found biomass and number of ants, grasshoppers and leafhoppers to be lower than in comparison with savanna ecosystems, although the 31 ant species collected correlated well with the species richness of other temperate zones of the world (Schlettwien & Gillomee, 1987).

Furthermore, aquatic invertebrates were the centre of studies by Bredenhand and Samways (2009), using benthic macroinvetebrates as bioindicators of water quality. Arkell (1979) studied the importance of freshwater crabs in the diet of kingfishers, and Coetzee (1989) focused on the effect of salinity on the Cape river shrimp. Coetzee and Gillomee (1985) focused their studies on insect-plant association, while McCall and Primack (1992) looked at the effect of flower characteristics, weather, time of day, and seasonality on visitation rates of insects. Studies on seed dispersal as an insect-plant interaction are very well represented in studies from this area (Lach, 2007). Witt and Gilliomee (2004; 2005) and Witt *et al*. (2004), Investigated the effect of the alien ant, *Linepithema humile* and other ants within the system.

8.2   METHODS

8.2.1   Sample site

The Jonkershoek Valley is defined by the Eerste River catchment area, surrounded by the Jonkershoek mountains to the north-east, Dwarsberg mountains to the south-east and Stellenbosch mountains to the south-west, making it part of the greater Boland mountain range (Figure 8.2.1). The Eerste, Berg, Lourens and Riviersonderend rivers have their various sources high in the mountains, but only the Eerste river flows through the valley (CapeNature, 2008).

The Jonkershoek Valley comprises the conservation areas of Jonkershoek Nature Reserves, Assegaaibosh Nature Reserve, Hottentots Holland Nature reserve and Jonkershoek Forestry Reserve as well as an agricultural zone (CapeNature, 2008) (Figure 8.1.1). Jonkershoek Valley climate consists of seasonal displacement of wind belts that cause the dominant westerly antitrade winds to carry rain from the oceans during winter. During the summer months, the easterly trade winds blow dry air from inland, causing low rainfall. Visser (1990) reported that the rain fall in the Jonkershoek Valley is in the range of 250-1200 m/yr, and among the highest in South Africa. The mean annual maximum temperatures ranges from 18°-30°C, while the mean annual minimum temperature range from 8°-12° C. Snow may occur on mountain tops, and frost is a feature of some low lying areas during the winter.

8.2.2   Step 1 – Selecting the study area

Thirty sites in the Jonkershoek Valley, Cape Floristic Region, South Africa (38°58' S; 18°55' E), were selected to represent six main biotope types, and an *a priori* range of disturbance levels from natural to disturbed. Sites were categorized in three natural fynbos vegetation biotope types: <0.5 m-high, Ericaceae-dominant fynbos (five sites), ±1 m-high Restionaceae-dominant fynbos (four sites) and >1.5 m-high Proteaceae-dominant Fynbos (five sites). In addition, a grass-invaded fynbos (five sites) and two exotic pine plantations categories: <10 yr old pine stands (six sites) and >10 yr old pine stands (five sites) were used (Figure 8.2.1).

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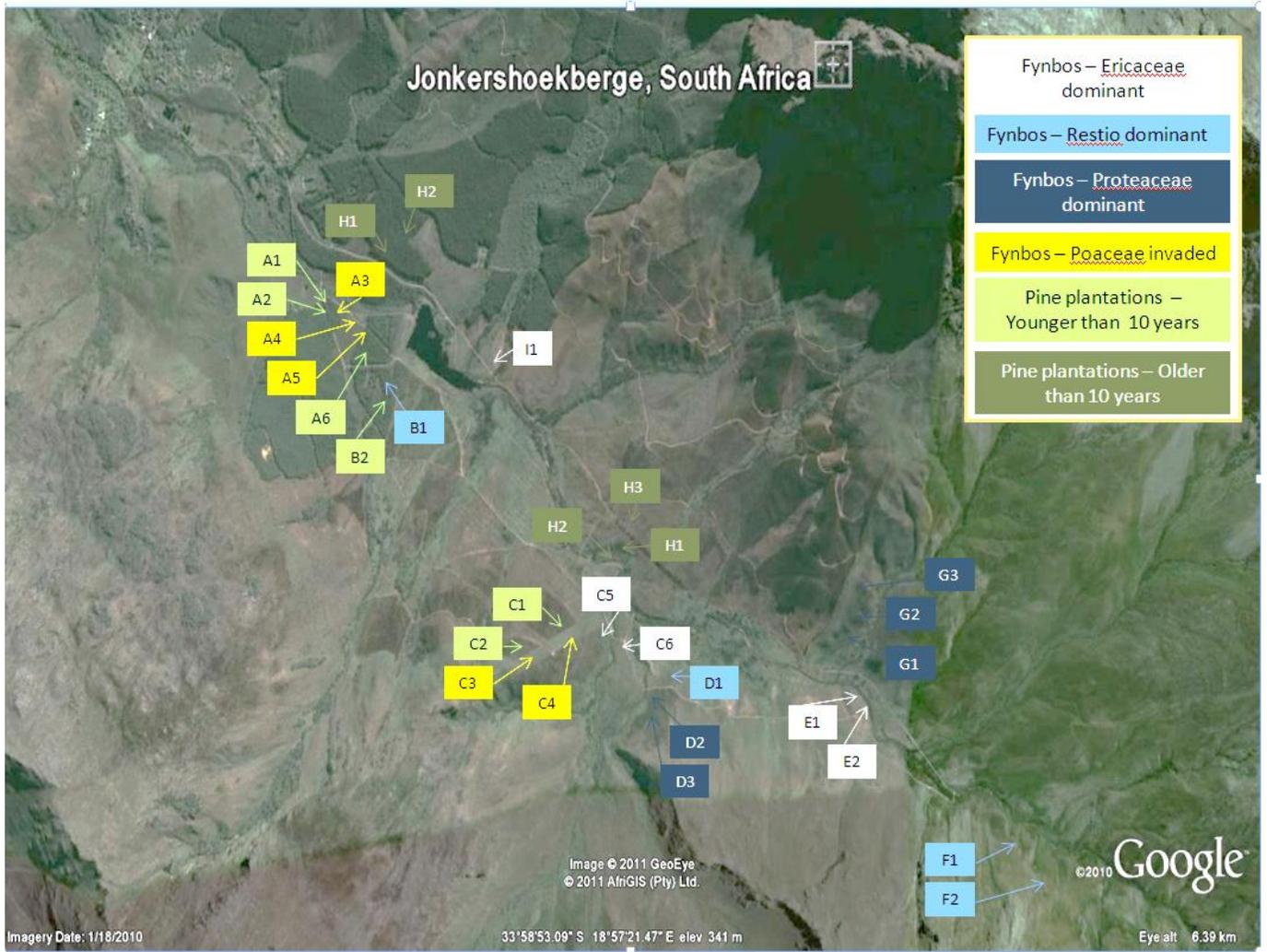
Figure 8.2.1 Geographical distribution sampling sites within the Jonkershoek Valley (Aerial photo of Jonkershoek Valley, downloaded from GoogleEarth (GoogleEarth, 2011))

8.2.3   *Step 2 – Determining the level of disturbance*

Each site was evaluated according to integrity and disturbance levels, and rated from near-natural to least-natural, using mean values from the opinions of a group of Masters and PhD research students within Entomology department of Stellenbosch University. This was achieved by making use of a rubric scoring table system.   The disturbance level was then calculated taking in consideration, pollution (0 = no visual signs of pollution : 10= rubbish dump), invasive aliens (0 = no invasive plant species present : 10= 100% alien plants), distance from nearest disturbance (0= more than 1000 m from nearest road ; 10 = less than 5 m from nearest road), human activity (0 = no visual signs of any human activity : 10 = building with constant human presence) and biotope structure (0 = signs of new growth and no unhealthy plants : 10 = diseased plants and no sign of natural new

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growth). This was a subjective approximation of data. While this non-parametric method does provide a means of ranking the biotopes from least disturbed to most disturbed against which we can correlate the relative abundances of the morphospecies that show positive correlation. Disturbance level is given here as a reverse scale and thus normal statistical negative slope in the data are seen as a positive correlation.

8.2.4   *Step 3 - Field sampling*

Ten 350 ml pitfall traps (Woodcock, 2000), were placed at random at each of the 30 sites for seven days each in March, June, September and December 2006, representing the four seasons.

All sampled specimens were sorted to morphospecies. The abundance of each of the morphospecies was used to test for correlation against the level of disturbance and to identify the species that shows positive correlation (increase in disturbance results in a decrease of sample individuals) using Microsoft Office Excel 2007 (Reding & Wermers, 2008).

8.2.5   *Step 4 - Calculation of BQI value for each site*

The morphospecies identified in step 3, were then used to calculate the *BQI* value for each site making use of the formula given in chapter 4 (p.111-131).

$$BQI = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \bar{x}_i}{\log \sigma} \right) \text{ when } \frac{\log x_{ij} - \log \bar{x}_i}{\log \sigma} > 0$$

Morphospecies which occurred as singletons across the sites, as well as those with the same abundance in all sites present were deleted from data set to make calculations easier. The reason for this exclusion is that neither of these 'types' of morphospecies contributes to above average populations and therefore irrelevant for the *BQI*.

Biogeographical maps, of *BQI* status, were drawn up making use of GoogleEarth (GoogleEarth, 2011) and Microsoft Office Powerpoint 2007 (Finkelstein, 2006).

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8.2.6   *Step 5 Statistical analysis*

Non-parametric Mann-Whitney U-tests (Lehmann, 2006), that compare two data sets, were used to test whether there was a significant difference between the plantations and fynbos sites, between the invasive Poaceae and Ericaceae dominant fynbos, as well as between biotopes with younger than 10 year and older than 10 year old alien pine trees. The non-parametric Kruskall-Wallis test, for comparing more than two data sets, was used to test for significant differences between all types of fynbos using Microsoft Office Excel 2007 (Reding & Wermers, 2008).

Piechard evaluation was done to illustrate the invertebrate composition changes through the seasons for Jonkershoek, with Chi-square analysis to test if there is significant variation between seasons.

8.2.7   *Additional analysis of specific sites*

Estimated species richness was calculated making use of Chao index of estimation (Colwell, 2011) for each site. Diversity was estimated with Simpson-Yule index (D =1-C version). All calculations were done with Microsoft Office Excel 2007 (Reding & Wermers, 2008).

Ants were selected as bioindicators due to most specimens could be identified to at least genus level and having stronger correlation of relative abundance with various environmental variables (i.e.: fynbos type; soil pH levels; aspect, slope; soil type; distance for nearest forest, water body/river, road, foot trail; percentage canopy cover at foot, knee, hip and head height; and percentage leaf litter). I then tested regression for each morphospecies against each environmental variable within Microsoft Office Excel 2007 (Reding & Wermers, 2008), to identify possible bioindicators for each of the disturbance types present within the Jonkershoek valley.

8.3   RESULTS

A total of 74 673 specimens was sampled (Figures 8.3.1.1-5). Hymenoptera, especially ants, was the overall dominant taxon in the pooled dataset, although springtails dominated the winter,

while beetles were dominant in spring. Ants showed a 73.6% decline in abundance during winter ($\aleph^2$=37.24, df=3, p<0.01), while springtails increased by 345% during winter ($\aleph^2$=9.44, df=3, p<0.05). Beetles showed the greatest variance between seasons, with an increase in abundance of 2174% from March to September ($\aleph^2$=11.5, df=3, p<0.01). Crustaceans, although low in abundance, showed a significant increase over summer ($\aleph^2$=11.5, df=3, p<0.01). All taxa, with the exception of ants (winter) and orthopterans (spring) were lowest in abundance during autumn.
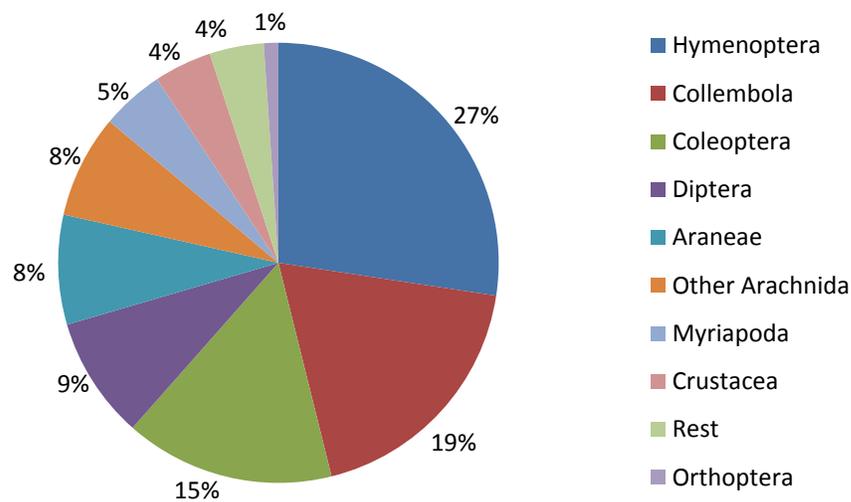
Figure 8.3.1.1 Relative abundance of arthropods from the pooled data combined for the whole year
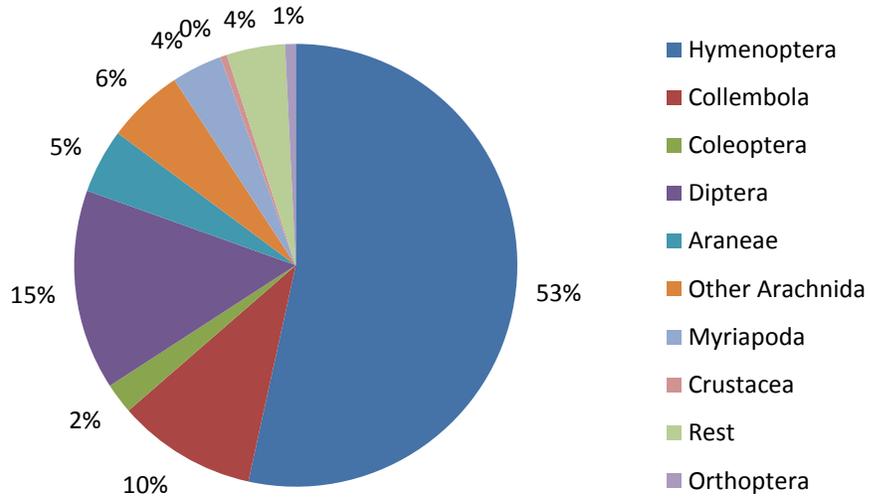
Figure 8.3.1.2 Relative abundance of arthropods in March 2006
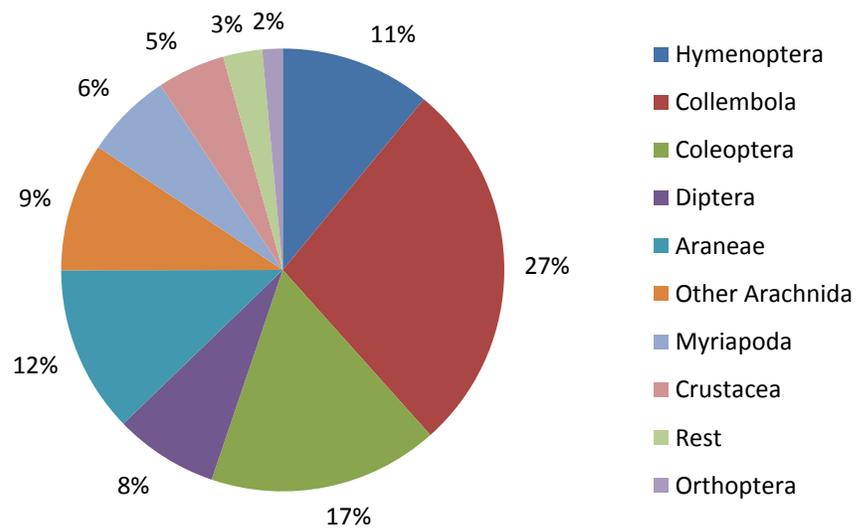
Figure 8.3.1.3 Relative abundance of arthropods in June 2006
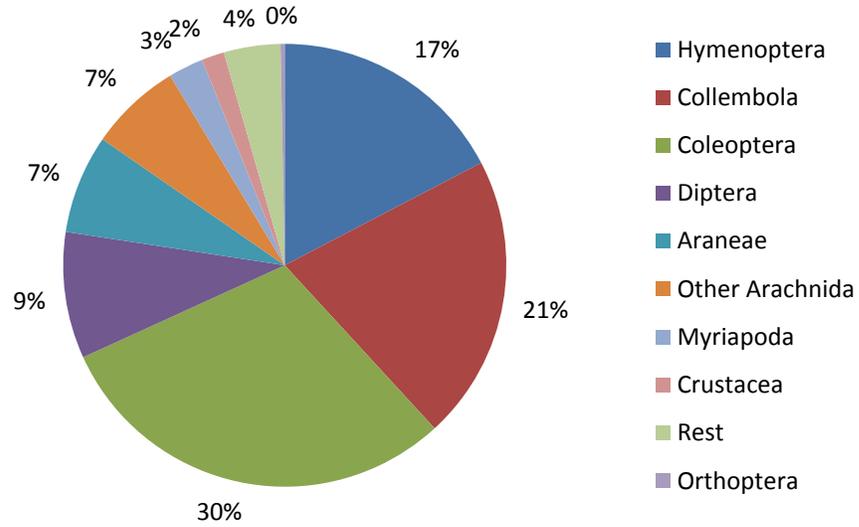
Figure 8.3.1.4 Relative abundance of arthropods in Septermber 2006
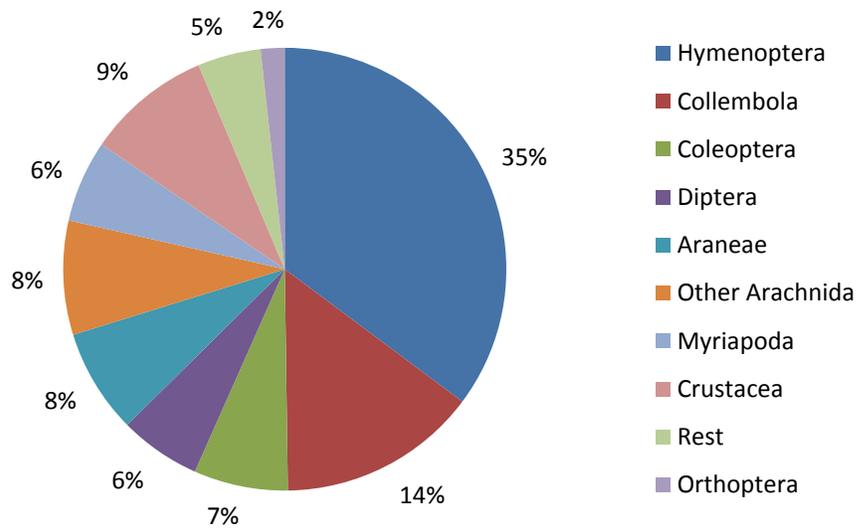
Figure 8.3.1.5 Relative abundance of arthropods in December 2006

A total of 294 morhpospecies showed positive correlation to disturbance levels, and these were used in the calculation of the *BQI* values for each site. Figure 8.3.2 illustrates the geographical distribution of the *BQI* values according to site location. The older pine plantations had the lowest *BQI* values. Within the older pine plantation category, the sites closest to border with the nature reserve and next to the river, had the lowest *BQI* values. Generally, for all biotope types, with the exception of Proteaceae dominated fynbos, the farther away from the river the higher the *BQI* values. Sites farther away from the circular road around the park also had higher *BQI* values. The younger pine plantations (<10 year old) had increased *BQI* values with increase height into the valley, and farther from the agricultural areas.
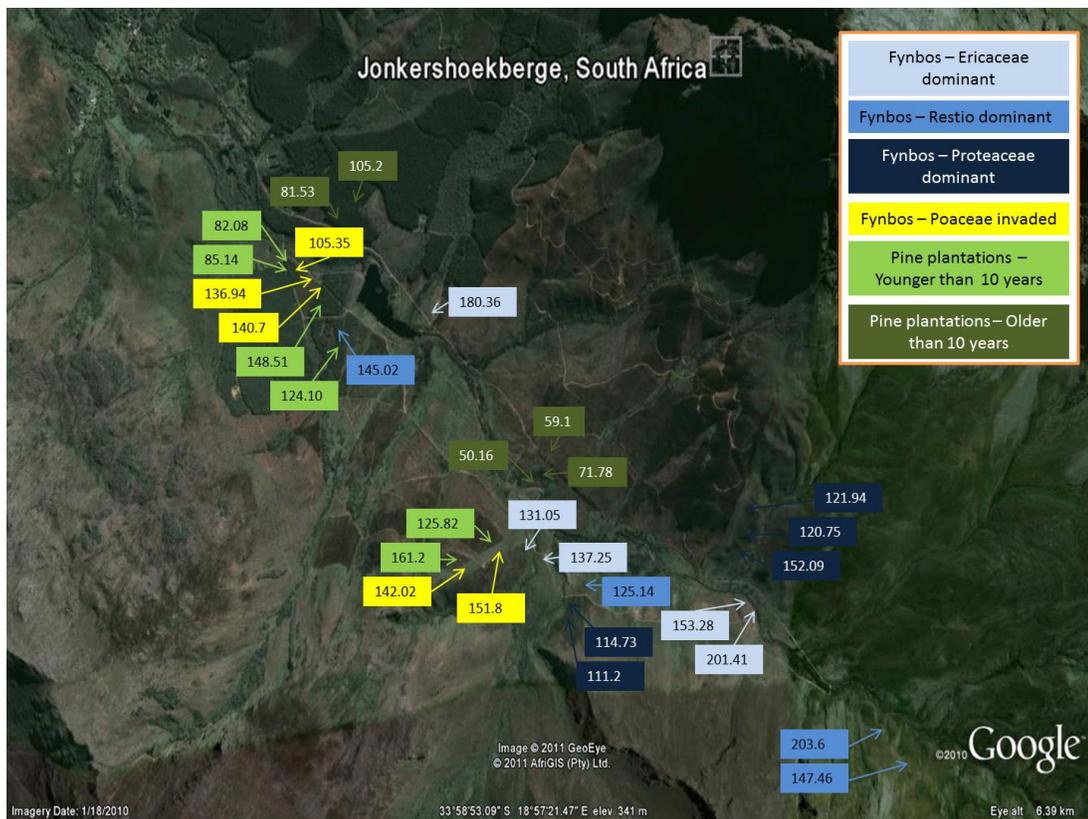
Figure 8.3.2 Geographical distribution of *BQI* values calculated for each site sampled (Aerial photo of Jonkershoek Valley, downloaded from GoogleEarth (GoogleEarth, 2011))

The highest value was in the Restio-dominant fynbos, situated farthest from urbanization. The Erica dominated fynbos showed by far the best overall *BQI* values. Proteaceae dominated

fynbos had relatively low values for pristine fynbos sites, but still higher than for the pine plantations.

Fynbos biotopes, with much higher values showed significant differences towards the pine plantations (p<0.001; U=29; n_a=14; n_b=11).   Between the young and old pine plantations the younger pine plantations showed higher values that proved to be significant different from the older plantations(p<0.001; U=2; n_a=6; n_b=5). The Erica dominated fynbos, seen as the more pristine sites, showed higher scores and differed significantly from the the Poaceae invaded fynbos (p<0.001; U=7; n_a=5; n_b=5). Comparison of all three categories of fynbos sampled also showed a significant difference, with the Proteaceae dominated fynbos having much lower values than rest (p<0.001; H=56.29; df = 2).

Table 8.3.3 shows the comparisons between the disturbance levels, *BQI* values, estimated species richness, as well as diversity as measured by various diversity indices. Ericaceae dominated fynbos showed the highest *BQI* values, correlating well with the lowest disturbance levels. The Proteaceae dominated fynbos had the highest diversity and species richness. The older pine plantations had the lowest values for all four categories.

Table 8.3.4 a & b illustrates the correlation between various ant species and environmental variables. The strongest correlations were between the ant genera *Pheidole,* as well as *Camponotus* species, which were most abundant in biotopes at some distance from pine plantations.

Table 8.3.3 Comparison of the disturbance level, *BQI* values, estimated species richness and Simpson-Yule diversity for each site over the whole sampling period

Site	South	East	Disturbance level	*BQI* value	Estimated species richness	Simpson Yule diversity
Fynbos - Ericaceae dominated (knee-higheight)			**2.32**	**160.67**	**181.28**	**0.06**
C.v	33°59'11"	18°57'13"	4.34	131.05	172.38	0.07
C.vi	33°59'12"	18°57'14"	4.43	137.25	211.75	0.05
E.i	33°59'25"	18°58'11"	1.61	153.28	168.00	0.06
E.ii	33°59'26"	18°58'12"	1.02	201.41	157.20	0.08
I.i	33°58'36"	18°56'51"	0.22	180.36	197.05	0.05
Fynbos - Restionanceae dominated (hip-height)			**2.75**	**155.30**	**172.95**	**0.07**
B.i	33°59'03"	18°56'38"	3.60	145.02	203.00	0.07
D.iii	33°59'15"	18°57'20"	2.50	125.14	153.70	0.05
F.i	33°59'40"	18°58'32"	1.85	203.60	190.88	0.07
F.ii	33°59'42"	18°58'29"	3.04	147.46	144.20	0.10
Fynbos - Proteaceae dominated (head-height)			**3.47**	**124.14**	**182.40**	**0.15**
D.i	33°59'18"	18°57'19"	4.50	114.73	191.25	0.07
D.ii	33°59'22"	18°57'18"	5.20	111.20	210.00	0.08
G.i	33°59'19"	18°58'08"	1.25	152.09	202.96	0.18
G.ii	33°59'18"	18°58'08"	3.22	120.75	157.57	0.15
G.iii	33°59'17"	18°58'09"	3.16	121.94	150.20	0.28
Poaceae invaded Fynbos			**4.05**	**135.36**	**177.98**	**0.08**
A.iii	33°58'31"	18°56'19"	5.99	105.35	192.57	0.15
A.iv	33°58'31"	18°56'21"	5.47	136.94	168.44	0.04
A.v	33°58'39"	18°56'36"	2.89	140.70	181.73	0.07
C.iii	33°59'13"	18°57'11"	3.60	151.80	192.57	0.06
C.iv	33°59'14"	18°57'10"	2.30	142.02	154.57	0.08
Pine plantation (< 10 year old)			**5.05**	**121.14**	**174.60**	**0.07**
A.i	33°58'13"	18°56'38"	6.60	82.08	243.03	0.07
A.ii	33°58'14"	18°56'34"	6.60	85.14	159.11	0.09
A.vi	33°58'41"	18°56'36"	3.30	148.51	150.38	0.06
B.ii	33°59'04"	18°56'57"	6.40	124.10	170.38	0.05
C.i	33°59'12"	18°57'11"	4.20	125.82	162.17	0.06
C.ii	33°59'13"	18°57'09"	3.20	161.20	162.53	0.06
Pine plantation (> 10 year old)			**7.21**	**73.56**	**128.47**	**0.06**
H.i	33°59'07"	18°57'28"	8.00	50.16	137.55	0.06
H.ii	33°59'07"	18°57'27"	6.90	71.78	129.00	0.05
H.iii	33°59'08"	18°57'26"	8.17	59.10	151.22	0.07
J.i	33°58'21"	18°56'33"	7.80	81.53	128.23	0.05
J.ii	33°58'18"	18°56'34"	5.20	105.20	96.33	0.08

Table 8.3.4a Significant correlations between relative abundance of ant species and environmental variables

Species	R^2 value	Trend	Species	R^2 value	Trend
INDICATE VERY STRONG CORRELATION					
DISTANCE FROM FOREST					
Pheidole sp. 1	68%	Positive			
Pheidole sp. 2	63%	Positive			
INDICATE STRONG CORRELATION					
FYNBOS TYPE			DISTANCE FROM FOREST		
Formicidae (pooled data)	56%	Positive	*Camponotus* sp.1	49%	Positive
Camponotus sp. 2	46%	Positive			
INDICATE CORRELATION					
ASPECT			SOIL pH LEVEL		
Plagiolepis sp.1	39%	Negative	*Pheidole* sp.1	33%	Negative
Linepithema humile	25%	Negative	*Pheidole* sp. 2	24%	Negative
Tetramorium quadrispinosum	21%	Negative	*Tetramorium* sp.6	21%	Negative
SLOPE			SOIL TYPE		
Formicidae	39%	Positive	*Tetramorium* sp.6	35%	Negative
Monomorium sp. 1	30%	Positive	*Ocymyrmex* sp.1	22%	Positive
Pheidole minor	27%	Positive			
Pheidole major	26%	Positive	% LEAF LITTER		
% COVER AT HEAD HEIGHT			*Ocymyrmex* sp.1	30%	Negative
Camponotus sp.2	29%	Positive			
Pheidole sp.2	28%	Positive	% COVER AT KNEE-LEVEL		
Tetramorium sp.4	27%	Negative	*Ocymyrmex* sp.1	35%	Positive
Tetramorium quadrispinosum	20%	Negative	*Tetramorium quadrispinosum*	33%	Positive
% COVER AT HIP-HEIGHT			*Tetramorium* sp.6	21%	Negative
Tetramorium sp.6	22%	Negative	*Crematogaster* sp.1	20%	Positive

Table 8.3.4b Significant correlations between relative abundance of ant species and environmental variables

Species	R^2 value	Trend	Species	R^2 value	Trend
INDICATE CORRELATION					
PLANT HEIGHT			% COVER ABOVE 2m		
Tetramorium sp.6	32%	Positive	*Ocymyrmex* sp.1	34%	Negative
Formicidae (pooled data)	28%	Negative	*Tetramorium quadrispinosum*	24%	Negative
Ocymyrmex sp.1	22%	Negative	*Tetramorium* sp.6	22%	Positive
DISTANCE TO ROAD			DISTANCE FROM PINE PLANTATION		
Pheidole minor	21%	Positive	Formicidae (pooled data)	34%	Positive
			Tetramorium frigidum	28%	Negative
DISTANCE FROM RIVER			*Rhoptromyrmex transversinodis*	27%	Positive
Camponotus sp.2	27%	Negative	*Camponotus niveosetosus*	27%	Positive
Camponotus sp.1	25%	Negative	*Solenopsis punetaticeps*	26%	Positive
			Camponotus sp.3	25%	Positive
DISTANCE FROM FOOTTRAIL			*Technomyrmex pallipes*	24%	Positive
Tetramorium sp.6	20%	Positive	*Lepisiota* sp.1	23%	Positive
			Camponotus sp.2	22%	Positive
FYNBOS TYPE			*Linepithema humile*	20%	Negative
Pheidole sp.2	32%	Positive			
			PRESENCE OF PINE PLANTATION		
Solenopsis punetaticeps	28%	Positive	*Tetramorium sp.*6	39%	Positive
Camponotus sp.1	24%	Positive	Formicidae (pooled data)	32%	Negative
Lepisiota sp.1	23%	Positive			
Pheidole sp.1	20%	Positive			

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8.4   DISCUSSION

The cool winter rainy season is less suitable for highly active arthropods, although springtails can increase in abundance with the increase of moisture (Bokhorst *et al.,* 2008). Ants showed a large decrease during winter, especially the invasive species *Linepithema humile*, which stores food for the colder months (Gordon, 2010). This strategy is more suitable for biotopes within summer rainfall regions, and thus the ant's internal clock might be the reason for them not spreading farther, as well as not making full use of the time when resources are most common within the fynbos biome. Winter is also the recovery season in the fynbos biome (Wilson *et al*., 2010). Seeds germinate, and young plants flourish, especially in the spaces opened up from the previous season's fires. As these plants grow, niches that have been temporarily destroyed become available again. Beetles in particular seem to have strategies best suited to take advantage of these changes. Their adult numbers increase substantially with the new plant growth and with the appearance of flowers. Beetle larvae are also most active during winter when resources are abundant. This life stage strategy results in emergence of adults during spring, so they can lay eggs before the summer months that have the potential of fire (Procheş & Cowling, 2006).

Summer is the fire season in the fynbos biome (Wilson *et al*., 2010), and when plant matter is highly flammable owing to the presence of high wood biomass and sclerophyllous leaves and stems, containing tannins, resins and essential oils (Specht, 1979). Most arthropods do not have the ability to outrun fire, and thus have adapted their life strategies accordingly (van Wilgen, 2009). My data indicate that ants are numerous during this time, and are not impacted by the effects of fire, owing to their reproductives being safely underground with stored resources and their young (Agosti *et al.,* 2000). Ants continue to be abundant into autumn, while other taxa decrease due to the scarcity of resources. Ants are also seen as pioneers entering disturbed areas first (Costa, 2010). After summer, fires within a biotope, ants are the taxon with above 50% presence within pitfall traps.

The *BQI* values were lowest in the biotopes adjoining the agricultural areas (especially the young pine plantations nearest to the vineyards). There was also a correlation between the *BQI* values with age of the plantation, indicating a succession process. When pine trees increase in size

and demand more light and water, fynbos vegetation is outcompeted (Cowling *et al*., 2009). Pine trees also change soil chemistry and composition, as well as creating a leaf litter layer that moves the ecosystem away from typical fynbos conditions (Jackson *et al*., 2009). Inevitably, these changes impact on the arthropod assemblages, as seen here. Grasses are not naturally dominant plants in the fynbos types found within our sample sites (Mucina & Rutherford, 2006). This is clear by their dominance only when there is human disturbance within the biotopes. The *BQI* data here show significant differences in quality between the grass invaded fynbos and the natural Erica dominated fynbos. The grass invaded fynbos showed significantly lower *BQI* values than the rest of the natural biotopes. This might indicate that the arthropod populations are still synchronized with the fire cycle of the fynbos biome (van Wilgen, 2009) and naturally move to biotopes with less biofuel compared to high-risk sites with a high biomass of dry vegetation. However, the grass invaded fynbos had not burned for many years, and had an excess of biofuel, indicated by a low *BQI*.

The *BQI* data clearly indicate an edge effect taking place, where biotopes closest to the circular road that surrounds the riparian vegetation of the river, influenced the abundance of the arthropod populations and thus the environmental quality. This might be due to pollution types in the form of air, runoff, noise and chemical.

8.5   CONCLUSION

My data indicated that the agricultural activities and tourism within the Jonkershoek valley might have an effect on biotope quality. Furthermore, that afforestation of pine plantation does lower the quality of a biotope within the fynbos. This case study shows that the *BQI* is an useful management tool in evaluating environmental conditions and biotope quality.

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Chapter 9

CONCLUSION

9.1   INTRODUCTION

Society's conceptualisation of the consequences of biotic impoverishment, and of the extent to which the human species depends on healthy biological systems, is rudimentary at best (Karr, 2008). Therefore, it is essential that this generation implement the basic structure for conservation, restoration, monitoring and management to optimise the current optimal environmental conditions for humanity into the future. A good starting point is to conceptualise the notion of environmental health and from this foundation, implementation can begin to take place.

In chapter 2 (p. 12-46) I establish a final definition for ecosystem health as:

An ecosystem is healthy if it can sustain an optimal number of species with optimal population size and sustain their ecological processes, thus providing an optimal heterogeneous sustainable system with sufficient resources, and indicate adequate resistance when under perturbational stress, but still allowing natural succession to take place.

This definition allows us to further develop a more meticulous taxonomy of ecosystem perturbations or ills, one that can provide a basis for calculating statistics on the success of various efforts at rehabilitation (Jørgensen *et al*., 2010). Humanity's ability to conserve ecological health will be determined by our ability to identify the roots of the biodiversity crisis, which is found in the proliferation of 'disease" found in many forms of human injustice (Angermeier & Karr, 1994).

For us to survive into the future, we need to adjust Gifford Pinchot's consumption-oriented resource conservation ethic, which calls for harvesting our surroundings, to provide the greatest good for the greatest number of people for the longest time (Karr, 2008). When harvesting our surroundings we must consider providing the greatest good for the greatest number of species, for the longest time.

Biological monitoring using arthropods has many advantages. Arthropods being mobile and sensitive to changing conditions, often respond by moving away from, or towards, adverse or optimal conditions respectively (Samways *et al*., 2010). Arthropod populations are often very large, so that sampling a few hundred individuals will not

negatively impact the population and provide significant amounts needed for proper statistical analysis (Mills, 2007). Arthropods are distributed worldwide in a wide range of habitat types, with most taxa being represented (Price *et al*., 2011). A bioindicator species that proves to be successful in one part of the world can often direct future studies to find similar species in the same taxon in a similar environment in another part of the world.

Most arthropods are small and show a diversity of growth rates, life history styles, body sizes, food preferences, and ecological preferences, and can at the species level, often be linked to specific environmental variables (Price *et al*., 2011). The smallest environmental change can thus affect their populations. These specializations enable studies on a specific taxon according to the required solution. Arthropods have in most cases fast breeding rates and short generation times, and therefore are often highly responsive numerically to environmental change (McGeoch, 2007). Environmental changes may be subtle and a result of complex interactions between abiotic and biotic components that cannot be measured directly. But these variables, can be seen within the behaviour or mortality of these species. Sometimes, the biotic community continues to change long after physical or chemical traces of the impact are no longer measurable, but the effects can still be seen in the arthropod community (McGeoch, 2007). Arthropods are often easy and safe to sample. No elaborate equipment is usually required, making the use of arthropods as bioindicators more cost effective compared to numerous other methods and taxa.

My literature review identified that Ephemeroptera, Plecoptera, Trichoptera and Diptera, are useful taxa as bioindicators in the aquatic environment (Azrina *et al.,* 2006; Boonsoong *et al.,* 2009; Imoobe & Ohiozebau, 2010; Paparisto *et al.,* 2010). Odonata make good indicators of the riparian environment (Simaika & Samways, 2009). In the marine environment Crustaceans, especially the orders Isopoda, Amphipoda and Decapoda are the most useful as bioindicators (Nahmani & Rossi, 2003; Anderson *et al.,* 2006; Muenz *et al.,* 2006).

In the terrestrial environment, spiders in the predatory guild (Gollan *et al.,* 2010), orthopterans in the herbivore guild (Jana *et al.,* 2006), springtails in the decomposer guild (Greenslade, 2007), butterflies in the pollinator guild (Zografou *et al.,* 2009), and ants as scavengers (Andersen *et al.,* 2002), make good bioindicators. Beetles have many good bioindicator species due to their high diversity and wide distribution.

Research is not conservation biology if it has no practical implications for conservation (McGeoch, 1998) and with ever-increasing pressure on natural resources (Fowler, 2008), it is important to have reliable methods for evaluating the health of a specific biotope. It is against this background, I set out to develop the Biotope Quality Index (*BQI*), a measure of environmental quality of a given biotope, and to look at the long-term effects of environmental impacts on a specific biotope, or comparison between biotope in the same area for better conservation management practices. The *BQI* proposed here is not only a rapid, but also an affordable means of doing environmental evaluations, as well as an easily adaptable method across biotope types, taxa and sampling protocols.

9.2   BIOTOPE QUALITY INDEX: CONCLUSIONS

My study established that the *BQI* is much more suitable than any of the diversity indices currently being used by environmental impact assessors to evaluate environmental health. The *BQI* correlated much more strongly with the level of disturbance than any of the diversity indices, mainly as the diversity indices do not measure ecosystem quality but quantitative species composition.

Acar and Troutt (2008) identified three deficiencies with existing diversity indices, causing insufficient discrimination among related but dissimilar situations.   Firstly, they lack normalization, which is corrected for in the *BQI* by each species being correlated with the mean population size of that same morphospecies.   Secondly, diversity indices lack calibration, corrected for in the *BQI* by using the same scale in the form of the same morphospecies and methodology used at all sites sampled. Finally, most indices are non-linear while the *BQI* shows results on a linear scale making it suitable for comparison of sites. Furthermore, both the number of morphospecies, as well as the number of replicates of sites used, will determine the quality of the fit to the real situation, meaning that more is better (Thomson *et al*., 2007).

I furthermore evaluated how the *BQI* might differ if we use only a specific guild, and found that the *BQI* evaluation of beetles as broad spectrum control group (consisting members of all guilds) fared poorly against its comparison in the guild divisions. The foraging (ants) and decomposer (springtails) guilds stood out as the most promising bioindicators, while the predator guild (spiders) showed negative correlation to the level of disturbance.

I recommend that before assessment, that a proper set of bioindicators for a specific biome, and an efficient trapping method is identified. A simple correlation test of morphospecies abundance versus a non-parametric index for the level of disturbance will indicate strong positive correlation (>33.33%), weak positive/negative correlation (0-33.33%), and strong negative correlation (>33.33%). I recommend the selection of only the morphospecies that indicate strong positive correlation (>33.33%) for use within the *BQI*. This step will insure that alien species that prosper within disturbed areas are excluded, for example *Linepithema humile,* to prevent misleading results.

Seasonal change seems to also have an effect on the *BQI* data. Therefore, I recommend that for comparative studies, sampling must take place preferably in the same season. Furthermore, the pooled data for the whole year gave a better correlation than the individual seasons, and so, if possible, data must be collected over all seasons and pooled for best results.

Of the seasons, autumn (March) sampling, at least in the Cape Floristic Region, can be recommended, because it gave the best correlating *BQI* results, at least for pitfall sampling. This might be different for another region in another environment, or if different taxonomic groups are chosen as bioindicators, as well as if a different sampling method is used.

9.3   SUGGESTION FOR A PRACTICAL METHODOLOGY

9.3.1   *Step 1 – Selecting the study area*

Identify sites that need assessing within the study area (preferably an even spread over the whole study area). Create a map showing the distribution of these sites. Ground truthing will provide better understanding of the boundaries of the biotopes, as well as ease of accessibility to sites. Make use of the ground truthing to identify and group your biotope types, preferably by vegetation. Mucina and Rutherford's (2006) *Vegetation of South Africa, Lesotho and Swaziland* is recommended for consultation for southern African conditions. The larger the number of sites, the better the indication of the current situation, as the results also will be more statistically significant. I recommend the use of species accumulation curves, to determine whether significant number of morphospecies have been included to assess the situation.

9.3.2   Step 2 – Determining the level of disturbance

Evaluate each site according to integrity and disturbance levels, by rating biotopes according to *a priori* range of disturbance levels from near-natural conditions to least-natural conditions. I recommend use of as many evaluators as possible and use the mean value of their judgements.   Categories to consider are pollution levels, invasive alien presence, distance from nearest disturbance, human activity within area, biotope age and biotope structure, among others.   These might be subjective data, but because of the lack of a successful method for measuring biotope quality, this non-parametric *a priori* method gives data where assessors can correlate the relative abundance of the morphospecies to identify the specimens that show positive correlation.   Also, it is important to bear in mind that disturbance level has a reverse scale, and so the normal statistical negative slope in the data is seen as a positive correlation when analysing data.

9.3.3   Step 3 - Field sampling

From evidence in the literature, choose a suitable taxonomic group(s) to use as a bioindicator(s), in addition to a sampling methodology that best fits the selected taxonomic group and hypothesis being tested. Plan a schedule that allows sampling of all sites, preferably within the same biome type in similar seasonal and weather conditions. It is advisable to undertake data collection over all seasons to improve results.

9.3.4   Step 4 – Testing for correlation

All specimens are sorted to morphospecies level, and then their individual abundance levels are used to test for correlation against the level of disturbance. This is done to identify only the morphospecies that show positive correlation (an increase in disturbance results in decrease of specimens).

9.3.5   Step 5 - Calculation of the BQI value for each site

The morphospecies identified in step 4, are then used to calculate the *BQI* value for each site making use of the formula given in chapter 4 (p.111-131).

$$ EQI \ = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \bar{x}_i}{\log \sigma} \right) \ \text{when} \ \frac{\log x_{ij} - \log \bar{x}_i}{\log \sigma} > 0 $$

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Morphospecies that occur only as singletons across all sites, as well as species with the same abundance levels in all sites can be deleted from the data set to ease calculations, as none of these species contribute anything meaningful to the sensitivity of the index.

9.3.6 Interpretation of BQI evaluation

The data retrieved from the *BQI* evaluation then provide a scale that can be used to compare biotope quality between sites, as well as over time. The biotope with the higher value shows higher biotope quality. Where the effect of a given procedure is tested, a higher value over time indicates successful restoration compared to a lower value indicating a negative effect of a disturbance. Plotting the *BQI* values on a map can indicate an effect vector, where lowest values can be found nearest to disturbance with the greatest effect on the environment. Data can further be used in non-parametric analysis to test for significant differences between biotopes.

9.4 CONCLUSION

Environmental engineers and conservationists can use the *BQI* as an evaluation tool to determine if some adjustment (e.g. the construction of a road) had a negative effect on the environment with before and after assessments. Restoration projects would have a measurement to evaluate success against and with which to set goals. Conservationists would also be able to identify biotopes with lower *BQI* scores, compared to the rest of their conservation area, and start looking for what might have a negative effect within these biotopes. This was the case in the case study (Jonkershoek), where an environmental impact assessment making use of the *BQI* evaluation, indicated that the agricultural disturbance and tourism within the Jonkershoek valley can have negative effects on biotope quality. Furthermore, afforestation with pine plantations seems to lower the quality of a biotope.

I also established that the *BQI* are a much better index of ecosystem health or biotope quality than the diversity indices normally used for biotope evaluation as surrogates for biotope quality.

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10.1.1 Appendix 1: Glossary

Abundance   Number of specimens or organisms within a population (Cotgreave, 2002).

Accumulators   Organisms which take up and accumulate chemicals in measureable quantities, e.g. heavy metals in Carabidae (Butovsky, 2011).

Actual abundance   The number of specimens within a population where all the specimens within the system can be counted or are known (Cotgreave, 2002).

Anoxic conditions   A condition or an environment that lacks oxygen (Choudhary, 2003).

Assemblages of species   The combinations of species that co-occur at the same time and place (Cotgreave, 2002).

Autocatalysis   An agency that drives ecosystem development (the increase of organization) (Ulanowicz, 1997)**.**

Autopoiesis   A system organized as a network of processes of production of components that produces the components which: through their interactions and transformations continuously regenerate and realize the network of processes that produce them and constitute it as a concrete unity in the space in which they exist by specifying the topological domain of its realization as such a network (Maturana & Verela, 1980).

Bioassay organisms   Selected organisms used as laboratory reagents to detect the presence and/or concentration of pollutants, or to rank pollutants in order of toxicity, e.g. in sediment pollution **(**De Lange *et al.*, 2004).

Biodiversity   The degree of variation of life forms within given species, ecosystem, biome, or an entire planet (Stiling, 1999).

Biodiversity indication   A group of taxa or functional group, the diversity of which reflects some measure of the diversity (e.g. character richness, species richness, level of endemism) of other higher taxa in a habitat or set of habitats (McGeoch, 2007).

Biogeography   Analysis and understanding of spatial biological phenomena in terms of past and present factors and processes (Hengeveld, 1992).

Bioindicators   A species or group of species that readily reflects the abiotic or biotic state of an environment, or represents the impact of environmental change on a habitat, community or ecosystem (McGeoch, 2007).

Biological indicator   Detection and monitoring of changes in biota to reflect changes in the environment (Wilhm & Dorris, 1968). A biological species or groups of species whose function, population or status can reveal what degree of ecosystem or environmental integrity is present (Karr, 1981)

Biome   A major type of natural vegetation that occurs wherever a particular mix of climatic and edaphic conditions is envounter (Ladle & Whittaker, 2011).

Biomonitoring   Repeated application of bioindicator taxa to provide information on the environmental conditions, or effects thereof, to which they were initially identified as correlating, sensitive and for which baseline standards, thresholds or relationships have already been determined (McGeoch, 2007).

Biotope   The smallest geographical unit of the biosphere that can be delimited by convenient boundaries and is characterized by its biota (Lincoln *et al*., 1998).

Biotope quality   A measure of the status and condition that the biotope is in based on the concept of Environmental Health (Rapport, 1992).

Carnivore   An animal or plant that requires a staple diet consisting mainly or exclusively of animal tissue trough predation of scavenging. (Boitani, & Powell, 2012) **.**

Classical competition theory   Process controlling community structure is competition where limited resources are essential for coexistence of all species (Pickett *et al*., 2007).

Climax theory   A community is at its best when the successional sequence has maximal biomass, the most complex nutrient cycles, greatest productivity, greatest species diversity, and is able to maintain itself indefinitely when freed from major disturbances (Bratton, 1992).

Coarse-filter ecosystem management   Management strategies that aim to maintain entire communities by protecting large extents of habitat and by preserving the key ecological components and processes that sustain these communities (Carignan & Villard, 2002).

Community   A naturally occurring aggregation of organisms belonging to a number o different species, occupying a common ecosystem and interacting with each other within this area (Morris, 1992).

Complex system   A system that are made up of a number of interacting parts, components of which vary in their type, structure and function within the whole system (Costanze, 1992).

Connectance strength (food web)   The number of links represented within a real or model web, as a proportion of the number of topologically possible (Puttman, 1994).

Consumption-oriented resource conservation ethic   Harvesting our surrounding, to provide the greatest good for the greatest number of people for the longest time (Meffe, 2000).

Decomposers guild   Specimens that feed on dead organic matter (Swift *et al*., 1979).

Degradation   Reduction in the ecosystems' actual or potential uses (Yudav, 2007).

Density independence theory   Non-equilibrium theory whereby the environmental transformation and populations densities are not always high enough for competition or for other biotic processes, such as significant predation or parasitism (Price, 2003).

Detectors   Species occurring naturally in the study area and which may show a measurable response to environmental change, e.g. changes in behaviour, mortality and age class structure (Nash *et al.,* 1998).

Diapause   A definite preparatory phase usually initiated by a temperature-independent factor, involving metabolic changes (Mansingh, 1971).

Disturbance level   An estimation of degradation taken place within an ecosystem seen as the inverse of integrity according to the triage concept (Samways, 2005).

Diversity   See **Biodiversity.**

Diversity indices   Statistical analyses which are intended to measure the differences among individuals of a data set consisting of various types of objects (Cover & Thomas, 1991).

Ecology   The scientific study of the relationships tha living organisms have with each other and within their abiotic environment (Stilling, 1999)

Ecological indicator   A species or a group of species that demonstrate the effects of environmental change (such as habitat alteration, fragmentation and climate change) on biota or biotic systems (McGeoch, 2007).

Ecological integrity The capacity of an ecosystem to support and maintain a balanced, integrated adaptive community of organism, having a species composition, diversity and functional organization comparable to that of similar undisturbed ecosystems in the region (Caringnan & Villard, 2002).

Ecological succession   A process of directional change in the vegetation and animal life of a particular area possibly leading to a more stable community which is referred to as the climax (Scharler, 2009).

Ecoregion   Regions of relative homogeneity in ecological system or in relationships between organisms and their environments (Ladle & Whittaker, 2011)

Ecosystem   A biological community of interacting organisms and their physical environment (Price *et al*., 2011).

Ecosystem distress syndrome   Situation whereby disturbances can offset the system completely, resulting in an irreversible process of systems decline leading to total destruction, characterized by reduced primary productivity, loss of nutrient capital, loss of species diversity, dominance by short-lived, opportunistic, and often exotic

species, increased fluctuation in key populations, retrogression in biotic structure and increased incidence of disease (Jørgensen *et al.*, 2010).

Ecosystem function   Any ecosystem level attribute that can be measured in and compared between ecosystems (Weisser & Siemann, 2004).

Ecosystem health   An ecosystem is healthy if it can sustain an optimal number of species with optimal population size and their ecological processes, thus providing an optimal heterogeneous sustainable system with sufficient resources, and indicate adequate resistance when under perturbational stress, but still allowing natural succession to take place (*There are various definitions in the literature review within the thesis that differ and therefore, I concluded with my own definition that took all the other definitions in consideration as a base for the development of the BQI*).

Ecosystem integrity   See **Ecological integrity.**

Ecosystem management (as a management category)   A management style with explicit goals which are executable by implementation of policies, protocols and practices (Carignan & Villard, 2002).

Ecosystem quality   The status of the ecosystem based on its natural functioning (Woodley *et al.*, 1993).

Ecosystem resilience The capacity of ecosystems to absorb disturbance (Ladle & Whittaker, 2011).

Ecosystem services   The processes of products of natural ecosystems and species that provide a utilitarian value (Ladle & Whittaker, 2011).

Ecosystem structure The components of and interactions present between the physical abiotic environment and the organisms living within an ecosystem (Francis, 2009).

Edge effect   The ecological influence of the altered physical and biotic properties that typically characterize the edge of a habitat type and which extend towards the core of the habitat patch (Ladle & Whittaker, 2011).

Environmental impact assessment Evaluations that looks at the environment as a whole, the diversity, interactions between species, but mostly at the function of environment at a biotope scale to determine the state of its biological integrity (Jørgensen & Fath, 2008).

Environmental indicator   A species or group of species that responds predictably, in ways that are readily observed and quantified to environmental disturbance or to a change in environmental state (McGeoch, 2007).

Environmental stressor   Any physical, chemical or biological entity or scenario that can induce an adverse response that affect the quality of an environment negatively (Truett, 2007).

Environmental surrogates   A physical, biological or climatic variable used as a proxy to derive ecological classification (Ladle & Whittaker, 2011).

Equitability   See **Evenness.**

Eutrophication   The enrichment of a biotope with nutrients, primarily in the form of phosphorus and nitrogen (Conley *et al*., 2009).

Evenness   The pattern of distribution of the individuals between the species (Henderson, 2003).

Exploiters   Species whose presence indicates the probability of disturbance or pollution, e.g. the Argentine ant (Roura-Pascual *et al.,* 2008).   .

Fine-filter ecosystem management   Management approaches that focus on the protection of a certain number of elements, supposing that their status within the ecosystem reflects the status of other elements associated with them (Meffe *et al*., 2002).

Fluctuating environment theory   Non-equilibrium theory whereby the environment changes seasonally or irregularly so that competitive ranking of species change temporally too, and no one species can establish dominance (Lehav, 2004).

Focal group   Specimens that performs a reference role but represents a subset of a larger group of interest selected specifically for its qualities as a predictor set (Hammond, 1994).

Food web   A network of feeding connections in an ecological community (Pascual & Dunne, 2006).

Fungivore guild   Specimens that primarily or solely feed on living fungi (mycophages) (Carlile *et al.,* 2001).

Growth rate   The rate at which the number of individuals in a population invreases in a given time period as a fraction of the initial population (Speight *et al*., 2009).

Guild A group of species that exploit the same class of environmental resources in a similar way (Simberloff & Dayan, 1991).

Habitat   The resources and condition present in an area that produce occupancy by an organism (Egan & Howell, 2001).

Habitat quality   The status of the resources and conditions present that would effect the abundance of the occupancy of the organism (Egan & Howell, 2001).

Health   An ideal state where there is a lack of occurrence of disease, trauma, or dysfunction (Ulanowicz, 1992).

Herbivore guild   Specimens that feed on living plant material (Peeters *et al.,* 2001)**.**

Homeostasis   The property of a system that regulates its internal environment and tend to maintain a stable, relatively constant condition of properties (Trojan, 1984).

Integrity   See **Ecological integrity.**

Interspecific interactions   Interactions between individuals of different species (Speight *et al*., 1999).

Intraspecific interactions   Interactions between individuals of specimens of the same species (Speight *et al*., 1999).

Key group   Specimens used as a basis for extrapolation to a group for which incomplete data are available, while a key group's principal role is to provide a focus for efforts made to document and estimate species richness in a given arena (Hammond, 1994).

Keystone species   A species that has a disproportionately large effect on its environemt relative to its abundnance (Paine, 1995).

Macroecology   A top down and mutli-scale approach to analyses of the structure of biota focused on the emergent outcomes of statistical analysis of key properties in order to understand the processes involved in structuring ecological systems (Ladle & Whittaker, 2011).

Monitoring   The intermittent surveillance carried out in order to ascertain the extent of compliance with a predetermined standard or the degree of deviation from an expected norm (Hellawell, 1991).

Morphospecies   A species defined predominantly by morphological characters (Mahner & Bunge, 1997).

Network ascendency   The product of the aggregate amount of material or energy being transferred in an ecosystem, multiplied by the coherency with which the outputs from the members of the system relate to the set of imputs to the same components (Ulanowicz, 1997).

Niche The potential distribution of a species based on their trophic position and feeding habits (Rockwood, 2006).

Omnivore guild   Specimens that are opportunistic, generalist feeders that consume both plant and animal material as their primary food source (Singer & Bernays, 2003).

Perturbation Change to a system's contribution or environments outside normal range of variation (Costanza, 1992).

Predation extension theory Theory that add predation to the classical competition theory (Pickett, 2007).

Predator guild   Organisms which kill and eat other organisms (Molles, 2002).

Quiescence   The response of individual arthropods to a sudden unanticipated, non-cyclic and usually short duration deviation of normal weather conditions, mainly causing growth retardation (Mansingh, 1971).

Reference group   Specimens used as a basis for extrapolation to a group for which incomplete data are available (Hammond, 1994).

Relative abundance A simple count of individuals within each taxon, and used to compare samples that have been retrieved in the same way (Schneider, 2009), indicating how rare or common a species is relative to other species in a given biotope or community (McGill *et al.*, 2007).

Resilience   The systems ability to maintain structure and patterns of behovior in the face of disturbance (Holling, 1986).

Restoration   The recreation of both the structural and functional attributes of a damaged ecosystem (Newton, 2007).

Season   Season is a division of the year, marked by changes in weather, climate and daylight availability, resulting from the yearly revolution of the earth around the sun and the tilt of the earth's axis relative to the plane of revolution (Khavrus & Shelevytsky, 2010).

Seasonal changes   Changes in temperature, rainfall, wind and humidity (Speight *et al.*, 2009).

Sentinels   Sensitive organisms introduced into the environment, e.g. as early-warning devices or to restrict the effect of an effluent (Gerhardt, 1996).

Slow competitive displacement theory   Non-equilibrium theory that assumes that competitive abilities are similar between species, and although competition is important random variation and chance play a large part in determining the dominant species (Krebs, 2008).

Spatial variation extension theory Adaption to classical competition theory whereby species compete for a single resource but the environment favours different species in different areas, thus it is possible for all specie to coexist in a system of variant patches (Picket *et al.*, 2007).

Species richness   An estimate of the total number of species present in a defined physical area, as determine by some sampling method (Henderson, 2003).

Stable community   Community with where no change can be detected in the population sizes and number of species over a given time period (Beeby & Brennan, 2008).

Stable ecosystem   Scenario when all the variables return to the initial equilibrium after being disturbed (Odenhaugh, 2005).

Standard deviation   Indication of how much variation exists from the average (Gotelli & Ellison, 2004).

Stressed environment   Environment with low levels of equitability as the system becomes dominated by disturbances of pollution tolerant species (Henderson, 2003).

Succession   A concept terminating in production of a climax community which will perpetuate itself generation after generation until altered by some disturbance factor (Khemakhem *et al*., 2010).

Surveillance   An extended programme of surveys undertaken to provide a time series, to ascertain the variability and/ or range of states or values which might be encountered over time (Hellawell, 1991).

Survey   An exercise in which a set of qualitative or quantitative observations are made, usually by means of standardized procedure and within a restricted period of time, but without any preconception of what the findings ought to be (Hellawell, 1991).

Sustainability   A biological system's capacity to endure and remain diverse and productive over time (Stilling, 1999).

Target group   Specimens that are merely a group that is under investigation or the object of attention (Hammond, 1994).

Therapeutic nihilism   Theory whereby the aim is to rather leave the state of illness untreated, and let it go its own way, so that there is either unaided recovery, there is adaption or there is extinction (Hargrove, 1992).

Triage   A system that correlates ecological integrity with the disturbance level and evaluates the necessity and feasibility of restoration (Samways, 2005).

Variability   Variation within the population dynamic of a species (Stilling, 1999).

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10.2   Appendix 2   Example of non-parametric disturbance level rubric evaluation sheet

Pollution

0	No visual signs of any pollution
1	0-5% of biotope covered with pollution
2	5-10% of biotope covered with pollution
3	10-15% of biotope covered with pollution
4	15-20% of biotope covered with pollution
5	20-25% of biotope covered with pollution
6	25-30% of biotope covered with pollution
7	30-35% of biotope covered with pollution
8	35-40% of biotope covered with pollution
9	40-50% of biotope covered with pollution
10	>50% of biotope covered with pollution

Human activities

0	No visual signs of humans ever visiting sites
2	Very rare visits by humans within area (Some form of indication that humans was here before)
4	Weekly visits by humans within area (Tourist footpath with no or very little sign of humans)
6	Daily visits by humans within area (Footpath with lots of footprints within area)
8	Hourly visits by humans within area (Human activity or road for vehicle visible)
10	Permanent human activity in area (Buildings present within area)

Invasive aliens

0	No visual signs of any invasive or alien plant species
1	0-5% of biotope consist of alien and invasive plant species
2	5-10% of biotope consist of alien and invasive plant species
3	10-15% of biotope consist of alien and invasive plant species
4	15-20% of biotope consist of alien and invasive plant species
5	20-25% of biotope consist of alien and invasive plant species
6	25-30% of biotope consist of alien and invasive plant species
7	30-35% of biotope consist of alien and invasive plant species
8	35-40% of biotope consist of alien and invasive plant species
9	40-50% of biotope consist of alien and invasive plant species
10	>50% of biotope consist of alien and invasive plant species

Biotope quality

0	No visual signs of pests or disease on plants
1	0-5% visual signs of pests or disease on plants
2	5-10% visual signs of pests or disease on plants
3	10-15% visual signs of pests or disease on plants
4	15-20% visual signs of pests or disease on plants
5	20-25% visual signs of pests or disease on plants
6	25-30% visual signs of pests or disease on plants
7	30-35% visual signs of pests or disease on plants
8	35-40% visual signs of pests or disease on plants
9	40-50% visual signs of pests or disease on plants
10	>50% visual signs of pests or disease on plants

Distance from nearest disturbance

0	>1000m from nearest disturbance
1	500-1000m from nearest disturbance
2	300-500m from nearest disturbance
3	200-300m from nearest disturbance
4	150-200m from nearest disturbance
5	80-150m from nearest disturbance
6	40-80m from nearest disturbance
7	20-40m from nearest disturbance
8	10-20m from nearest disturbance
9	5-10m from nearest disturbance
10	<5m from nearest disturbance

Growth

0	All plants show signs of new growth
1	90-100% plants indicate new growth
2	80-90% plants indicate new growth
3	70-80% plants indicate new growth
4	60-70% plants indicate new growth
5	50-60% plants indicate new growth
6	40-50% plants indicate new growth
7	30-40% plants indicate new growth
8	20-30% plants indicate new growth
9	10-20% plants indicate new growth
10	0-10% plants indicate new growth

Pollution value	
Invasive aliens value	
Nearest disturbance value	
Human activity value	
Biotope quality value	
Growth value	

Total

Average (Disturbance level)	

234