TAXONOMY, FORM, OR FUNCTION: EVALUATING THREE APPROACHES TO USING CARABID BEETLES (COLEOPTERA: CARABIDAE) AS BIOINDICATORS IN A COASTAL FOREST

by

Sean G. Henderson

B.Sc., University of Northern British Columbia, 2008

THESIS SUBMITTED IN PARTIAL FULFILLMENT OF
THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE
IN
NATURAL RESOURCES AND ENVIRONMENTAL STUDIES
(BIOLOGY)

THE UNIVERSITY OF NORTHERN BRITISH COLUMBIA

August 2010

Your file Votre référence
ISBN: 978-0-494-75806-9
Our file Notre référence
ISBN: 978-0-494-75806-9

Abstract

Arthropods have received considerable attention as biological indicators (bioindicators) as they are widespread, and easy to sample. Due to a shortage of trained taxonomists and limited resources, many taxa cannot be used, however. In this thesis, I use carabid beetles (Coleoptera: Carabidae), which are taxonomically well known, to compare three approaches of bioindication, using a traditional taxonomic approach as the baseline. The taxonomic approach distinguished coarse differences in forest canopy removal, as did a recognizable taxonomic unit approach. The latter failed to provide the same level of precision based on diversity data, however. A functional trait approach yielded non-significant results, but demonstrated potential to provide more in-depth information, albeit at a considerable additional investment of time. All three approaches were found to have advantages and disadvantages, so I conclude that the choice of approach will depend on the availability of resources and the objectives of each bioindication study.

Table of Contents

List of Tables

List of Figures

Acknowledgements

I sincerely thank my supervisor, Dr. Staffan Lindgren for being my academic mentor for the past four years. Staffan provided me with the opportunity to experience research during my undergraduate degree and subsequently allowed me to pursue my love of research through this graduate degree. His patience, understanding, and guidance were incredible throughout the entire process and for this I am very grateful.

I thank my committee member, Melissa Todd, for making me a part of this research project and affording me the opportunity to pursue a Master's degree while working in such a beautiful area. Melissa's enthusiasm, expertise, and easy-going attitude made field work and the entire thesis process absolutely enjoyable.

I am indebted to my committee member, Dr. Chris Johnson, for providing me with quality education from the undergraduate level and continuing to do so at the graduate level. Chris captivated me during my undergraduate years through his lectures and furthered my interest in science, in addition to providing me with early research experience. His input throughout the thesis process has been invaluable.

To the many members of the Coast Region Experimental Arthropod Project team and insect sorters I am forever grateful. Louise Waterhouse, Brian D'Anjou, Jeff Meggs, Jessica Ainsworth, Alexandria Marshall, Brock Harpur, James Leigh, Jordan Lindgren, Stephanie Peesker, and many others, I thank you for your efforts and long hours in making this project happen. I am also grateful to Danny Shpeley and Dr. George Ball for their warm welcome, training, and species confirmations while I visited the Strickland Entomological Museum, University of Alberta.

I thank my family, who are also my best friends, for providing me with a foundation, support, and love that is unparalleled. I thank my Dad and Mom, for being more than a father and mother by also being mentors and friends, and working tirelessly to afford me a life of endless opportunity. I thank my sister Lisa, for always being the assistant while I was the mad scientist, even though it was she who set the academic bar. I thank my sister Jill, for ensuring that I remained a tidy scientist. I thank my brother James, for maintaining my sanity through evening video game sessions. I thank my sister Kim, for reminding me that not everyone loves 'bugs'. I thank my brother-in-law Kris, for becoming a brother to me, and reminding me that not everyone likes 'bugs'. I thank my brother-in-law D.J., for listening to me talk about 'bugs' for half an hour when he asked what I did. I thank my best friends (who I also consider part of my family) Tom, Matt, and Amrit, for ensuring that I remembered to have fun along the way.

Finally, and most of all, I thank my wife, Mandy. She has provided me with every support I could ever ask for and has been beside me every step of the way. She is the kindest, most caring and loving woman in the world to me and I owe so very much of my success to her. I also thank her for learning to love 'bugs', even if it was just for me.

Chapter 1- Introduction

1.1 Introduction and Background

There is an increasing concern over the conservation of biological diversity at a time where already cash-strapped governmental and non-governmental organization's budgets are faced with a global economic crisis. Due to this concern for biodiversity in concurrence with limited funding, efficient methods for assessing the state of ecological function, particularly of anthropogenically disturbed areas, are in high demand. Bioindication offers one potential solution to this growing problem. Andersen (1999; P 61) defines biological indicators or 'biodindicators' as "readily measured components of the biota that are used to provide general information about the complex ecosystems in which they occur". Bioindicators are appealing to researchers and managers as they allow assessment of the impact of disturbance on an ecosystem without sampling all of the biota within that ecosystem, thus resulting in greater cost efficiency (Rainio & Niemelä, 2003).

Conservation biologists urgently need reliable surrogate measures of biological diversity (i.e., bioindicators) (Balmford et al., 1996) in order to conserve areas that are rich in species and/or contain rare species. Bioindication is a means for the identification of areas that should be candidates for conservation through identifying areas of high biodiversity; however, using bioindication for this purpose requires careful consideration as different approaches may provide different results. Traditional approaches to using bioindicators to detect shifts (disturbance or otherwise) in an ecosystem have used long-

lived species such as plants and vertebrates, which may work for long-term monitoring, but fall short when rapid assessments are required (Oliver & Beattie, 1993). This is a by-product of the long generation times associated with these organisms. A change in the ecosystem may not show a direct impact on a plant or vertebrate species until many years later.

Bioindication studies have often fallen short with regard to clearly defining objectives of the study such as what is being indicated, why is it being indicated, and how long is it going to be indicated for (McGeoch, 1998). Furthermore, authors have frequently failed to choose the most applicable or best suited organism, instead opting to use the investigator's own focal organism and justifying its use (Andersen, 1999). McGeoch (1998) sought to improve the practice of selecting bioindicators by outlining a set of objective criteria. These criteria fall into two broad categories. The first is *economic and logistic feasibility*, which includes financial cost, time efficiency, and personnel required. The second category is *biological efficacy*, which includes taxonomic requirements and information available, distributional data, sensitivity to disturbance, representation in a variety of habitats, functional groups, and reliability.

Due to their short generation times, ubiquitous nature, and ease of capture (Oliver & Beattie, 1993), arthropod taxa may satisfy these criteria better than plant and vertebrate bioindicators. However, one of the chief problems when using arthropod taxa as bioindicators is the 'taxonomic impediment'; a lack of stable and well known taxonomy in addition to a lack of trained taxonomic experts (Derraik et al., 2002). Due to the

enormous potential for arthropods as reliable and cost effective bioindicators, several approaches are being explored to overcome these difficulties. The most common approach to date has been to select a taxon that has a stable and well known taxonomy, such as carabid beetles. This approach will henceforth be referred to as the 'taxonomic approach'. I will examine this approach by looking at the species richness, activity-abundance, and evenness of carabid beetles.

An alternative approach is to use recognizable taxonomic units (RTUs or morphospecies) as suggested by Oliver & Beattie (1993). This approach allows non-specialists to sort arthropods into groups based on easily observable characteristics with a low level of training, thus significantly reducing the cost of assessment and monitoring programs. When evaluated against corrected (i.e. an expert corrects a parataxonomist) RTUs and species, RTUs were slightly less reliable, but considerably less expensive (Oliver & Beattie 1996). This approach, henceforth referred to as the 'recognizable taxonomic unit approach', offers a means to bypass the taxonomic impediment and drastically reduce costs. However, reliability must be carefully evaluated to ensure that sufficient resolution is maintained. In order to examine this approach I will measure the RTU richness, activity-abundance, and evenness of carabid beetle RTUs.

The final approach relies on functional diversity as an indicator of biodiversity by examining the range of morphological traits within an ecosystem. This approach has garnered considerable attention recently. Ribera et al. (2001), for example, used morphometric measurements of carabid beetle body structures and appendages that infer

functional adaptations, as well as available information, and related these to environmental variables in order to examine habitat alterations at a very fine resolution. The 'functional diversity approach' may be used as an additional step to the taxonomic approach as it requires the identification of species prior to being used. Functional diversity may offer an increased level of resolution and reliability at an increased cost. This may be useful in instances where minute differences in habitat could translate to long term, larger scale effects on the ecosystem. This approach must be compared through conclusions that may be drawn based on its results to the RTU and taxonomic approaches in order to determine its relative utility.

1.2 Bioindicator Background

Bioindicators are useful as indicators of environmental change (Rainio & Niemelä, 2003). They are measurable components of the biota that are capable of relaying important information about the complex ecosystems in which they are found, such as the degree of disturbance or amount of recovery progress from a state of disturbance to the original pre-disturbance state that has been made (Andersen, 1999). However, there is some uncertainty as to what approach is the most practical and most reliable.

The vast majority of bioindication in the past was done using long-lived plant and vertebrate species despite arthropods being more abundant, easier to sample, and often significantly more sensitive to disturbance due to their inability to move large distances and rapid generation times, especially in the short term (Bragagnolo et al., 2007; Kitching et al., 2000; Rohr et al., 2007). The use of vertebrates (mainly mammals) stems both from

the taxonomic impediment associated with invertebrates as well as a keen public interest in warm-blooded megafauna (Kitching et al., 2000). The majority of managers and research scientists also prefer to use bioindicators that are from within their own field of expertise, or their 'pet organism' (Andersen, 1999) rather than the most applicable bioindicator for the purposes of their study, assessment, or monitoring project.

The clear advantages of utilizing terrestrial arthropods as bioindicators lead to the approach gaining momentum in the early 1990s (Kremen et al., 1993). It was suggested that arthropod bioindicators may be more applicable than their vertebrate counterparts for terrestrial bioindication due to their high density and population sizes throughout the world, occupation of a wide variety of niches, ease and reliability of sampling, rapid sample processing, and little to no societal backlash regarding sampling methods. Kremen et al. (1993) also discussed the advantages of reference collections of arthropods. The maintenance of reference collections allows for repeated verification by taxonomic experts as well as direct comparison of species during a monitoring program. When using large vertebrates this approach is simply not feasible, and does not allow for a fine scale resolution of ecosystem change monitoring.

1.3 Selection of Bioindicators

While much attention has shifted to arthropods as valuable candidates for bioindication, the central questions to bioindication studies are still often overlooked, i.e., what is being indicated (Andersen, 1999) and what are the goals of bioindication (McGeoch, 1998)? Biodiversity, responses to habitat alteration or destruction, amongst many others are

potential answers to these questions and must be addressed prior to commencing a study, assessment, or monitoring program. Only with clearly defined objectives can one select the most suitable indicator taxon. Alternatively, these same questions may be used to determine the most appropriate type of bioindication approach.

According to McGeoch (1998), several conclusions may be drawn regarding the selection criteria of bioindicators based on these studies. First, the bioindicator must meet financial and logistic constraints. A bioindicator candidate may accurately represent the desired characteristic for which it is an indicator, but may not be a feasible choice due to high costs or impractical requirements of its sampling, identification or interpretation may not be a feasible choice. Second, the bioindicator must satisfy the basic requirements of activity-abundance, sensitivity to disturbance, specificity of habitat requirements, a well known taxonomy and life history, and be easily trapped. Finally, the limitations of the bioindicator must be known and acknowledged. For example, there is enormous natural variation in populations from year to year for many of the taxa that are investigated (Andersen, 1999). The remedy for any shortcoming in addressing these criteria may very well lie in an alternative approach to taxonomic bioindication, or a better balance, satisfying a greater degree of criteria may be achieved.

Rohr et al. (2007) suggest the following three steps for developing an effective monitoring program: 1) characterize the community (inventory); 2) identify valid surrogates for biodiversity and; 3) establish efficient methods to monitor the surrogates and ecologically sensitive or important taxa. The authors explain that a characterization

(inventory) of the community is necessary to establish a baseline to compare to following any perturbations to the community. The inventory allows for the selection of surrogates of biodiversity through the observation of trends and correlations amongst species or groups of species. The final step for the proposed method of developing a monitoring program is the development of efficient methods to monitor the surrogates. These methods should be targeted at the surrogates to avoid processing non-surrogates at an additional cost. The recognizable taxonomic unit approach offers a solution to the unnecessary processing of non-surrogates through the inclusion of all possible surrogates in RTU groups while the functional group approach offers a solution through the identification of functional groups or a specific species of interest.

The arthropods constitute an extremely diverse phylum and the selection of a bioindicator from within this taxon requires careful research and planning. Andersen (1999) addresses the difficulty of selecting appropriate indicator candidates from within such a vast array of possibilities. He suggests selection criteria that reflect the taxon's: 1) distribution, activity-abundance, and richness; 2) functional role in ecosystems; 3) sensitivity to environmental perturbations; 4) ability to be sampled and identified; and 5) ability to reliably interpret the response of the indicator. If ecological and life history information is not available for the bioindicator organism then interpretation may either be meaningless or impossible to achieve.

1.4 Ground Beetle (Coleoptera: Carabidae) Bioindicators

For most non-biologists ground beetles represent the 'standard' beetle; they are ubiquitous, easily trapped and identified (Stork, 1990). Because of these features and their ecological diversity, ground beetles are widely used as bioindicators (Rainio & Niemelä, 2003). The use of a single taxon identified to species constitutes a traditional approach to using arthropods as bioindicators

Lövei & Sunderland (1996) report that the family Carabidae contains more than 40 000 described species worldwide, and explain that they are well-proportioned cursorial beetles with prominent mandibles and palps, long slender legs, striate elytra, and sets of punctures with tactile setae. Lövei & Sunderland (1996) attribute the popularity of carabid beetles as study subjects to their activity-abundance, species richness, and attractive colouration. The great species richness of the taxa allows for a greater precision in bioindication studies.

The habitat and microhabitat distribution of carabid beetles may be influenced by several factors including: 1) temperature and humidity extremes; 2) food conditions; 3) presence and distribution of competitors; and 4) life history and season (Lövei & Sunderland, 1996). These factors result in sensitivity to habitat change, and when combined with the great activity-abundance, ease of capture, and understanding of taxonomy for the family, make carabid beetles good candidates for bioindication studies (Lindroth, 1961-1969; Larochelle & Larivière, 2003).

Rainio & Niemelä (2003) examine carabid beetles in relation to the bioindicator criteria set forth by McGeoch (1998). They note that taxonomic and ecological knowledge of carabids is quite good and their distribution is expansive, with a presence in all major habitats with the exception of the driest regions of deserts (Lövei & Sunderland, 1996). The next criterion that Rainio & Niemelä (2003) examined was that of habitat specialization. They note that carabids may be divided into generalist species that occupy a wide range of habitats, and specialists occurring in one or a few habitats. These two types of carabid beetles are often found in community assemblages in the same habitats, making it possible to utilize an entire assemblage as a bioindicator, thus adding to the hypothesis that recognizable taxonomic units may be a viable surrogate for species.

Rainio & Niemelä (2003) also discuss the requirement of a bioindicator to be able to provide an early warning of habitat change through changes in their diversity and/or populations, and cite numerous examples of studies successfully using carabid beetles for just such a purpose. They also discuss the cost and ease of trapping carabid beetles, noting that pitfall trapping is exceptionally cost effective. Ideally, bioindicator candidates should express results that are independent of sample size, meaning a small or large sample size should yield similar results. However, Rainio & Niemelä (2003) note that this is not the case with carabid beetles due to their patchy distribution. In order to minimize this effect they suggest using a high number of replicates.

The final points Rainio & Niemelä (2003) address in their evaluation of the potential use of carabids as bioindicators pertain to how their responses reflect those in other species,

and how natural variation can be differentiated from anthropogenically forced variation. How well carabids reflect other species responses requires further study (Rainio & Niemelä 2003), but they have been shown to correlate well with responses in some other arthropod taxa such as spiders and ants. In terms of differentiating anthropogenically induced variation from natural variation, it is important to realize that annual variation in carabid populations is common and largely dependent upon weather and other abiotic as well as biotic factors. Rainio & Niemelä (2003) suggest that control sites will often allow for the differentiation between anthropogenic and natural variation, but depending on study site locations, different weather and climatic conditions may require consideration.

Rainio & Niemelä (2003) concluded that carabids meet the majority of the criteria for a desirable bioindicator. The issues that arise from their use are minor compared to other bioindicator candidates. The application of carabid beetles as bioindicators in monitoring, assessment, or research projects should depend largely upon the objectives of the project and an assessment of the strengths and weaknesses of carabids as bioindicators in achieving those objectives. In some instances more detailed information may be required in which case the alternative approach of functional diversity could be employed, while in others less detailed information would suffice in which case recognizable taxonomic units may be used.

Carabids have been used successfully as bioindicators for a wide range of studies and this provides the rationale for their use in this comparative study of bioindication approaches. In addition to being a well established indicator taxon, carabids also have enough

morphological variety within a typical ecosystem that they may be sorted into RTUs by those with little to no identification training. Dauber et al. (2005) utilized carabids in combination with other arthropod taxa to highlight how surrounding habitat quality and land use affects invertebrate populations, while Croci et al. (2008) successfully used carabid beetles as bioindicators to evaluate the importance of urban woodlots as candidates for conservation.

Recently, Work et al. (2008) examined the possibility of utilizing carabids as bioindicators at large geographic scales. They addressed the issue that most governmental entities and non-governmental organizations would need to use bioindicators over very large geographic ranges. The study found that while some species of carabids could be found in all study sites across Canada, they were not particularly sensitive for bioindication of habitat alteration and responded differently to the habitat alteration based on their geographic location as well as the silvicultural treatment they were subjected to. Although large-scale applications of a single bioindicator species do not appear to be feasible, the study reaffirmed the sensitivity of carabids for local scale-studies.

Much of the evaluation of carabids as bioindicators has focussed on forest ecosystems, especially in reference to responses to silvicultural treatments (Butterfield et al., 1995; Abildsnes & Tømmerås, 2000; Heliölä et al., 2001; Koivula, 2002; Allegro & Sciaky 2003; Lemieux & Lindgren, 2004; Moore et al., 2004; Klimaszewski et al., 2005; Work et al., 2008). This is largely because most studies have been conducted in temperate forests of countries dependent upon forestry as a major industry, where there is a strong

incentive to identify silvicultural treatments that minimize biodiversity impacts on the landscape. Bioindication with carabid beetles could also be of great use in tropical areas where much of the world's biodiversity can be found, however. An enormous number of carabids remain unidentified in these regions and even where species are identifiable there is a severe lack of taxonomic expertise.

The objectives of this thesis are threefold: 1) to establish a baseline bioindication study using a traditional taxonomic approach with carabid beetles as the surrogate organism; 2) to explore alternative approaches to bioindication, through recognizable taxonomic units and functional traits, that may overcome impediments associated with the taxonomic approach; and 3) to evaluate and compare the three bioindication methods and make recommendations based upon these findings. In chapter two, I will examine the first objective by using a traditional taxonomic approach by evaluating carabid beetle species richness, activity-abundance and evenness. In chapters three and four, I will explore recognizable taxonomic units and functional traits, respectively, as potential alternative approaches to using arthropods as bioindicators. Within these chapters I will fulfill objective two. Finally in chapter five, I will evaluate and compare the three methods and offer recommendations and proposed future research, satisfying the third and final objective.

Literature Cited

Abildsnes, J. & B.Å. Tømmerås. 2000. Impacts of experimental habitat fragmentation on ground beetles (Coleoptera, Carabidae) in a boreal spruce forest. Annales Zoologici Fennici 37:201-212.

Allegro, G. & R. Sciaky. 2003. Assessing the potential role of ground beetles (Coleoptera, Carabidae) as biondicators in poplar stands, with a newly proposed ecological index (FAI). Forest Ecology and Management 175:275-284.

Andersen, A.N. 1999. My bioindicator or yours? Making the selection. Journal of Insect Conservation 3:61-64.

Balmford, A., M.J.B. Green, & M.G. Murray. 1996. Using Higher-Taxon Richness as a Surrogate for Species Richness: I. Regional Tests. Proceedings: Biological Sciences The Royal Society, 263: 1267-1274.

Bragagnolo, C., A.A. Nogueira, R. Pinto-da-Rocha, & R. Pandini. 2007. Harvestment in an Atlantic forest fragmented landscape: evaluating assemblage response to habitat quality and quantity. Biological Conservation 30:389-400.

Butterfield, J., M.L. Luff, M. Baines, & M.D. Eyre. 1995. Carabid beetle communities as indicators of conservation potential in upland forests. Forest Ecology and Management 79:63-77.

Croci, S., A. Butet, A. Georges, R. Aguejdad, & P. Clergeau. 2008. Small urban woodlands as biodiversity conservation hot-spot: a multi-taxon approach. Landscape Ecology 23:171-1186.

Dauber, J., T. Purtauf, A. Allspach, J. Frisch, K. Voigtländer, & V. Wolters. 2005. Local vs. landscape controls on diversity: a test using surface-dwelling soil macroinvertebrates of differing mobility. Global Ecology and Biogeography, 14:213-221.

Derraik, J.G.B., G.P. Closs, K.J.M. Dickinson, P. Sirvid, B.I.P. Barratt, & B.H. Patrick. 2002. Arthropod morphospecies versus taxonomic species: a case study with Araneae, Coleoptera, and Lepidoptera. Conservation Biology 16:1015-1023.

Heliölä, J., M. Koivula, & J. Niemelä. 2001. Distribution of Carabid Beetles (Coleoptera, Carabidae) across a Boreal Forest-Clearcut Ecotone. Conservation Biology 15:370-377.

Jokimäki, J., E. Huhta, J. Itämies, & P. Rahko. 1998. Distribution of arthropods in relation to forest patch size, edge, and stand characteristics. Canadian Journal of Forest Research. 28:1068-1072.

Kitching, R.L., A.G. Orr, L. Thalib, H. Mitchell, M.S. Hopkins, & A.W. Graham. 2000. Moth Assemblages as Indicators of Environmental Quality in Remnants of Upland Australian Rainforest. Journal of Applied Ecology 37:284-297.

Klimaszewski, J., D.W. Langor, T.T. Work, G. Pelletier, H.E.J. Hammond, & C. Germain. 2005. The effects of patch harvesting and site preparation on ground beetles (Coleoptera, Carabidae) in yellow birch dominated forests of southeastern Quebec. Canadian Journal of Forest Research 35:2616-2628.

Koivula, M. 2002. Boreal carabid-beetle (Coleoptera, Carabidae) assemblages in thinned uneven-aged and clear-cut spruce stands. Annales Zoologici Fennici 39:131-149.

Kremen, C., R.K. Colwell, T.L. Erwin, D.D. Murphy, R.F. Noss, & M.A. Sanjayan. 1993. Terrestrial Arthropod Assemblages: Their Use in Conservation Planning. Conservation Biology 7:796-808.

Larochelle, A. & M.-C. Larivière. 2003. A natural history of the ground-beetles (Coleoptera: Carabidae) of America north of Mexico. Sofia, Bulgaria : Pensoft.

Lemieux, J. & B.S. Lindgren. 2004. Ground beetle responses to patch retention harvesting in high elevation forests of British Columbia. Ecography 27:557-566.

Lindroth,C.H. 1961-1969. The ground-beetles (Carabidae, excl. Cicindelinae) of Canada and Alaska, Opuscula Entomologica Supplements, 20 (1961), 24 (1963), 29 (1966), 33 (1968), 34 (1969), & 35 (1969). Entomologiska Sällskapet, Lund, Sweden.

Lövei, G.L. & K.D. Sunderland. 1996. Ecology and Behavior of Ground Beetles (Coleoptera: Carabidae). Annual Review of Entomology 41:231-256.

McGeoch, M.A. 1998. The selection, testing and application of terrestrial insects as bioindicators. Biology Reviews 73:181-201.

Moore, J., Ouimet, R., Houle, D., & Camiré, C. 2004. Effects of two silvicultural practices on ground beetles (Coleoptera, Carabidae) in a northern hardwood forest, Quebec, Canadian Journal of Forest Research 34:959-968.

Oliver, I. & A.J. Beattie. 1993. A Possible Method for the Rapid Assessment of Biodiversity. Conservation Biology 7:562-568.

Oliver, I. & A.J. Beattie. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. Ecological Applications 6:594-607.

Rainio, J. & J. Niemelä. 2003. Ground beetles (Coleoptera: Carabidae) as bioindicators. Biodiversity and Conservation 12:487-506.

Ribera, I., S. Dolédec, I.S. Downie, & G.N. Foster. 2001. Effect of Land Disturbance and Stress on Species Traits of Ground Beetle Assemblages. Ecology, 82:1112-1129.

Rohr, J.R., Carolyn G. Mahan, & K.C. Kim. 2007. Developing a monitoring program for invertebrates: guidelines and a case study. Conservation Biology 21:422-433.

Stork, N.E. 1990. The Role of Ground Beetles in Ecological and Environmental Studies. Andover, Hants, England. Intercept Limited.

Work, T.T., M. Koivula, J. Klimaszewski, D. Langor, J. Spence, J. Sweeny, & C. Hebert. 2008. Evaluation of carabid beetles as indicators of forest change in Canada. Canadian Entomologist 140:393-414.

Chapter 2- Ground beetles (Coleoptera: Carabidae) as Bioindicators to Evaluate Harvesting Effects in an Alternatively Harvested Coastal Forest

2.1 Abstract

Carabid beetles are the most commonly used arthropod taxon as bioindicators. The reasons for this are their ubiquitous nature and relative sensitivity to habitat alteration, as well as their relatively well known taxonomy. I examine their utility as bioindicators in an alternatively harvested study forest with varying levels of retained canopy structure located in a mature stand composed primarily of Douglas-fir and western hemlock in British Columbia's Sunshine Coast. Pitfall trapping was conducted in each of four canopy treatment types (closed (control), gap, semi-open, and open) for three weeks in the summers of 2007 and 2008. Only 15 species of carabids were trapped. Carabid beetle activity-abundance and species richness did not differ between semi-open (60% canopy closure) and open canopy closure treatments (40% canopy closure), but were significantly lower there than in closed (80% canopy closure) and gap canopy treatments (75% canopy closure), which also did not differ significantly. Evenness results were not significant, and were not evaluated further. These findings indicate that the carabid assemblage in the study area can be used as bioindicators of coarse changes to forest structure.

2.2 Introduction

Carabid beetles (Coleoptera: Carabidae) have received much attention as potential bioindicators (Butterfield et al., 1995; Allegro & Sciaky 2003; Rainio & Niemelä, 2003;

Work et al., 2008). In particular, they have been widely used to reflect forest habitat changes in the wake of disturbance (Abildsnes & Tømmerås, 2000; Heliölä et al. , 2001; Koivula, 2002; Lemieux & Lindgren, 2004; Moore et al. , 2004; Finch, 2005; Klimaszewski et al., 2005).

McGeoch (1998) describes a process for testing the suitability of taxa as potential bioindicators and defines two broad requirements that all candidates should possess, of which carabid beetles meet. The first requirement is that of economic and logistical feasibility which includes financial cost, time efficiency and personnel requirements. The second requirement is that of biological efficacy which includes aspects such as taxonomic, distributional, reliability, representational, and sensitivity criteria. Rainio & Niemelä (2003) subsequently used McGeoch's method to evaluate the suitability of carabid beetles as bioindicators.

Rainio & Niemelä (2003) explain that carabid beetles meet many of the criteria for bioindicator candidates. Their taxonomy and ecology are well known, they are widely distributed and have generalist and specialist species in many geographic areas, respond rapidly to habitat alteration, and sampling is easy and cost efficient. However, they also note that carabids are often criticized as unsuitable bioindicators, because there are too many generalist species, environmental requirements vary by species with some more sensitive to disturbance than others, and common trapping techniques such as pitfall trapping depend largely upon the activity of the beetles. In addition, due to the patchy distribution of ground beetles, surveys are not independent of sample size, but may be

influenced by the number and location of traps used. In addition, the degree to which carabid beetle diversity reflects the diversity of other species is poorly studied, and natural variation must be taken into consideration as population sizes vary from year to year and with changes in environmental conditions. While these criticisms have merit, they are generally less substantive than those for other arthropod bioindicators or classic plant or vertebrate bioindicators.

Work et al. (2008) found that at a larger scale, such as one that would be used by governments, the use of carabid beetles as bioindicators requires region specific calibration. While many species of carabids are distributed across provinces and even a large country like Canada, they shift in dominance depending upon the region and ecosystem. The authors note that carabid responses to disturbance are markedly similar throughout Canada, however, albeit with different species at different levels of dominance.

The objective of the study is to examine the response and resolution of a traditional taxonomic approach using carabid beetles as potential bioindicators in a coastal B.C. study forest containing harvest treatments with varying degrees of canopy structure retention. Thus, this study evaluates the utility of carabid beetle activity-abundance, species richness, and evenness as bioindicators of ecosystem disturbance.

2.3 Materials and Methods

2.3.1 Study Area and Field Data Collection

Carabid beetles were collected during field sampling for the Coast Region Experimental Arthropod Project (CREAP), Research, Innovation and Knowledge Management Branch, British Columbia (BC) Ministry of Forests and Range (Todd et al., 2008). The study area is located in the Robert's Creek Study Forest of B.C.'s Sunshine Coast (Figure 2.1) in the Coastal Western Hemlock zone, Dry Maritime biogeoclimatic subzone

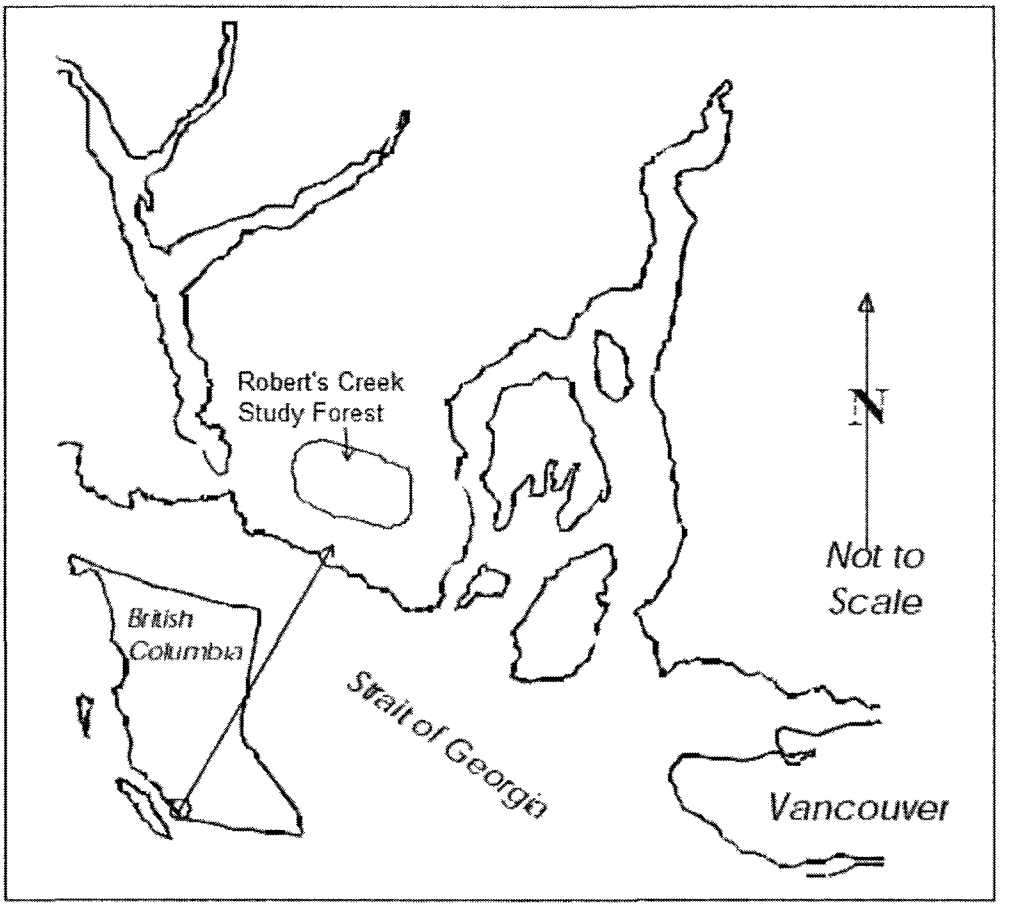

Figure 2.1 Location of the Robert's Creek Study Forest for alternative harvesting practices research.

(CWHdm) (Meidinger & Pojar, 1991). The forest itself is composed primarily of Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) with sparse amounts of western red cedar (*Thuja plicata*). The study forest averaged 120 years old at the time of harvesting treatment, with fire-origin mature stands (D'Anjou, 2002). The study forest is composed of alternatively harvested treatments of varying canopy structure retention. There are four harvest treatment types (unharvested controls, dispersed retention, extended rotation, and clearcut), with two harvest treatment units in each type, for a total of eight harvest treatments (refer to Figure 2.2 and Table 2.1 for details). Harvest treatments were applied in three phases (Demonstration, Phase 1, and Phase 2) from 1993 to 1999 using a cable-logging system to ensure minimal ground disturbance, with some thinning and windthrow salvage treatments applied in the dispersed retention units prior to 2000. Pitfall samples were collected from the eight harvest treatment units in 2007 and 2008.

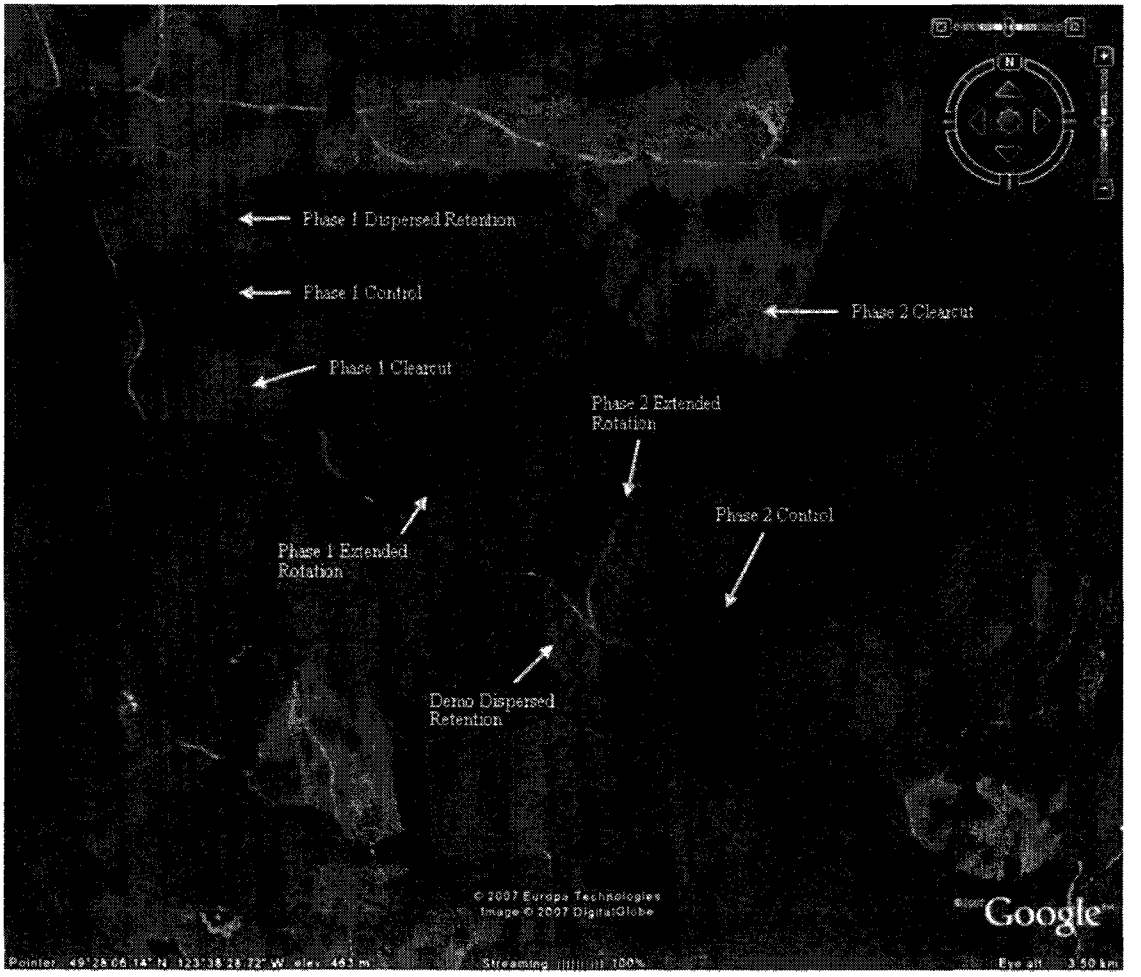

Figure 2.2 Canopy closure treatments of the Robert's Creek Study Forest

Table 2.1. Target retention, harvest prescriptions, and descriptions of treatment blocks at the Robert's Creek Study Forest, BC.

Canopy Treatment Type	Canopy Closure Percent	Harvest Prescription	General Site Description
Open	40% average canopy closure for trap arrays.	Clearcut with scattered residual single tree retention (1 stem per hectare (sph); Douglas-fir and western red cedar).	Large amounts of coarse woody harvest debris (CWD). Understorey primarily young Douglas-fir, western hemlock and western red cedar (<10m tall) with some shrub (salal, *Vaccinium* sp. and *Rubus* sp). Some herbaceous ground cover.
Semi-open	60% average canopy closure for trap arrays.	Single tree dispersed retention. Left dominant Douglas-fir and western red cedar at 57 and 95 sph; subsequent thinning to 25 sph.	Large amounts of harvest CWD. Variable understory development, including regenerating trees and shrub species. Large, scattered veteran trees (>50m). Rich herbaceous ground cover.
Gap	75% average canopy closure for trap arrays.	Extended rotation. Cut eleven 4-5m and eighteen 6-8m parallel corridors, removing approximately 11% and 18% stand volume respectively.	Moderate amounts of pre-harvest CWD and some harvest debris; very little understory development in narrow corridors, more in wider corridors; understorey is primarily bryophytes and salal, with some tree regeneration (<5m tall) in wider corridors
Closed	80% average canopy closure for trap arrays.	None. Unharvested control.	Douglas-fir leading followed by western hemlock and western red cedar. Very closed-canopy, open understory mature (pre-gap formation) CWHdm forest. Moderate CWD; bryophyte-dominated ground cover.

Source of harvest information: D'Anjou (2002)

Within each harvest treatment unit three sample sites, each consisting of an array of four pitfall traps, were established. The only exception to this was in 2007, when only two sample sites were established in the Phase 2 Clearcut (see Figure 2.2), with a third site added in 2008.

I utilized shallow pitfall traps (Pearce et al., 2005) to avoid vertebrate by-catch without affecting trap efficacy (Meggs, 2007). Pitfall traps are a means to measure activity-abundance rather than abundance as the catch numbers rely on the active movement of organisms in order to be trapped. Pitfall traps consisted of a 10cm diameter PVC pipe coupling placed into a hole in the ground into which a 250ml plastic deli container (11.5cm diameter opening and 4.2cm tall) was inserted, flush with the surrounding ground, to act as the trap (Meggs, 2007). The trap was filled halfway (approximately 125ml) with a 50:50 solution of low-toxicity propylene glycol antifreeze and water to act as both a preservative and a killing agent. A protective cover made of a 15cm white plastic plate was installed above each trap at a height of approximately 2cm and secured in place with three bamboo skewers. Each of the three sites was randomly located within the harvest treatment unit. Sites were separated by at least 150m to ensure independence, and located at least 50m from roads, treatment edges, and riparian areas to avoid edge effects. Individual pitfall traps within the four-trap array were spaced 17-25m apart to avoid depletion effects and assume independence (Digweed et al., 1995). In 2007, traps were opened immediately following installation whereas in 2008 they were allowed to sit closed for one week following installation to avoid the 'digging in effect' whereby certain ground arthropods, including some carabids are attracted to disturbance (Digweed et al.,

1995). Traps were left open for a 19-21 day period from mid-June to early-July in both 2007 and 2008. A total of 92 pitfall trap samples were collected from 23 pitfall array sites across the eight treatment units in 2007 (Meggs 2007) and 96 pitfall trap samples were collected from 24 pitfall array sites across the same treatment units in 2008 (Todd et al., 2008). The contents of each trap were rinsed with water and stored in 70% ethanol prior to separating and identifying all carabid beetles. The sample collection explained here represents the same sample collection for all subsequent chapters.

Canopy closure percentages were obtained for each pitfall trap location using a spherical densiometer at approximately 1m above the trap.

2.3.2 Carabid Species Identification

All pitfall trap samples were initially sorted to recognizable taxonomic unit (RTU) (refer to Chapter 3). Each beetle was then identified to species using a set of well established keys (Lindroth, 1961-1969) by observing morphological characteristics under a dissecting microscope. Reference collections were created as species were identified. To ensure accuracy, specimen confirmation was obtained from D. Shpeley at the Strickland Entomological Museum, University of Alberta.

2.3.4 Analysis

Trap catches from each pitfall array site were pooled together to yield a total catch per site and represent the carabid beetle assemblage at the site. Canopy closure percent for each site was calculated as the mean of canopy closure percentages for the four traps in

the pitfall array. This resulted in 23 sample units for 2007 data and 24 sample units for 2008 data. I used an analysis of covariance to determine if there was a significant difference in the carabid beetle assemblage between harvest treatment types. Mean carabid beetle species diversity (richness), activity-abundance, and Shannon's equitability (evenness) of each site served as the response variables by canopy treatment type (n=4; n=6 sites per canopy treatment type), with canopy closure percent as a covariate. This analysis was conducted for each year of data independently. A year to year comparison was not conducted due to a difference in sampling methodology (digging in versus not digging in). If a significant treatment effect was found in the ANCOVA model, protected t-tests were employed to examine all pair-wise comparisons. A log likelihood ratio test was performed to compare the standard ANCOVA model with a mixed effects ANCOVA that included canopy treatment type replicate and pitfall array plot as random effects. The models were not significantly different ($P = 0.90$), suggesting that pseudoreplication was not an issue and the standard ANCOVA model was used. Activity-abundance data were log transformed to meet the assumptions of an ANCOVA model, namely equal variance and normal distribution of errors. I confirmed that these assumptions were met by visually inspecting the residual plots.

I generated species accumulation curves to determine if sampling intensity was sufficient for both years. These curves allow for comparison of the cumulative species number as well as sampling effort between years and different canopy closure types (Larsen et al., 2003). Species accumulation curves were generated using EstimateS software (Colwell, 1997).

Differences in species assemblages by canopy closure type and canopy closure percent were explored in order to determine how the different harvest treatments impacted the carabid beetle assemblage. This was done using Non-metric Multidimensional Scaling (NMS) using PC-ORD 5.31 software (McCune & Mefford, 2006). NMS works by ordinating samples so that ordination space increases as samples become dissimilar, thus the more similar samples are the closer they appear in ordination space (Sprules, 1980). Data were log transformed and standardized to their own maxima, while species accounting for less than 1% of the trap catch were excluded (Lemieux & Lindgren, 2004). These transformations were necessary to reduce the influences of extremely abundant and extremely rare species on the results (McCune & Grace, 2002). Monte Carlo tests (500 runs) were performed to determine if the NMS axes represented non-random solutions.

2.4 Results

A total of 1425 carabid beetles representing 13 species were caught in 92 traps in 2007 (Table 2.2), and 1108 carabid beetles representing 13 species were caught in 96 traps in 2008 (Table 2.3). One species was unique to 2007 and one to 2008 bringing the species total for the study to 15. First and second order jackknife estimates from the species accumulation curves suggest that 16 or 17 species were likely active in the sampling area during the sampling period. Three species, *Pterostichus algidus* (LeConte), *P. herculaneus* (Mannerheim), and *Scaphinotus angusticollis* (Mannerheim), accounted for the majority of the carabid beetles captured at 89.5% and 82.6% of the catches in 2007

and 2008, respectively. The majority of beetles were found to be associated with the closed canopy control and gap treatments. Canopy closure percent and canopy treatment type data by site is available in appendices A and C for the 2007 and 2008 sampling years.

For 2007 samples, the analysis of covariance model showed a significant difference in mean carabid beetle species richness based on treatment type ($F_{3,18} = 12.610$, $P < 0.001$) while percent canopy closure was not statistically significant ($F_{1,18} = 3.9487$, $P = 0.062$). Similarly, trap array data from 2008 revealed significant differences in the mean diversity of carbid species ($F_{3,19} = 7.9143$, $P = 0.001$) whereas percent canopy closure did not have a significant effect ($F_{1,19} = 1.12$, $P = 0.300$).

Further examination of treatment type showed that in 2007 closed canopy treatments were not significantly different from gap treatments ($t_{1,10} = 0.724$, $P = 0.48$) but were significantly different from both semi-open ($t_{1,10} = -2.528$, $P = 0.021$) and open ($t_{1,10} = -2.192$ $P = 0.042$) canopy treatments (Figure 2.3). Gap canopy treatments were significantly different from semi-open ($t_{1,10} = -3.321$, $P = 0.004$) and open ($t_{1,10} = -2.883$, $P = 0.010$) canopy treatments while semi-open and open canopy treatments were not significantly different ($t_{1,10} = 0.140$, $P = 0.890$). In 2008, closed canopy treatments were not significantly different from gap ($t_{1,10} = 0.116$, $P = 0.910$) or semi-open canopy treatments ($t_{1,10} = -1.99$, $P = 0.061$), but were significantly different from open ($t_{1,10} = -2.247$, $P = 0.037$) canopy treatments. Gap canopy treatments were significantly different from semi-open ($t_{1,10} = -2.176$, $P = 0.042$) and open ($t_{1,10} = -2.464$, $P = 0.023$) canopy

treatments while semi-open and open canopy treatments were not significantly different ($t_{1,10} = 0.77$, $P = 0.450$).

Table 2.2. Number of individuals of each carabid beetle species caught by canopy closure type in 2007 from the Robert's Creek Study Forest.

Species	Closed	Gap	Semi-Open	Open*	Total	% of Catch
***Cychrus tuberculatus* Harris**	15	6	6	19.2	46.2	3.22
Leistus ferruginosus Mannerheim	1	2	0	0.0	3.0	0.21
Loricera decempunctata Eschsholtz	0	0	0	0.0	0.0	0.00
Notiophilus sylvaticus Eschsholtz	5	4	0	0.0	9.0	0.63
***Omus dejeani* Reiche**	15	23	7	4.8	49.8	3.47
Promecognathus crassus LeConte	1	0	0	0.0	1.0	0.07
***Pterostichus algidus* LeConte**	165	197	5	2.4	369.4	25.72
Pterostichus crenicollis LeConte	0	0	7	2.4	9.4	0.65
***Pterostichus herculaneus* Mannerheim**	303	369	78	27.6	777.6	54.15
***Pterostichus lama* Menetries**	5	0	6	9.6	20.6	1.43
Pterostichus neobrunneus Lindroth	12	2	0	0.0	14.0	0.97
Scaphinotus angulatus Harris	0	2	0	0.0	2.0	0.14
***Scaphinotus angusticollis* Mannerheim**	103	30	0	0.0	133.0	9.26
Syntomus americanus Dejean	0	0	0	0.0	0.0	0.00
Synuchus impunctatus Say	0	0	1	0.0	1.0	0.07

*Presented catch numbers were standardized in order to correspond with other treatments where 6 sites were present (only 5 sites in 2007 open). Bolded entries are those used in the Non-metric Multidimensional Scaling.

Table 2.3. Number of individuals of each carabid beetle species caught by canopy closure type in 2008 from the Robert's Creek Study Forest.

Species	Closed	Gap	Semi-Open	Open	Total	% of Catch
***Cychrus tuberculatus* Harris**	11	7	5	11	34	3.07
Leistus ferruginosus Mannerheim	0	0	0	0	0	0.00
Loricera decempunctata Eschsholtz	0	2	0	0	2	0.18
Notiophilus sylvaticus Eschsholtz	4	2	0	0	6	0.54
***Omus dejeani* Reiche**	16	42	45	18	121	10.92
Promecognathus crassus LeConte	0	1	1	0	2	0.18
***Pterostichus algidus* LeConte**	100	214	2	2	318	28.70
Pterostichus crenicollis LeConte	1	0	2	0	3	0.27
***Pterostichus herculaneus* Mannerheim**	163	146	65	36	410	37.00
***Pterostichus lama* Menetries**	1	1	7	9	18	1.62
Pterostichus neobrunneus Lindroth	0	3	0	1	4	0.36
Scaphinotus angulatus Harris	0	0	0	0	0	0.00
***Scaphinotus angusticollis* Mannerheim**	135	51	1	0	187	16.88
Syntomus americanus Dejean	0	2	0	0	2	0.18
Synuchus impunctatus Say	0	0	1	0	1	0.09

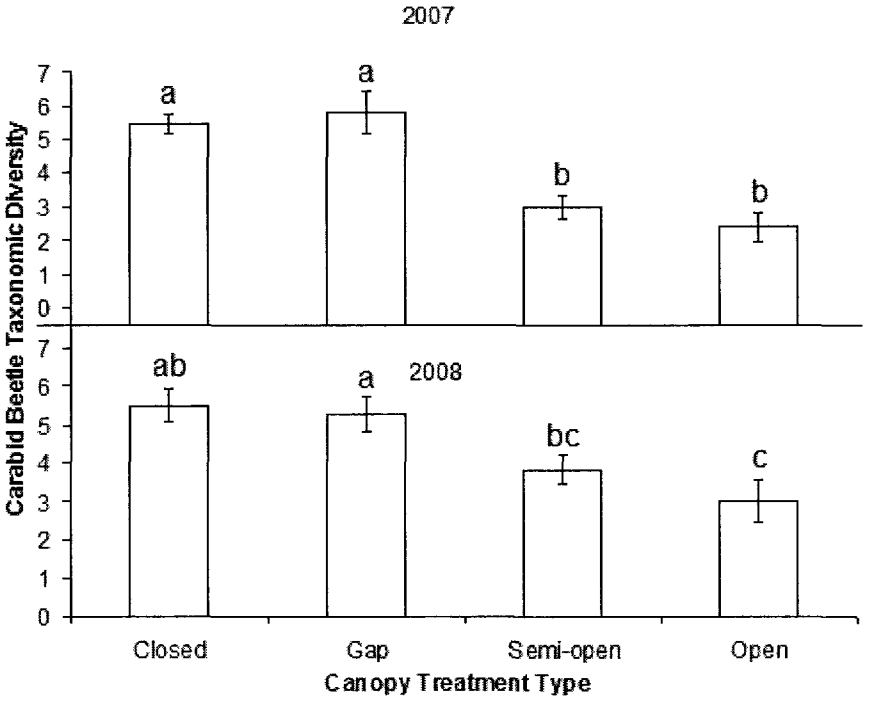

Figure 2.3 Comparison of mean number of species per trap array by canopy treatment type from 2007and 2008 trapping. For significant ANCOVA ($\alpha = 0.05$), bars denoted with the same letter are not significantly different.

Analysis of covariance of carabid beetle activity-abundance, which is independent of taxonomic species groupings, differed significantly by canopy treatment type both in 2007 ($F_{3,18} = 45.7968$, $P < 0.001$) and 2008 ($F_{3,19} = 15.0448$, $P < 0.001$), but canopy closure did not have a significant effect in either 2007 ($F_{1,18} = 4.2795$, $P = 0.052$) or 2008 ($F_{1,19} = 2.0313$, $P = 0.170$).

In 2007, closed canopy treatments were not significantly different from gap canopy treatments ($t_{1,10} = 0.255$, $P = 0.8$), but were significantly different from semi-open ($t_{1,10} = -5.988$, $P < 0.001$) and open ($t_{1,10} = -5.911$, $P < 0.001$) canopy treatments (Figure 2.4).

Gap treatments were also significantly different from semi-open ($t_{1,10}$ = -5.988, P < 0.001) and open ($t_{1,10}$ = -6.436, P < 0.001) canopy treatments while semi-open and open canopy treatments were not significantly different from each other ($t_{1,10}$ = 1.673, P = 0.112). In 2008, closed canopy treatments did not significantly differ from gap canopy treatments ($t_{1,10}$ = 0.423, P = 0.677) but did significantly differ from semi-open ($t_{1,10}$ = -2.713, P = 0.014) and open ($t_{1,10}$ = -2.989, P = 0.008) canopy treatments. Gap

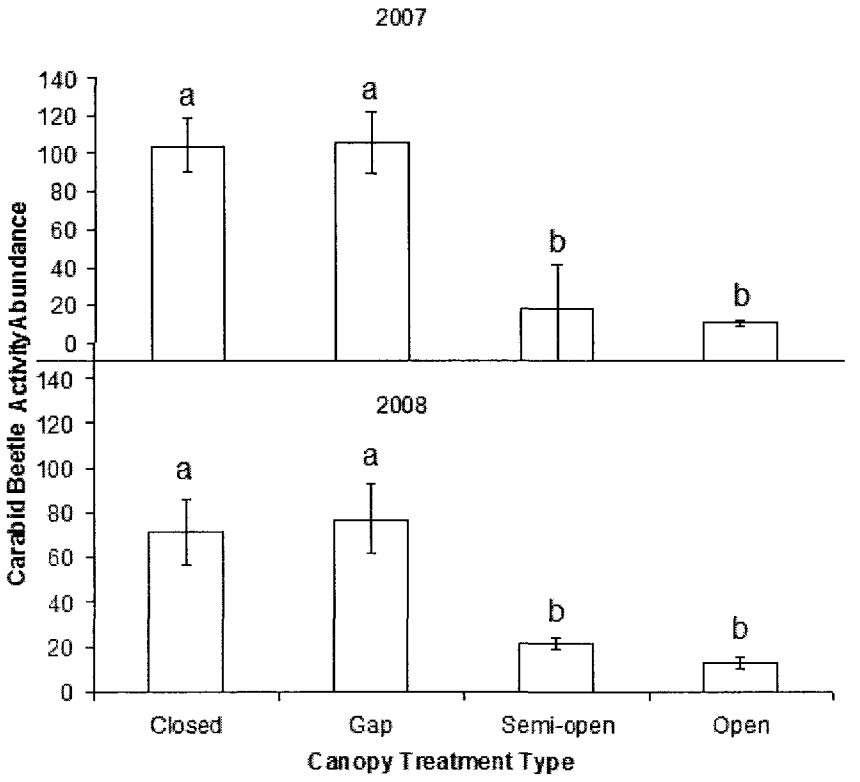

Figure 2.4 Comparison of mean carabid beetle activity-abundance per trap array by canopy closure type from 2007 and 2008 trapping years. For significant ANCOVA (α = 0.05), bars denoted with the same letter are not significantly different.

canopy treatments were significantly different from both semi-open ($t_{1,10}$ = -3.219, P = 0.004) and open ($t_{1,10}$ = -3.490, P = 0.002) canopy treatments, which were not significantly different from each other ($t_{1,10}$ = 0.958, P = 0.350).

The analysis of covariance of species evenness in response to canopy closure treatment and canopy closure percent did not yield a significant result in either year ($F_{3,18}$ = 0.3901, P = 0.761; $F_{3,19}$ = 1.8552, P = 0.172).

Species rarefaction curves for 2007 for all canopy treatment types appear to asymptote or nearly asymptote relative to the x-axis (Figure 2.5). For 2008 none of the canopy treatment type curves approached their asymptotes. A species accumulation curve reaching an asymptote suggests that sampling effort was sufficient and captured the majority of species in the sampling area.

Non-metric multidimensional scaling (NMS) of 2007 data produced a solution where the first two axes (Axis 1 and Axis 2) accounted for 63.9% of the variation (40.0% and 23.9% of the variance, respectively) (Figure 2.6). Monte Carlo tests of whether axes represented non-random solutions were significant at P = 0.024 and P = 0.028, respectively. Similarly, analysis of 2008 data grouped by canopy closure and using canopy closure percentage as a covariate produced a solution within which the first two axes accounted for 51.8% of the variation in the data. The axes represented 27.7% and 24.1% of the variance, respectively, and Monte Carlo tests suggested non-random solutions (P = 0.012 and P = 0.004, respectively). A significant P-value means that the

axis represents where the species are found in relation to the sites and canopy closure percentage. In both years closed and gap canopy treatments cluster together, apart from semi-open and open canopy treatments which also cluster together.

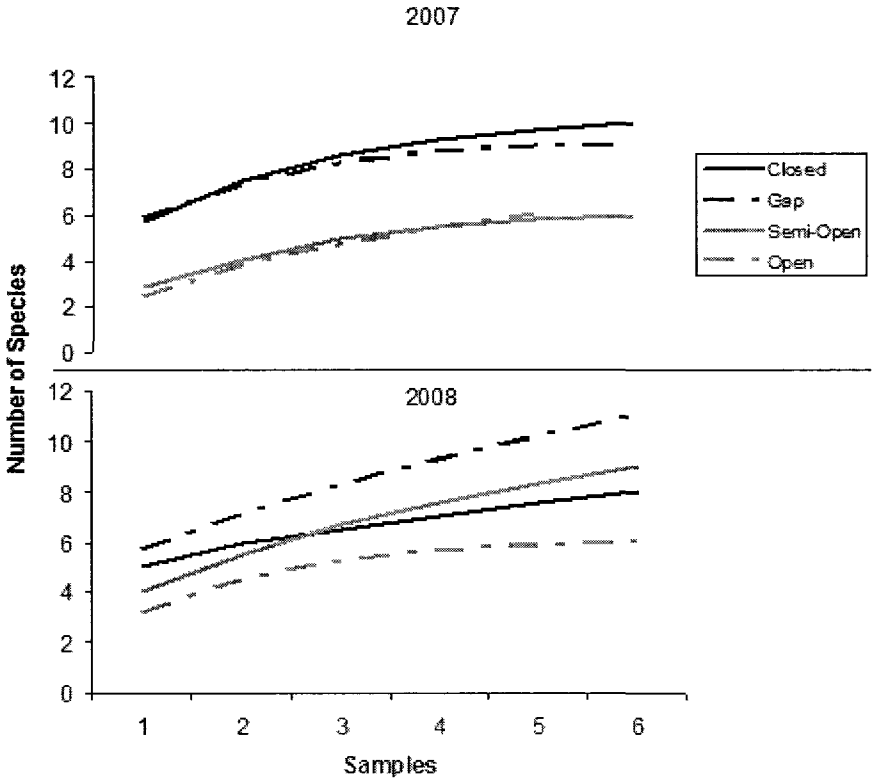

Figure 2.5 Species rarefaction curves based on trap array activity-abundance data by canopy treatment type for 2007 and 2008 trapping period. A curve that reaches asymptote suggests that sampling effort was sufficient while a curve that has not yet reached asymptote suggests that some species were not accounted for.

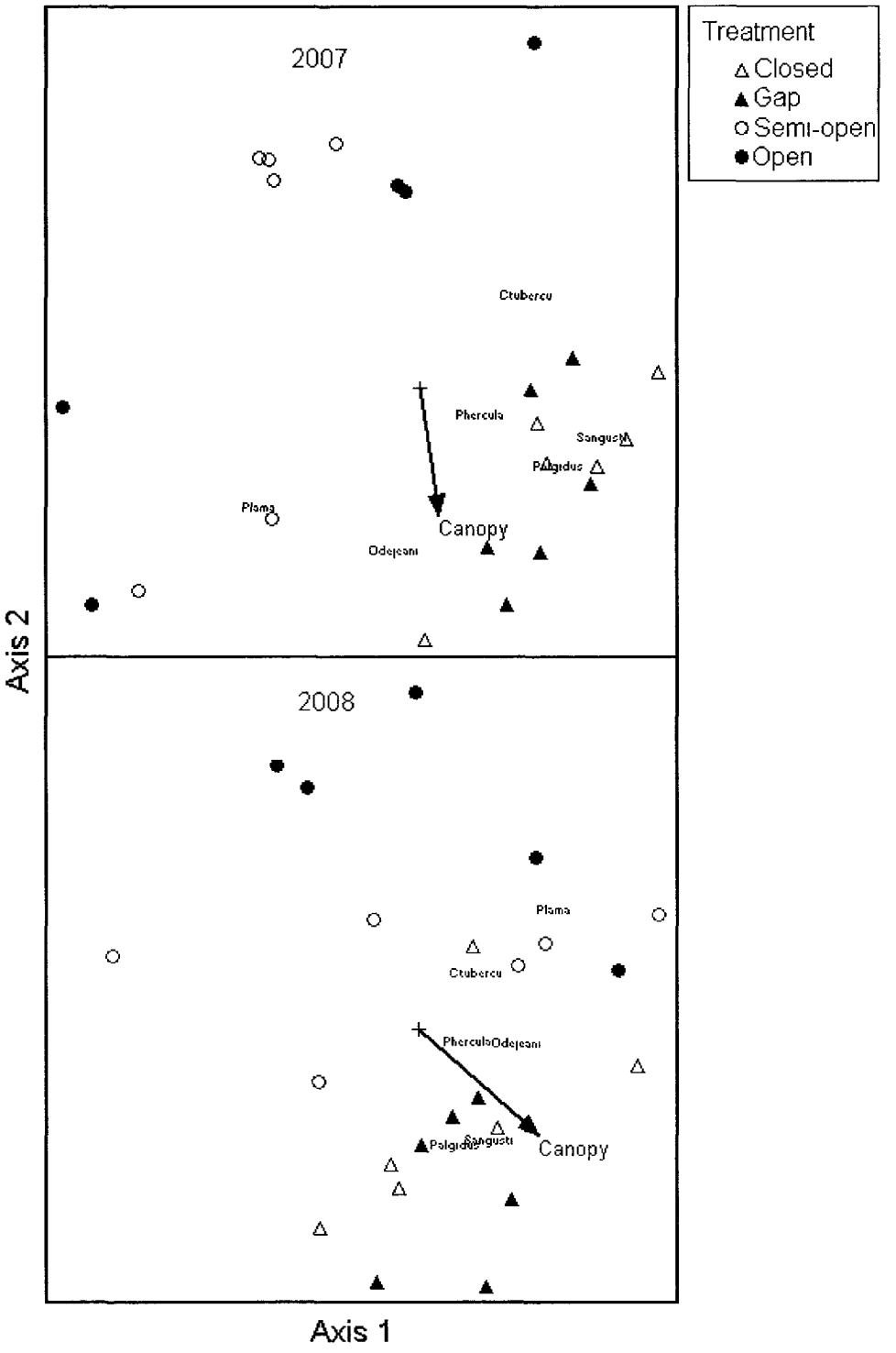

Figure 2.6 NMS of 2007 and 2008 carabid beetle taxonomic data as related to canopy closure type and canopy closure percentage. The closer points are together the more similar they are. Species are grouped with the trap arrays (individual circles and triangles) that they are most strongly associated with.

2.6 Discussion

Pitfall trapping provides an efficient means of sampling epigaeic arthropods' activity-abundance responses to environmental change and is commonly used for biological and ecological monitoring studies (Schowalter et al., 2003; Work et al., 2008). I found a relatively low number of species compared to other studies that used pitfall trapping for carabid beetles. Craig (1995) found 28 species as a result of a full-year trapping effort on Vancouver Island, Lemieux and Lindgren (2004) found 28 species over two complete summers in Northern B.C., and Niemelä et al. (1993) trapped 39 species in a multi-year study near Hinton, Alberta. The low species count in my study is likely attributable to the relatively short trapping period (three weeks), or may simply be representative of assemblages from the Sunshine Coast of B.C. Another study conducted in the south coast of B.C. indicates that it may be the latter, as only 18 species were found in that study (Pearsall, 2006). Species rarefaction curves by treatment type seem to indicate the former rather than the latter, however, at least for 2008 (Figure 2.5). In addition, there is the likelihood of a season effect as our sampling took place only in the late spring and summer months of each year.

My finding that in all instances the gap canopy treatment yielded no significant difference from the control treatment agrees with Klimaszewski et al. (2005), who found that smaller gap treatments resulted in assemblages that more closely resembled those of closed canopy controls. Moore et al. (2004) also found few differences in beetle activity-abundance when comparing strip clearcuts and selective harvesting to closed canopy control treatments. In addition to the maintenance of specific habitat condition by gap

treatments, such as light and moisture, additional similarity may be caused by dispersal capability. The majority of the species in my study were brachypterous, and therefore had limited dispersal capability. A small gap may be possible to traverse; however, large openings resulting from clearcutting may represent an insurmountable barrier to flightless carabids (Heliölä et al. 2001). Activity-abundance of carabid beetles appears to be a sensitive indicator of more intensive disturbances.

Studies using carabid beetles as bioindicators have revealed different trends in activity-abundance and species richness responses. My results are consistent with several studies that have shown decreased abundance and species richness of carabids when habitat is altered from its original state (e.g. Abildsnes and Tømmerås, 2000; Finch, 2005). However, some carabids may respond very positively to the removal of vegetation and introduction of open habitat (Butterfield et al. 1995; Heliölä et al. 2001). Generalists and open habitat specialists disperse to large clearcuts to take advantage of disturbance. Forest specialist species, however, are generally impacted more heavily by the removal of vegetation (Klimaszewski et al., 2005). An increase in activity-abundance and species richness of carabid beetles is not necessarily positive when considering forest health; conversely a decrease in carabid activity-abundance and species richness should not necessarily be considered a negative response.

From a management perspective, the use of activity-abundance and species richness estimates are more practical than assemblage comparisons so long as it is noted that responses of increased or decreased abundance and/or diversity are indications of habitat

alteration. For example, Work et al. (2008) found that despite wide distributions, carabid species respond differently in different ecosystems. The use of diversity and activity-abundance rather than assemblage comparison is further confirmed through studies such as Lemieux & Lindgren (2004) where species were found to respond to habitat disturbance in ways that are contrary to published accounts, such as decreased levels of diversity when others observe increased levels. While a firm understanding of the focal ecosystem is still necessary to interpret results from carabid activity-abundance and species richness estimates, the ecosystem specific life history of the beetles themselves need not be re-examined for every shift in geographic location as it appears necessary to compare assemblages that contain the same species, as concluded by Work et al. (2008).

The study forest is largely composed of mature forest and has little open habitat outside of the prescribed treatment areas. It is therefore unlikely for disturbance specialists to be present and able to colonize the open and semi-open treatments unless source populations were present in the general area prior to harvest. Disturbance specialists are characterized by their ability to rapidly colonize newly disturbed areas (Butterfield et al., 1995). In studies where disturbance specialists are present they can be observed in increased numbers in clearcuts or types of disturbed sites (e.g. Klimaszewski et al., 2005). The carabid beetles in my study are adapted to mature forests and are not well equipped to deal with drastic shifts in habitat characteristics, e.g., the assemblage is characterized by largely brachypterous species, thereby making rapid colonization improbable (Larochelle & Larivière, 2003). Larochelle & Larivière (2003) also describe the species that are represented in my study as being forest species. The results of the non-metric

multidimensional scaling (NMS) models follow this description as the majority of species associate with the control and gap treatments, while the most abundant species are strongly associated with the control and gap treatments (Figure 2.6). The 2007 data more distinctly separate the control and gap treatment types from the semi-open and open treatment types (Figure 2.7), while the 2008 data show slightly more dispersion; however, the trend remains the same (Figure 2.8). This could potentially be due to year to year climatic variation or simply due to increased levels of regeneration.

My findings suggest that the assemblage of carabid beetles in the study area largely perform similar functional roles, in that they are large, brachypterous, forest generalists, and therefore the results of a functional approach may not yield useful additional information. However, this may bode well for a recognizable taxonomic unit approach as the resolution that may be lost by not being able to distinguish between different specialist species will not be an issue if there are only generalists. Further, the most informative response variable was carabid beetle activity-abundance, a value that does not change in any of the alternative approaches.

Management implications from this study may be quite profound, assuming that the response seen by carabid beetles is representative of other taxa. With regard to semi-open treatments (dispersed retention), managers must assess the cost and benefit of such a harvest practice as there was no significant difference in carabid activity-abundance and species richness achieved from a traditional clearcut. Gap treatments, however, appear to be extremely promising as they maintained both activity-abundance and richness of

carabids. This may prove useful for harvesting high-value timber from ecologically sensitive areas (such as old-growth forests). The results of this study suggest that gap harvesting maintains habitat structure for beetles, because it retains a structure most similar to the original forest, which is of excellent value in conservation of biodiversity. Dispersed retention harvesting appears to yield little value in this context. This may be because gap canopy treatments more closely resemble gap disturbance mechanisms that primarily consist of wind-throw events, while major stand-replacing disturbance events such as the fire are infrequent (D'Anjou, 2002).

The findings and recommendations of this study are detailed but have come with substantial effort. Such an investment may not be financially or logistically possible for the purpose of evaluating and/or monitoring post-disturbance. For this reason, a simpler and more accessible approach, such as a recognizable taxonomic unit approach, may be of interest. Alternatively, even when resources are available to conduct similar bioindication studies to the one I have presented here, a taxonomic approach to species richness and abundance does not address different habitat requirements. It therefore may be of interest to examine the functional roles of the beetles or at least examine more closely how they interact with their environment. This more detailed information may be obtained through a functional approach. In subsequent chapters I will explore the alternative approaches of functional and recognizable taxonomic unit approaches to bioindication while this study will be used as a benchmark to which I will compare their results.

Literature Cited

Abildsnes, J. & B.Å. Tømmerås. 2000. Impacts of experimental habitat fragmentation on ground beetles (Coleoptera, Carabidae) in a boreal spruce forest. Annales Zoologici Fennici 37:201-212.

Allegro, G. & R. Sciaky. 2003. Assessing the potential role of ground beetles (Coleoptera, Carabidae) as biondicators in poplar stands, with a newly proposed ecological index (FAI). Forest Ecology and Management 175:275-284.

Butterfield, J., M.L. Luff, M. Baines, & M.D. Eyre. 1995. Carabid beetle communities as indicators of conservation potential in upland forests. Forest Ecology and Management 79:63-77.

Colwell, R.K. 1997. EstimateS: Statistical estimation of species richness and shared species from samples, version 5. User's guide and application published at: http://viceroy.eeb.uconn.edu/estimates.

Craig, K. G. 1995. Variation in carabid community structure associated with coastal Douglas-fir forest successional stages. M.Sc. thesis. University of British Columbia.

D'Anjou, B. 2002. Roberts Creek Study Forest: harvesting, windthrow and conifer regeneration within alternative silvicultural systems in Douglas-fir dominated forests on the Sunshine Coast. Research Section, Vancouver Forest Region, B.C. Ministry of Forests, Nanaimo, B.C. Technichal Report TR-018/2002.

Digweed, S.C., C.R. Currie, H.A. Carcamo, & J.R. Spence. 1995. Digging out the "digging-in effect" of pitfall traps: influences of depletion and disturbance on catches of ground beetles (Coleoptera: Carabidae). Pedobiologia 39:561-576.

Finch. O. 2005. Evaluation of mature conifer plantations as secondary habitat for epigeic forest arthropods (Coleoptera: Carabidae; Aranae). Forest Ecology and Management 204:21-34.

Heliölä, J., M. Koivula, & J. Niemelä. 2001. Distribution of Carabid Beetles (Coleoptera, Carabidae) across a Boreal Forest-Clearcut Ecotone. Conservation Biology 15:370-377.

Klimaszewski, J., D.W. Langor, T.T. Work, G. Pelletier, H.E.J. Hammond, & C. Germain. 2005. The effects of patch harvesting and site preparation on ground beetles (Coleoptera, Carabidae) in yellow birch dominated forests of southeastern Quebec. Canadian Journal of Forest Research 35:2616-2628.

Koivula, M. 2002. Boreal carabid-beetle (Coleoptera, Carabidae) assemblages in thinned uneven-aged and clear-cut spruce stands. Annales Zoologici Fennici 39:131-149.

Larochelle, A. & M.-C. Larivière. 2003. A natural history of the ground-beetles (Coleoptera: Carabidae) of America north of Mexico. Sofia, Bulgaria : Pensoft.

Larsen, K.J., T.T. Work, & F.F. Purington. 2003. Habitat use patterns by ground beetles (Coleoptera: Carabidae) of northeastern Iowa. Pedo Biologia 47:288-299.

Lemieux, J. & B.S. Lindgren. 2004. Ground beetle responses to patch retention harvesting in high elevation forests of British Columbia. Ecography 27:557-566.

Lindroth,C.H. 1961-1969. The ground-beetles (Carabidae, excl. Cicindelinae) of Canada and Alaska, Opuscula Entomologica Supplements, 20 (1961), 24 (1963), 29 (1966), 33 (1968), 34 (1969), & 35 (1969). Entomologiska Sällskapet, Lund, Sweden.

McCune, B. & J.B. Grace. 2002. Analysis of ecological communities. MjM Software.

McCune, B. & M. J. Mefford. 2006. PC-ORD. Multivariate Analysis of Ecological Data. Version 5. MjM Software, Gleneden Beach, Oregon, U.S.A.

McGeoch, M.A. 1998. The selection, testing and application of terrestrial insects as bioindicators. Biological Review 73:181-201.

Meidinger, J. & J. Pojar. 1991. Ecosystems of British Columbia. B.C. Ministry of Forests Special Report 6.

Meggs, J.M. 2007. Ground Arthropod Sampling Methods, Roberts Creek Study Forest Pilot, Coast Regional Experimental Arthropod Project (CREAP). Unpublished Report.

Moore, J., Ouimet, R., Houle, D., & Camiré, C. 2004. Effects of two silvicultural practices on ground beetles (Coleoptera, Carabidae) in a northern hardwood forest, Quebec, Canada. Canadian Journal of Forest Research 34:959-968.

Niemelä, J., D. Langor, & J.R. Spence. 1993. Effects of clearcut harvesting on boreal ground-beetle assemblages (Coleoptera: Carabidae) in western Canada. Conservation Biology 7:551-561.

Pearce, J.L., D. Schuurman, K.N. Barber, M. Larrivée, L.A. Venier, J. McKee, & D. McKenney. 2005. Pitfall trap designs to maximize invertebrate captures and minimize captures of nontarget vertebrates. Canadian Entomologist 137:233-250.

Pearsall, I. A. 2006. Utility of Ground Beetles (Coleoptera: Carabidae) as Indicator Species for Monitoring Biodiversity Effects from Variable Retention Harvesting Practices. Available from: http://www.for.gov.bc.ca/hfd/library/FIA/2006/FSP_Y061029a.pdf

Rainio, J. & J. Niemelä. 2003. Ground beetles (Coleoptera: Carabida) as bioindicators. Biodiversity and Conservation 12:487-506.

Schowalter, T.E., Y.L. Zhang & J.J. Rykken. 2003. Litter invertebrate responses to variable density thinning in Washington forests. Journal of Applied Ecology 13:1204-1211.

Sprules, W.G. 1980. Nonmetric multidimensional scaling analysis of temporal variation in the structure of limnetic zooplankton communities. Hydrobiologia 69:139-146.

Todd, M.A., F.L. Waterhouse, J.M. Meggs, S.C. Saunders, B.S. Lindgren, & B.G. Marcot. 2008. Coast Region Experimental Arthropod Project: monitoring ground arthropod functional communities to evaluate the effectiveness of structural retention for biodiversity conservation. Ministry of Forests and Range, Coast Region Research Section, Annual Workplan.

Work, T.T., M. Koivula, J. Klimaszewski, D. Langor, J. Spence, J. Sweeny, & C. Hebert. 2008. Evaluation of carabid beetles as indicators of forest change in Canada. Canadian Entomologist 140:393-414.

Chapter 3- Carabid Beetles in BC Coastal Forests: An Evaluation of Recognizable Taxonomic Units for Bioindicator Studies.

3.1 Abstract

Carabid beetles have long been regarded as excellent candidates for use as bioindicators. Unfortunately, there is an ever increasing shortage of trained taxonomists, and often those available do not fit within the tight budget of investigators conducting bioindication studies or monitoring. An approach that does not require highly trained taxonomists to process samples involves the use of morphospecies, or more accurately "recognizable taxonomic units" (RTU), although this approach is often criticised. I evaluate the utility of a RTU approach using carabid beetles, and compare the results to a taxonomic study using the same data set. Using analysis of variance of recognizable taxonomic unit diversity and carabid beetle activity-abundance, as well as non-metric multidimensional scaling, I compare carabid beetle response to different canopy closure types in the Robert's Creek Study Forest, Sunshine Coast, British Columbia. The results show that, based on recognizable taxonomic unit diversity, closed and gap canopy closure treatments could not be distinguished from semi-open and open canopy closure treatments. The treatments could, however, be distinguished based on mean carabid beetle activity-abundance by treatment type as this analysis remained unchanged from the study using taxonomic species. In addition, canopy closure percent as a covariate appears to explain recognizable taxonomic unit diversity better than canopy closure treatment type alone. These findings suggest that the recognizable taxonomic unit approach is a questionable surrogate for a complete description of species composition, at least in areas with relatively low species diversity within a taxon such as carabids.

3.2 Introduction

Arthropods have long been of interest to scientists seeking more efficient ways of monitoring and assessing ecosystems. In practice, they have often been set aside due to large gaps in taxonomical resources, e.g., identification keys, and an ever increasing shortage of taxonomic expertise (Derraik et al., 2002). A proposed approach to overcoming these obstacles is to sort species into recognizable taxonomic units (RTU), often referred to as 'morphospecies'. This is achieved by sorting arthropods based on recognizable morphological traits that can be identified easily by researchers or technicians that are not taxonomists (Oliver & Beattie, 1996; Derraik et al., 2002).

Grimbacher et al. (2008) found that higher taxonomic groupings (i.e., RTUs) could be used to distinguish different habitat types (e.g., pasture versus interior tropical forest), but their efficacy decreased when attempting to distinguish between treatment types with more subtle differences. The efficacy of the recognizable taxonomic unit approach was further decreased by lowering the taxonomic resolution. For example, simply using body size was less effective than species sorted by a number of morphological characteristics. These findings confirm that the use of RTUs and the level to which they need to be sorted depends largely upon the objectives of the study or program, and also may depend on the diversity of arthropods across the study site.

While the RTU approach could result in considerable efficiencies, it has received criticism. Krell (2004), for example, points out that while many studies find that RTU

diversity compares quite well to taxonomic diversity, this is often more due to luck or chance. Often in these studies the sorting to RTU has quite low accuracy, i.e., many single species are split into two or more species while in other instances two or more species may be lumped together. The number of splits and lumps often balances and thus, the number of RTUs compares fairly well to numbers of taxonomically identified species. This balancing of splitting and lumping leads to the charge that the comparable results are mostly due to chance. Furthermore, the accuracy of the RTU approach appears to decline when using taxa that are difficult even for trained taxonomists to properly sort (Grimbacher et al., 2008).

Krell (2004) also raises the issue of reproducibility, a fundamental requirement of science. Many RTU studies would simply assign individuals to groups by numbers or arbitrary names (e.g. Morphospecies #1, Morphospecies #2, etc). This prevents other researchers from reproducing the results, as the determined differences between each "morphospecies" would not have been noted. Krell (2004) argues that this approach to naming is counterproductive, as it causes confusion in terms of the identification and naming of species in addition to not being repeatable. For this reason, international guidelines for the naming of species were established.

Many of the criticisms of Krell (2004) and others can be addressed simply through the determination of whether or not identification to species is necessary for the objectives of the study to be met. For instance, in a rapid biodiversity survey it is not necessary to

name every species, but the study should be repeatable. There also must be some form of safeguards in place to ensure that species are not being lumped or split incorrectly.

The RTU approach may be improved by deviating from its classic definition and beginning to resemble a taxonomic approach while still maintaining the aspects that made it appealing to begin with. For example, instead of no or very limited training, parataxonomists will need to be given some level of standardized training, RTUs will have to be appropriately named, and a key must be developed to ensure reproducibility. Ideally, this approach would lead to similar results as a taxonomic diversity study, while consuming far less time and resources. The objective of my study is to evaluate the sensitivity of using recognizable taxonomic unit diversity, within a discrete taxonomic group such as the carabid beetles, for detecting various levels of ecosystems disturbance in a coastal BC forest. Specifically, the ability of this approach to discriminate among different canopy treatment types will be compared to the results of a traditional taxonomic diversity study in the same forest (Chapter 2).

3.3 Methods

3.3.1 Recognizable Taxonomic Unit Identification

Using the data set described in Chapter 2, carabid beetles were separated out during the parataxonomic sort of pitfall trap samples, and then sorted into easily recognizable taxonomic units based on morphological features that could be identified with the naked eye consistently and easily (Todd & Meggs 2008). Features included drastic size differences, striation of the elytra, and sculpture of the elytra. Some of the RTU

groupings were equivalent to easily recognizable species or genera. The resulting RTU groups were composed of: *Cychrus tuberculatus* (Ctuberculatus- species), *Scaphinotus* sp. (Sangusticollis- species), *Omus dejeani* (Odejeani- species), Large *Pterostichus* sp.(>17mm) (Plama- Large species of *Pterostichus* genus), Medium *Pterostichus* sp. (7-17mm) (MedPtero- Medium species of *Pterostichus* genus), and Small carabid species (<7mm) (SmCarabi- e.g. *Notiophilus sylvaticus*). The reproducibility of identifying these groups was ensured through the use of a key (Todd & Meggs, 2008).

Table 3.1: Characteristics used to identify recognizable taxonomic units

RTU Group	**Identifying Characteristics**
Ctuberculatus	Bumpy formations on elytra, have appearance of water droplets
Sangusticollis	Smooth elytra with light striation, purple/red hue to elytra.
Odejeani	Dimples in elytra, very large mandibles
Plama	>17mm, striated elytra
MedPtero	7-17mm, striated elytra
SmCarabi	Carabids <7mm

3.3.2 Analysis

Analysis of covariance (ANCOVA) was performed on mean carabid beetle RTU richness and activity-abundance per trap array by canopy closure treatment and canopy closure percent. To examine all pairwise comparisons, protected *t*-tests were employed. It was not necessary to transform diversity data to meet the assumptions of the ANCOVA model (equal variance and normal distribution of errors) as the data met those criteria. Activity-abundance data had to be log transformed to meet these assumptions, however. Confirmation that the assumptions were met was achieved through visual inspection of residual plots.

Non-metric Multidimensional Scaling (NMS) was performed using PC-ORD 5.31 software (McCune & Mefford, 2006) in order to explore differences in species assemblages by both canopy closure percent and canopy closure type. A more detailed description of NMS may be obtained in Chapter 2: Materials and Methods.

3.4 Results

Because all carabid beetles were utilized in this analysis, the total number beetles in all six RTUs mirrors that presented in Chapter 2. In total, 1425 carabid beetles were captured in 92 traps in 2007 and 1108 were captured in 96 traps in 2008. These beetles represented all 6 recognizable taxonomic units in each of the sampling years. The *Scaphinotus* sp. and Medium *Pterostichus* sp. groups accounted for the vast majority of sampled carabids, representing 91.3% and 83.4% in 2007 and 2008, respectively (Tables 3.1 and 3.2).

For the samples collected in 2007, ANCOVA showed that the average diversity of carabid beetle recognizable taxonomic unit could be explained by a significant relationship with canopy treatment type ($F_{3,18} = 8.3609$, $P = 0.001$), while canopy closure percent was not significant ($F_{1,18} = 3.8363$, $P = 0.066$). The 2008 ANCOVA model approached significance for the canopy treatment type component ($F_{3,19} = 3.0811$, $P = 0.052$), and was also not significant for the canopy closure percent ($F_{1,19} = 2.5674$, $P = 0.126$).

For the data collected during 2007, the comparison of mean scores using the protected *t*-tests showed that closed canopy treatments were not significantly different from gap treatments ($t_{1,10} = 0.227$, $P = 0.82$), semi-open ($t_{1,10} = -1.953$ $P = 0.0666$), or open ($t_{1,10} = -1.826$, $P = 0.0845$) canopy treatments (Figure 3.1). Gap canopy treatments were significantly different from semi-open ($t_{1,10} = -2.250$, $P = 0.0372$) and open ($t_{1,10} = -2.104$, $P = 0.050$) canopy treatments while semi-open and open canopy treatments were not significantly different ($t_{1,10} = 0.268$, $P = 0.79$). Means comparisons were not conducted for 2008 as the ANCOVA model did not produce a significant result.

Non-metric multidimensional scaling (NMS) of the 2007 capture data produced a two dimensional solution which accounted for 85.5% of the variation where the axes represented 44.2% and 41.3% of the variance, respectively. Monte Carlo tests (500 runs) of whether axes represented non-random solutions were $P = 0.0518$ and $P = 0.0438$, respectively. Analysis of 2008 data yielded a solution where the first two axes accounted for 46.2% of the variation in the data. The axes represented 34.6% and 11.6% of the variance, respectively. Monte Carlo tests to determine if the axes represented non-random solutions were $P = 0.016$ and $P = 0.044$, respectively. A significant *P*-value suggests a relationship between the trap location of RTU, treatment type, and percent canopy closure. In both years, closed and gap canopy treatments cluster together, apart from semi-open and open canopy treatments which also cluster together.

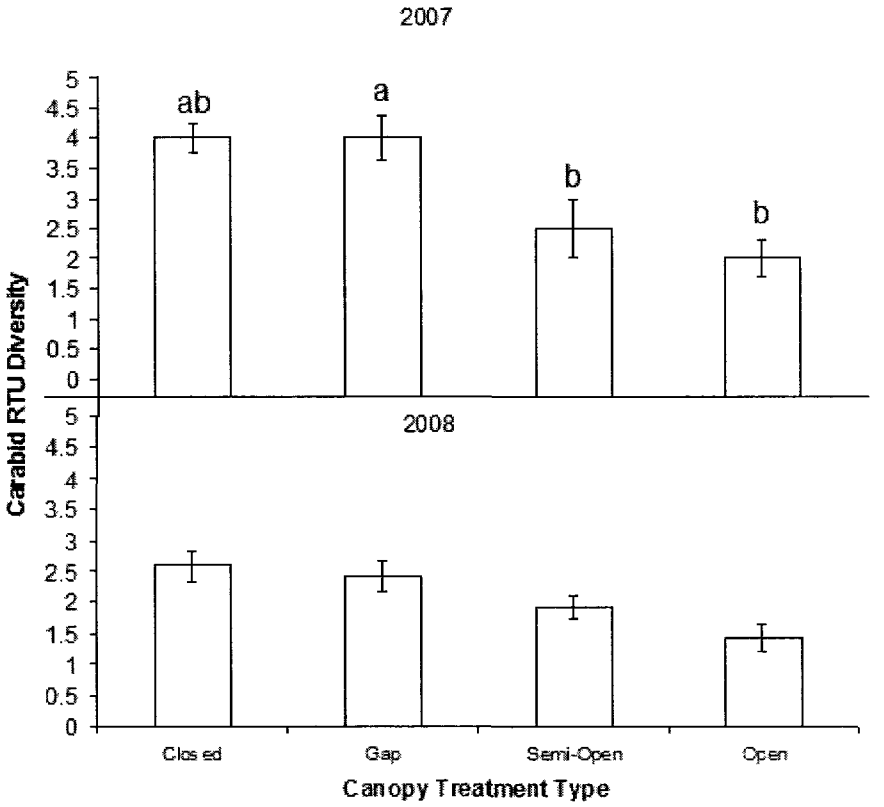

Figure 3.1 Comparison of mean number of recognizable taxonomic units per trap array by canopy closure treatment from 2007 and 2008 samples. For significant ANCOVA ($\alpha = 0.05$), bars denoted with the same letter are not significantly different.

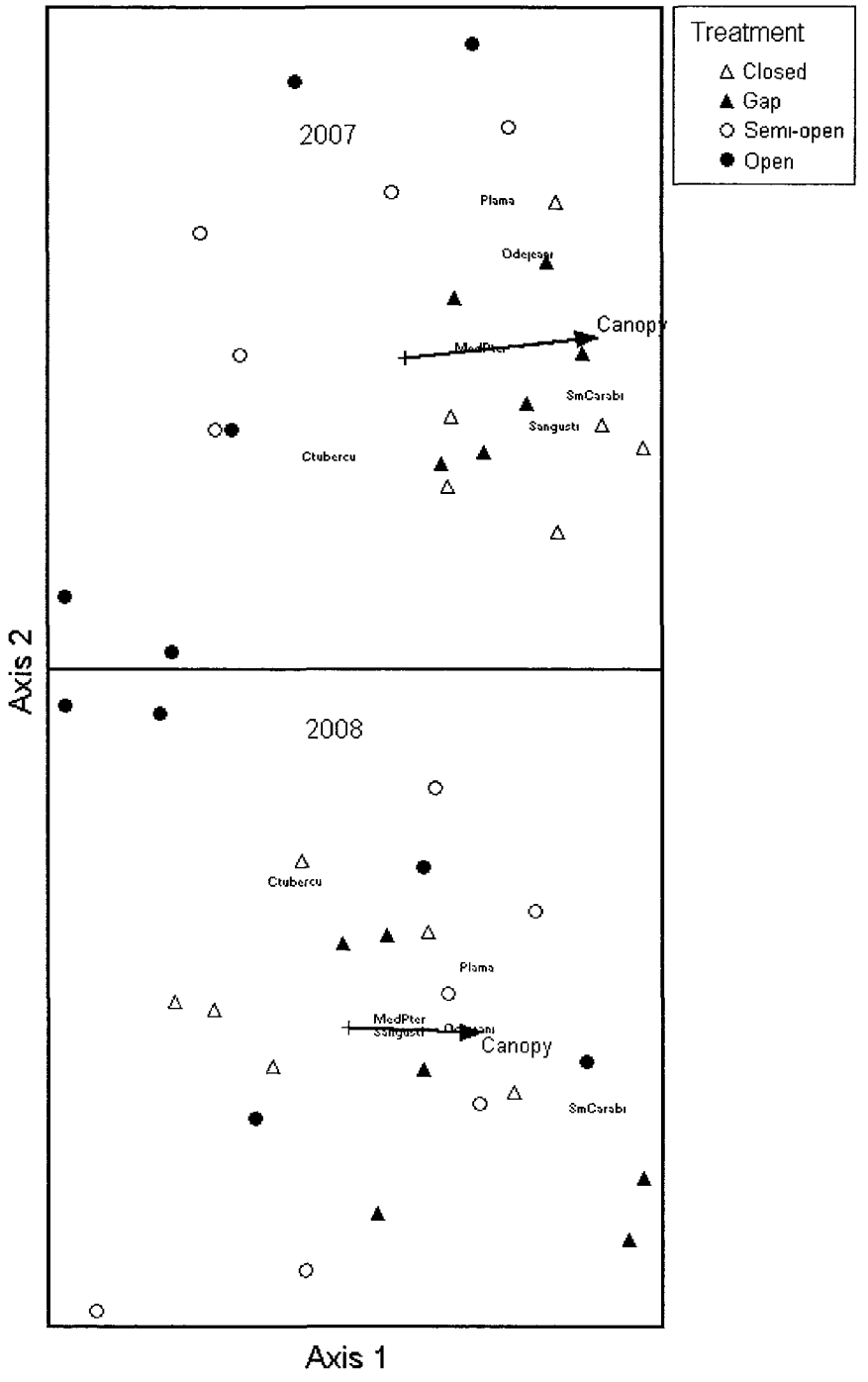

Figure 3.2 Non-metric multidimensional scaling models of RTUs based on canopy closure type and canopy closure percent from 2007(A) and 2008(B). Recognizable taxonomic units are grouped with the trap arrays (individual circles and triangles) that they are most strongly associated with.

Table 3.2: Number of individuals of each recognizable taxonomic unit caught by canopy treatment type in 2007.

RTU	Closed	Gap	Semi-Open	Open*	Total	% of Catch
Ctuberculatus	15	6	6	19.20	46.2	3.22
Sangusticollis	103	30	0	0.00	133	9.27
Odejeani	15	23	7	4.80	49.80	3.47
Plama	5	0	6	9.60	20.60	1.44
MedPtero	480	568	90	32.40	1170.40	81.62
SmCarabi	7	6	1	0.00	14.00	0.98

*Catch numbers were standardized in order to correspond with treatments where 6 sites were present as opposed to 5 in open treatment.

Table 3.3: Number of individuals of each recognizable taxonomic unit caught by canopy treatment type in 2008.

RTU	Closed	Gap	Semi-Open	Open	Total	% of Catch
Ctuberculatus	11	7	5	11	34	3.07
Sangusticollis	135	51	1	0	187	16.88
Odejeani	16	42	45	18	121	10.92
Plama	1	1	7	9	18	1.62
MedPtero	264	364	70	39	737	66.52
SmCarabi	4	6	1	0	11	0.99

3.5 Discussion

The six recognizable taxonomic units used in this study could be identified with only minor training, but were not able to distinguish even between the extreme canopy treatment types in 2007. The activity-abundance data, which was significant, remained the most informative response variable and only requires the identification to the level of being a carabid beetle. Consistent with the findings of Chapter 2, this suggests that treatments that maintain canopy structure tend to maintain diversity and overall activity-abundance of carabid beetles more so than treatments that more dramatically alter canopy structure. These findings were mirrored by taxonomic diversity studies such as Klimaszewski et al. (2005) (gap harvest treatments) and Moore et al. (2004) (strip clearcuts and selective harvesting) as well as the taxonomic and abundance study findings of Chapter 2. However, in 2008 there was no significant difference between any of the treatment types with regard to RTU diversity.

With regard to diversity, the results of the analysis using RTU in this study are different from the results of the carabid beetle taxonomic diversity study (Chapter 2). The recognizable taxonomic unit results were not as informative as the taxonomic diversity results. For carabids, significant differences in activity-abundance and taxonomic diversity measures between very different canopy closure types (i.e., closed and gap versus semi-open and open) could be reliably identified by the taxonomic approach. Grimbacher et al. (2008) found that higher beetle taxonomic units were only useful for differentiating treatments with a high degree of contrast. However, canopy closure percent as a covariate explains much of the RTU diversity, as it did with respect to

taxonomic diversity (Chapter 2). This suggests that the species are responding more so to the canopy closure percentage at the trap array level rather than to the overall conditions of the treatment.

The results of the non-metric multi-dimensional scaling (NMS) analyses were similar to the taxonomic study (Chapter 2). A distinction between closed/gap canopy treatments and semi-open/open canopy treatments occurred in both years (Figure 3.2). Resulting management decisions could potentially be less conservative if managers sought to conserve large groups like the medium *Pterostichus* sp. group, as they were found to persist in all treatment types. However, the NMS results of this analysis were very similar to those obtained in the Chapter 2 taxonomic analysis. This is because while only 6 RTU groups could be used in this analysis, the taxonomic analysis was also limited to a sample of 6 species due to the other species being too rare. When concerned with potentially sensitive or threatened rare species, NMS appears to offer limited to no insight.

Unfortunately, the RTU approach does not allow for repetition of some analyses utilized in the taxonomic diversity approach. Species rarefaction curves that were used to evaluate sampling effort are of no value as the number of recognizable taxonomic units is a function of the categories arising from trap samples, i.e., in my study no more than six carabid recognizable taxonomic units could be identified from any trap. This is because the most likely missing species are those that fall within the small carabid group and would therefore simply be lumped with that group. In addition, species evenness is of little use as instead of evaluating the evenness of the diversity and activity-abundance of

one to 14 species in each of the years it would only evaluate the evenness of the diversity and activity-abundance of one to six recognizable taxonomic units. The loss of this information must be considered by those considering employing a recognizable taxonomic unit approach.

The majority of past studies that have focussed on recognizable taxonomic units have concerned themselves with accuracy. That is, they have been concerned with comparing the number of recognizable taxonomic units identified to the number of species identified by a taxonomic approach (Oliver & Beattie 1993,1996, and Derraik et al. 2002). The two reasons for differing results were splitting (the act of identifying one species as two or more recognizable taxonomic units) and lumping (the act of identifying two or more species as one recognizable taxonomic unit).

In my study, I have overcome splitting and lumping issues that were described in past studies through the use of clearly defined recognizable taxonomic units. Parataxonomists have been able to use a key that allows them to sort carabid beetles to recognizable taxonomic units that are defined either by easily distinguishable genus or size (Todd & Meggs, 2008). Some species are therefore lumped together, but this has been incorporated in the study design from the beginning and there need not be concern over some parataxonomists splitting while others are lumping and each ending up with different beetles in different groupings. The accuracy of parataxonomists was extremely high, with inexperienced individuals rarely needing to be corrected (personal observation).

One of the chief criticisms of any recognizable taxonomic unit approach in the past has been that of repeatability. While all bioindication studies require calibration in the study area in question prior to drawing conclusions or implementing a monitoring program, it is expected that following this calibration the study may be repeated in the same area and yield similar results. In similar past studies, repeatability has been an issue, as depending upon the investigator, differing numbers of groups may be generated with differing amounts of species within them. This obvious flaw is sharply criticized by Krell (2004). These concerns were also alleviated in this study through the use of a key (Todd & Meggs, 2008), which allows for repetition within the Robert's Creek Study Forest. Having addressed potential sources of error in this manner, the recognizable taxonomic unit approach may be a viable surrogate for taxonomic diversity in the face of limited funding and limited resources, but it must be refined with an even more detailed key to distinguish among treatments more effectively. However, until such refinements are made, the taxonomic approach appears to offer a much more reliable and insightful analysis at this resolution for this study area and sampling period. Nevertheless, as only a single family (Carabidae) with very low diversity was used in this study, combined with limited habitat information, it is promising that differences could still be detected. Thus, in situations where only coarse response variables are required, the RTU approach may be a viable substitute for a traditional taxonomic approach if a sufficiently diverse fauna is examined.

Literature Cited

Derraik, J.G.B., G.P. Closs, K.J.M. Dickinson, P. Sirvid, B.I.P. Barratt, & B.H. Patrick. 2002. Arthropod morphospecies versus taxonomic species: a case study with Araneae, Coleoptera, and Lepidoptera. Conservation Biology 16:1015-1023.

Grimbacher, P.S., C.P. Catterall, & R.L. Kitching. 2008. Detecting the effects of environmental change above the species level with beetles in a fragmented tropical rainforest landscape. Ecological Entomology 33:66-79.

Klimaszewski, J., D.W. Langor, T.T. Work, G. Pelletier, H.E.J. Hammond, & C. Germain. 2005. The effects of patch harvesting and site preparation on ground beetles (Coleoptera, Carabidae) in yellow birch dominated forests of southeastern Quebec. Canadian Journal of Forest Research. 35: 2616-2628.

Krell, F. 2004. Parataxonomy vs. taxonomy in biodiversity studies – pitfalls and applicability of 'morphospecies' sorting. Biodiversity and Conservation 13:795-812.

McCune, B. & M. J. Mefford. 2006. PC-ORD. Multivariate Analysis of Ecological Data. Version 5. MjM Software, Gleneden Beach, Oregon, U.S.A.

Moore, J., Ouimet, R., Houle, D., & Camiré, C. 2004. Effects of two silvicultural practices on ground beetles (Coleoptera, Carabidae) in a northern hardwood forest, Quebec, Canada. Canadian Journal of Forest Research 34:959-968.

Oliver, I., & A.J. Beattie. 1993. A Possible Method for the Rapid Assessment of Biodiversity. Conservation Biology 7:562-568.

Oliver, I. & A.J. Beattie. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. Ecological Applications 6:594-607.

Todd, M.A. & J.M. Meggs. 2008. Parataxonomic key to the ground arthropods of the Roberts Creek Study Forest, British Columbia. Unpublished document available from M.A. Todd, BC Ministry of Forests and Range, Nanaimo, BC, 23 pp.

Chapter 4- Carabid Beetles in BC Coastal Forests: An Evaluation of Functional Diversity for Bioindicator Studies.

4.1 Abstract

One of the chief criticisms of using bioindicators is that the results are often not comparable across different study areas due to differing species pools. Functional similarity allows for this comparison to occur through the identification of species that fill similar functional roles within each ecosystem. I used morphological trait measurements of carabid beetles from the Robert's Creek Study Forest located on the Sunshine Coast of British Columbia to infer functional attributes of the beetles. RLQ analysis, a statistical technique using three tables titled R, L, and Q, was employed to examine how these morphological traits were distributed within four canopy cover treatment types of the study forest (closed (control), gap, semi-open, and open) in order to draw inferences on how carabid beetles with differing functional attributes were affected by harvest treatments. No significant results were obtained from the analysis, although some trends were evident. One of 15 species captured in the study (*Scaphinotus angusticollis*) appears to be a forest specialist, while the majority of species seem to be able to persist in two or more canopy treatment types. While the functional trait approach may have been useful in identifying specialists and generalists had the results been significant, it did not provide meaningful additional information over a traditional bioindicator based on taxonomic groupings (Chapter 2), and involved a significantly higher investment of time and effort. It is therefore not recommended for taxa likely to have limited functional trait diversity.

4.2 Introduction

Beetles are the largest order of insects, occupying a considerable range of habitats around the world and filling many functional roles (Lassau et al., 2005). The use of functional diversity for bioindication in the context of ecosystem function is of interest. The predictive power of bioindicators is greater when each indicator has been assigned to a functional group, as this allows the results to be applied beyond the specific habitat in which the indicator species was found (Stephens & Wagner, 2006). Regional specificity in taxonomic diversity makes analysing trends difficult across large geographic areas. Functional grouping of arthropods overcomes obstacles that are encountered with taxonomic approaches. The ability to compare results from completely different geographic regions is invaluable when faced with managing large geographic areas as many governments are. For a minor additional investment of time and resources far more useful results may be yielded.

Sorting data into functional groupings may be particularly useful for detecting the effects of change on ecosystem function (Grimbacher et al., 2008), and may even make it possible to detect shifts in microhabitat and microclimate use (Gibb et al., 2006; Nittérus et al., 2007). The ability to detect such shifts would be invaluable from a management perspective as it would greatly improve monitoring and conservation practices. These types of effects are what many managers and researchers are seeking to understand because maintenance of ecosystem function is often a priority during conservation of resources outside of reserves or conservation areas. We cannot conserve all species

affected by habitat alteration or destruction in managed landscapes, but we can conserve many of the species by maintaining functioning ecosystems (Walker, 1995; Myers et al., 2000).

The approach examined in this chapter, the three table RLQ analysis, has largely been used in plant studies to date (Thullier et al., 2006; Bernhardt-Römerman, 2008). However, the same appeal of being able to relate functional traits (table Q) to environmental variables (table R) through an abundance matrix (table L) applies to invertebrate (Ribera et al., 2008) and vertebrate (Cleary et al., 2007) studies. Functional diversity approaches, which examine organisms that use similar niches within an ecosystem, must be an area of high priority for landscape managers. They go beyond diversity and activity-abundance responses and actually look at what specific functional requirements are being affected by disturbance. For some taxa this is critically important, as species diversity and richness may not necessarily be as informative for discriminating between treatment impacts as functional diversity (Stephens & Wagner, 2006). In some cases species diversity and richness may be nearly as high in a clearcut as it is in an unharvested control stand even though the species present may have made a complete shift from specialists in the control to generalists in the clearcut. The functional diversity approach may be able to discriminate between treatments even better than taxonomically determined diversity. My objective is to evaluate the utility of a functional diversity approach obtained from carabid beetles with respect to their response to ecosystem disturbance consisting of varying levels of canopy removal. Specifically, the ability of

this approach to provide additional information to a traditional taxonomic diversity approach in the same forest (Chapter 2) will be evaluated.

4.3 Methods

4.3.1 Carabid Morphological Trait Sampling

I used the method of Ribera et al. (2001) to conduct morphological trait sampling and then infer function by group from the data set described in Chapter 2. Measured morphological traits were used as an index of the carabid beetles' ecology. Species that made up less than 1% of the catch from the 2007 and 2008 samples were excluded from the study to avoid potential biases associated with rare species, leaving a total of 6 species. Two individuals of each species from each of three different randomly selected sites were chosen and assumed to be representative of the general size and shape for that species (n = 6 per species). Measurements of the antennae, eyes, hind legs, and body that are assumed to infer function were taken following the methodology of previous work (Forsythe, 1987; Ribera et al., 1999; Ribera et al., 2001; refer to Table 4.1 for a detailed description of quantitative measurements). Measurements of the eye and antennae were to represent sensory ability, while measurements of body size and legs were to represent mobility. Qualitative life history information, such as food source and overwintering life stage as described by Ribera et al. (2001), was not recorded due to a general lack of information surrounding sampled carabid species.

Measurements were taken using a dissecting microscope equipped with a micrometer at 10x magnification. Measurements were log transformed to normalize data and each

variable was subjected to a regression against the log of total length. The residuals from the regressions were recorded by species and were used as an approximate representation of shape (Ribera et al., 2001).

Table 4.1 Morphological measurement variables and their corresponding coded names (Directly quoted from Ribera et al. (2001))

Code	Variable*
LYW	Diameter of the eye, measured from above
LAL	Length of the antenna
LPW	Maximum width of the pronotum
LPH	Maximum depth ("vaulting") of the pronotum
LEW	Maximum width of the elytra
LFL	Length of the metafemur (with the articulation segments), from the coxa to the apex
LTR	Length of the metatrochanter
LRL	Length of the metatarsi
LFW	Maximum width of the metafemur
LTL	Total length (length of the pronotum in the medial line plus the length of the elytra, from the medial line of the scutellum to the apex)

*The residuals of these variables from a regression with LTL, along with LTL, were used in the analysis.

4.3.4 Data Analysis

RLQ analysis (Doledec et al., 1996) was used as a means of analyzing the morphological traits of species. This was done by analyzing where the species were found within the different canopy treatment types for 2007 and 2008 data. RLQ analysis is an ordination technique that allows for a simultaneous three table analysis by which table R, the environmental variables (canopy closure type and canopy closure percent: appendices A & C), and table Q, the species' morphological traits (appendices B & D), are related to one another through a link table L composed of species activity-abundances from pitfall data (appendix E) (Ribera et al., 2001) (See Appendices for table data). Species are given weighted scores based on where they most often associate with the environmental

variables and the average score is plotted along with the standard error. If species are associated with sites it is assumed that their morphological features serve a functional role in adapting them to that particular set of environmental variables. Following each RLQ analysis, a random permutation test using 1000 repetitions was conducted to test whether the results of the cross matrix between tables R and Q were due to chance. All analyses were conducted using the ADE-4 package (Thiolouse et al., 1997) in R statistical software (R Development Core Team, 2009).

Table 4.2 Carabid beetle species representing greater than 1% of total catch in 2007 and 2008.

Species	Code
1 *Cychrus tuberculatus* Harris	Ctuberculatus
2 *Omus dejeani* Reiche	Odejeani
3 *Pterostichus algidus* LeConte	Palgidus
4 *Pterostichus herculaneus* Mannerheim	Pherculaneus
5 *Pterostichus lama* Ménétriés	Plama
6 *Scaphinotus angusticollis* Mannerheim	Sangusticollis

4.4 Results

Results of the 1000 random permutation test determined that a significant number of values randomly generated were equal to or greater than those of the cross-matrix of tables R and Q ($P = 0.145$ and $P = 0.093$ for 2007 and 2008 data, respectively), indicating the results of the analysis were not statistically significant. Nevertheless, the first two axes of the RLQ analysis for the 2007 data explained 79.1% and 20.9% of the variation, respectively, while the first two axes for the 2008 data accounted for 94.1% and 5.9% of the variation, respectively from the 36 carabid beetles sampled.

Pitfall array sites were ordered by the RLQ analysis along the first ordination axis based on their scores. Closed and open canopy treatments represented the two extreme ends of scores (-1.127 = closed canopy; 4.999 = open canopy; Figure 4.1). For 2008, the extremes of the RLQ analysis pitfall array plot scores were again represented by closed canopy (-1.13) and open canopy treatments (3.86).

Although the RLQ analysis was not significant, the first axis was positively correlated with maximum pronotum width (LPW), maximum pronotum height ("vaulting") (LPH), length of the metatrochanter (LTR), maximum femur width (LFW), and total length (LTL). In addition, negative correlations were found with antenna length (LAL) and femur length (LFL) (Table 4.3). The first axis of the RLQ analysis represents the largest portion of the solution from the model and is typically what is reported. Significance is reported at the overall model level, while individual correlations are deemed to be positive or negative in nature. In 2008, morphological traits also resulted in relatively strong correlations with the first axis of the RLQ analysis (Table 4.4). Antenna length (LAL), maximum width of the elytra (LEW), femur length (LFL), and length of the metatarsi (LRL) were all negatively correlated with the first axis, while eye width (LYW), width of the pronotum (LPW), height of the pronotum ("vaulting") (LPH), length of the metatrochanter (LTR), maximum width of the femur (LFW), and total length (LTL) were positively correlated with the first axis.

For both years, species were found on the first axis at the weighted average of the scores of their pitfall array sites (Figures 4.1 and 4.2). While trends were observed, these were not significant. Only *Scaphinotus angusticollis* appeared to be associated with a single

canopy closure type (closed). *Pterostichus algidus* and *P. herculaneus* are centered between closed and gap canopy types suggesting that they could be associated primarily with these canopy closure types. *Pterostichus lama* is centered around the semi-open canopy closure type but has a standard error that extends from gap to open canopy closures.

Table 4.3 2007 morphological trait correlation with the first ordination axis of the RLQ analysis.

Morphological Trait	Axis 1
LYW- eye width	-0.015
LAL- antennae length	-0.193
LPW- pronotum width	0.199
LPH- pronotum height	0.133
LEW- width of the elytra	-0.034
LFL- femur length	-0.204
LTR- trochanter length	0.130
LRL- length of the metatarsi	-0.099
LFW- femur width	0.180
LTL- total length	0.159

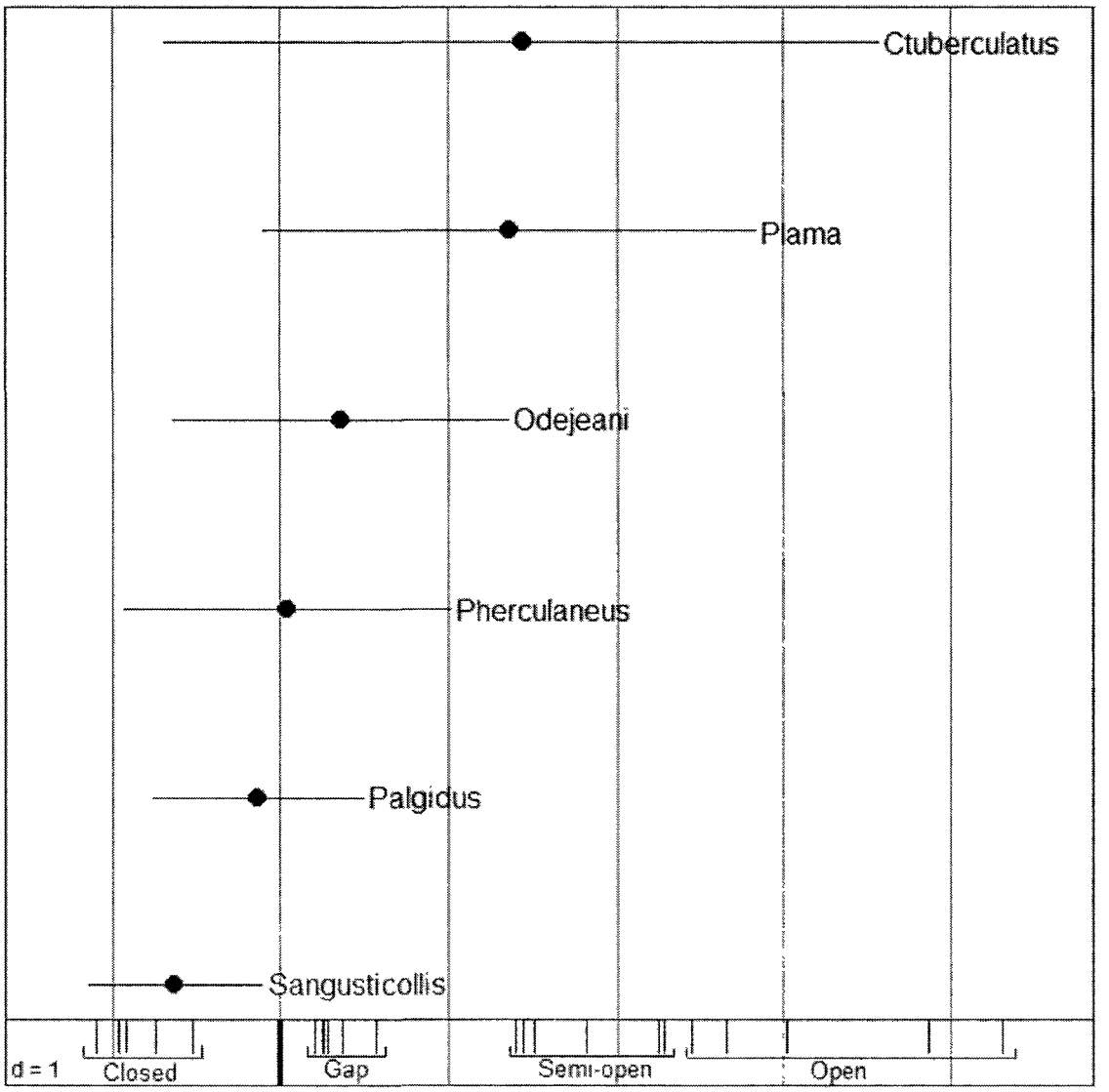

Figure 4.1 First axis of the 2007 RLQ analysis. Site scores of canopy closure type. Species are plotted by the weighted average of the scores of the sites in which they are found. Vertical lines within brackets for different treatments denote the score associated with individual trap array plots.

Table 4.4 2008 morphological trait correlation with the first ordination axis of the RLQ analysis.

Morphological Trait	Axis 1
LYW- eye width	0.142
LAL- antennae length	-0.141
LPW- pronotum width	0.417
LPH- pronotum height	0.255
LEW- width of the elytra	-0.053
LFL- femur length	-0.299
LTR- trochanter length	0.220
LRL- length of the metatarsi	-0.022
LFW- femur width	0.307
LTL- total length	0.134

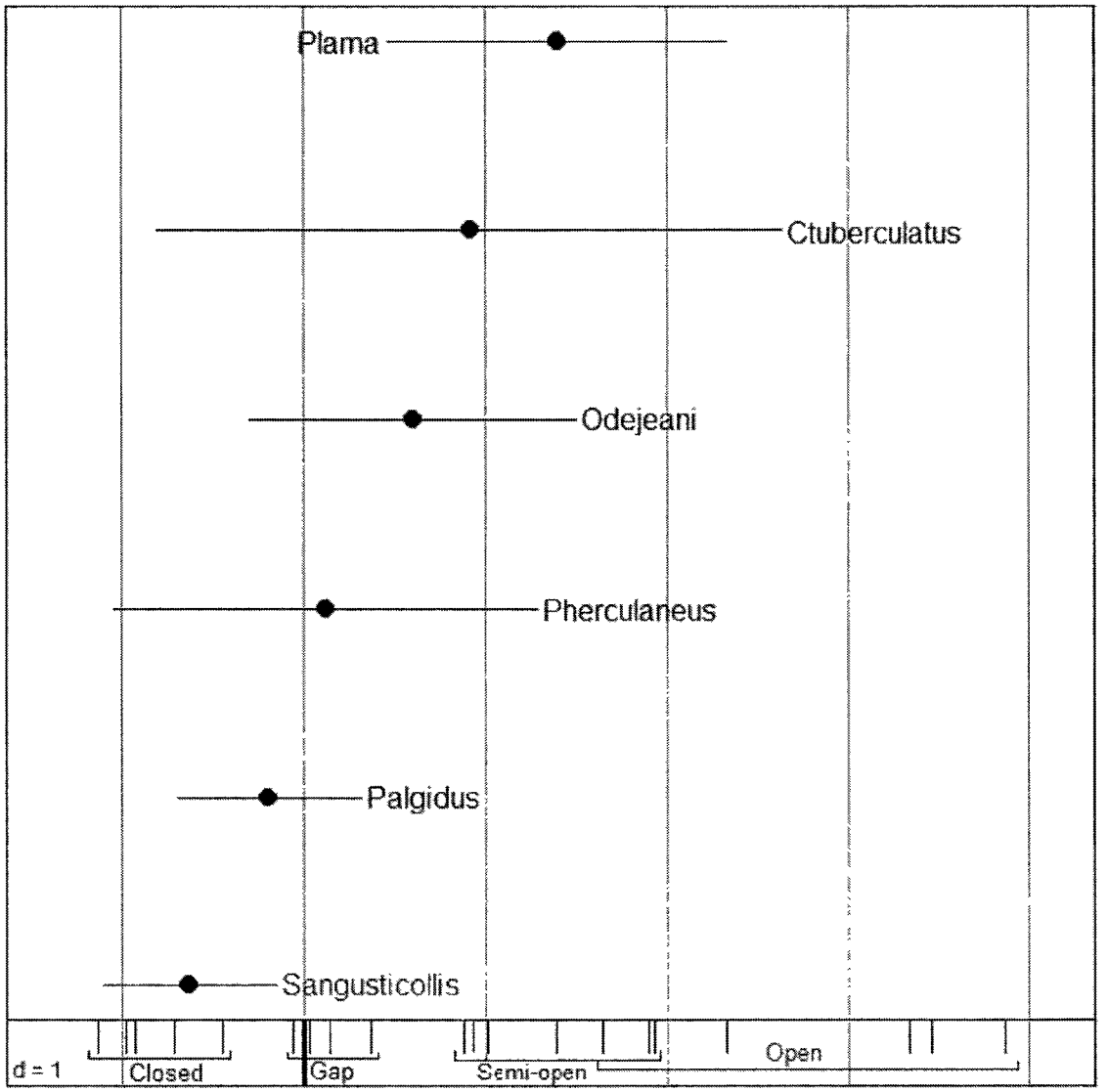

Figure 4.2 First axis of the 2008 RLQ analysis. Site scores of canopy closure type. Species are plotted by the weighted average of the scores of the sites in which they are found. Vertical lines within brackets for different treatments denote the score associated with individual trap array plots.

4.5 Discussion

Studies premised on beetle functional groups often use trophic role or feeding guild as the method for grouping (Gibb et al., 2006; Grimbacher et al., 2008). Alternatively, functional groups may be achieved through sorting to higher taxonomic units (e.g., genus or family) and generalizations about their functional role in the ecosystem can be made from available information (Lassau et al., 2005; Grimbacher et al., 2008). Finally, functional groups may be assigned to individual species with similar habitat requirements (Butterfield et al., 1995; Heliölä et al., 2001; Nittérus et al., 2007). These approaches all require sound life history information on the species or higher taxa in question, which unfortunately was not available for the species of this study. Ribera et al. (2001) offered an alternative approach by assuming that morphometric measurement variation of relevant structures represented different habitat specializations and therefore could be used as a surrogate for life history information. This would allow for the interpretation of specific habitat variables that are necessary for the conservation of species that were investigated.

The taxonomic approach (Chapter 2) appears, for this study area and for this particular situation, to provide the best indication of canopy condition. The addition of morphometric information did not improve the findings. This would seem to result from a strong species-specificity in morphometric variables. Therefore, even if the results of this approach had been significant, they simply highlighted aspects that were observed in the taxonomic study. For instance, *Scaphinotus angusticollis* appeared to be associated

with the control treatments, but this was already evident from the non-metric multidimensional scaling model using a traditional taxonomic approach (Chapter 2). The association of *S. angusticollis* with the control treatments in the functional approach may be due to its larger antennae and longer legs relative to the other species (Appendix E). One advantage with the RLQ analysis may be the visual representation of species affinities to various habitats (Figures 4.1 and 4.2). Nevertheless, the non-significant finding yielded by the functional approach required substantially more time and resources in addition to those used for the taxonomic study. Consequently, for the purposes of assessing this study area, and at the sampling intensity used, the functional approach does not make economic sense, at least not when restricted to carabid beetles. For a more accurate analysis of the benefit of a functional diversity approach the study should be repeated either at greater seasonal sampling intensity or in different geographic areas as increased carabid and ecosystem diversity may lead to different results, or using a higher level taxon, e.g., all beetles.

The RLQ analysis is very sparsely represented in the literature with Ribera et al. (2001) being the only substantial example that deals with arthropods. Based on the limited information available (this study and Ribera et al. (2001)) it appears as though the analysis provides greater detail of habitat association when there is a greater diversity of species and a greater number of environmental variables examined. For instance, the Ribera et al. (2001) study was conducted in 87 locations chosen to represent Scotland's ecosystems and had a sample that included 68 carabid species. The distinction of how different species use different habitats can be discerned from this approach; however, in

my study the same information was obtained already through the taxonomic approach (Chapter 2).

With the addition of further environmental variables, the functional approach may yield results that are much more refined in resolution and therefore may become increasingly appealing. The addition of variables such as soil moisture, woody debris, and shrub understory would likely serve this purpose. The variables of canopy treatment type and canopy closure percent were coarse, and could vary substantially in the case of canopy closure. Alternatively, the species diversity in this ecosystem may simply be too low to yield informative results. Ribera et al. (2001) found significant and very promising results using this approach, so the potential for the technique exists. It is possible that the lack of additional information yielded by the functional approach in my study is a by-product of both the relatively small number of carabid species represented, as well as the absence of many highly specialized carabid beetles within the assemblage of species I trapped. The species I trapped were forest generalists and had similar morphological measurements, so of the low diversity of beetles that were trapped the morphological diversity was also low. The absence of these highly specialized beetles, especially disturbance specialists, may simply be due to the absence of source populations in the area (Lemieux & Lindgren, 2004). Disturbance specialists would be expected to be sourced from forest fire sites or other sites disturbed by harvesting. It is possible that these types of species did not have a historical niche in the area and thus no source populations are present.

Literature Cited

Bernhardt-Römerman, M, C. Römerman, R. Nuske, A. Parth, S. Klotz, W. Schmidt & J. Stadler. 2008. On the Identification of the Most Suitable Traits for Plant Functional Trait Analysis. Oikos 117:1533-1541

Butterfield, J., M.L. Luff, M. Baines, & M.D. Eyre. 1995. Carabid beetle communities as indicators of conservation potential in upland forests. Forest Ecology and Management 79:63-77.

Cleary, D.F.R., T.J.B. Boyle, T. Setyawati, C.D. Anggraeni, E.E. van Loon, & S.B.J. Menken. 2007. Bird Species and Traits Associated With Logged and Unlogged Forest in Borneo. Ecological Applications 17:1184-1197.

Dolédec, S., D. Chessel, C.J.F. Ter Braak, & S. Champely. 1996. Matching species traits to environmental variables: a new three table ordination method. Environmental and Ecological Statistics 3:143-166.

Forsythe, T.G. 1987. The relationship between body form and habit in some Carabidae (Coleoptera). Journal of Zoology, London 211:643-666.

Gibb, H., R.B. Pettersson, J. Hjältén, J. Hilszczanski, J.P. Ball, T. Johansson, O. Atlegrim, & K. Danell. 2006. Conservation-oriented forestry and early successional saproxylic beetles: responses of functional groups to manipulated dead wood substrates. Conservation Biology 129:437-450.

Grimbacher, P.S., C.P. Catterall, & R.L. Kitching. 2008. Detecting the effects of environmental change above the species level with beetles in a fragmented tropical rainforest landscape. Ecological Entomology 33:66-79.

Heliölä, J., M. Koivula, & J. Niemelä. 2001. Distribution of Carabid Beetles (Coleoptera, Carabidae) across a Boreal Forest-Clearcut Ecotone. Conservation Biology 15:370-377.

Lassau, S.A., D.F. Hochuli, G. Cassis, & C.A.M. Reid. 2005. Effects of habitat complexity on forest beetle diversity: do functional groups respond consistently? Diversity and Distributions 11:73-82.

Lemieux, J. & B.S. Lindgren. 2004. Ground beetle responses to patch retention harvesting in high elevation forests of British Columbia. Ecography 27:557-566.

Myers, M., R.A. Mittermeier, C.G. Mittermeier, G.A.B. da Fonseca, & J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature 403:853-858.

Nittérus, K., M. Åström, & B. Gunnarsson. 2007. Commercial harvest of logging residue in clear-cuts affects the diversity and community composition of ground beetles (Coleoptera: Carabidae). Scandinavian Journal of Forest Research 22:231-240.

R Development Core Team (2009). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Ribera, I., D.I. McCracken, G.N. Foster, I.S. Downie, & V.J. Abernethy. 1999. Morphological diversity of ground beetles (Coleoptera: Carabidae) in Scottish agricultural land. Journal of Zoology, London 247:1-18.

Ribera, I., S. Dolédec, I.S. Downie, & G.N. Foster. 2001. Effect of Land Disturbance and Stress on Species Traits of Ground Beetle Assemblages. Ecology 82:1112-1129.

Stephens, S.S. & M.R. Wagner. 2006. Using ground foraging ant (Hymenoptera: Formicidae) functional groups as bioindicators of forest health in northern Arizona Ponderosa Pine forests. Environmental Entomology 35:937-949.

Thiolouse, J., D. Chessel, S. Dolédec, & J.-M. Olivier. 1997. ADE-4: a multivariate analysis and graphical display software. Statistics and Computing 7:75-83.

Thullier, W., D.M. Richardson, M. Rouget, S. Proches, & J.R.U. Wilson. 2006. Interactions Between Environment, Species Traits, and Human Uses Describe Patterns of Plant Invasions. Ecology 87:1755-1769.

Walker, B. 1995. Conserving Biological Diversity through Ecosystem Resilience. Conservation Biology 9:747-752.

Chapter 5- Conclusions and Recommendations

5.1 Conclusions

Using carabid beetles, the objective of my thesis was to compare the alternative approaches of recognizable taxonomic unit diversity and functional diversity to the traditional approach of taxonomic diversity. Based on the results from the three approaches I found that recognizable taxonomic unit bioindication may yield some similar results to taxonomic bioindication, while functional morphological bioindication may require further refinement or an increased sample size to yield comparable results. In fact, all approaches studied here could undoubtedly yield far more precise results with further refinement; however, the goal of this research was to compare different methodologies using a widely used taxon from a common study, not to refine existing techniques. Interestingly, the most informative response variable was present in both the taxonomic and recognizable taxonomic unit approaches, that being carabid beetle activity-abundance. This could potentially be a valuable finding as carabid beetle activity-abundance only requires the ability to identify a carabid beetle from amongst other beetles.

The taxonomic approach using carabid beetles is well established (Butterfield et al., 1995; Abildsnes & Tømmerås, 2000; Heliölä et al., 2001; Koivula, 2002; Allegro & Sciaky 2003; Rainio & Niemelä, 2003; Lemieux & Lindgren, 2004; Moore et al., 2004; Klimaszewski et al., 2005; Work et al., 2008). It is generally accepted as effective at detecting shifts in community from one ecosystem to another whether those shifts be due to alteration, destruction, or recovery of habitat, and my study confirmed this to be true

even when the overall species diversity is low (Chapter 2). Carabid beetles were easily captured through the use of pitfall traps and their taxonomic diversity and overall activity-abundance yielded significant results between canopy treatment types as expected. However, to achieve this result a considerable amount of time, effort, and training were required. Training and species confirmation had to come from professional entomologists and/or taxonomists in order to ensure accuracy. Indeed, the rationale for seeking a more cost effective, accessible and efficient means of bioindication persists, which is why I examined two additional approaches.

The first alternative approach I examined was the use of recognizable taxonomic units (Oliver & Beattie, 1993) (Chapter 3). This approach yielded somewhat similar results to the taxonomic approach, in part because both approaches used the same activity-abundance data. The recognizable taxonomic unit approach was not able to differentiate treatments by diversity measures, however. The major issues of splitting and lumping (Oliver & Beattie, 1996; Derraik et al., 2002; Krell, 2004) and repeatability (Krell, 2004) were overcome through the use of a key (Todd & Meggs, 2008) and some training of parataxonomists. Because sorting was based on obvious morphological characteristics, each sample took a minimal amount of time and great accuracy could be achieved even by non-entomologists and/or non-taxonomists. This allows for the approach to be accessible to a wide range of investigators and may allow for many bioindication studies to occur that could not have otherwise.

The steps taken to overcome the main criticisms of the recognizable taxonomic approach (e.g. the key and training of parataxonomists) required additional time investment on the part of the investigators. This time investment required substantially less resources than what would have been required for the taxonomic study, as it involved only one-time input from a taxonomist, and a substantial time savings resulted. The cost savings and accessibility advantages combined with comparable results to taxonomic studies may make the recognizable taxonomic unit approach very appealing to researchers. With additional refinement, and in areas of higher species diversity, this approach could yield even higher levels of resolution. That I was able to observe differences in diversity responses even with a limited number of species in the trap data is promising for the future of the recognizable taxonomic unit approach. In an ecosystem where even one or two additional RTU groups could be elucidated a significant result may be achieved. This could lead to increased amounts of bioindication studies and monitoring programs which would be a boon to conservation efforts. The application of the RTU approach will depend upon the level of detail required by the investigator, however. For instance, if rare species identification is required the RTU approach would not be a suitable choice.

In Chapter 4, I examined functional diversity. Most past research on functional diversity for bioindication has focussed on sorting arthropods into functional roles like specialists, generalists, and opportunists (Stephens & Wagner, 2006) or into feeding guilds (Grimbacher et al., 2008). These approaches require substantial life history and biological information. Literature on the species identified in this study was sparse and therefore these approaches were not viable. The approach taken by Ribera et al. (2001) consisted of

taking morphometric measurements of body structures, such as eyes, antennae, and legs that could infer function, such as sensory ability and mobility, and this was judged to be a viable means of examining functional diversity for the purposes of my study.

The RLQ analysis yielded models that were not significantly different from random. Therefore, these results suggest that the approach did not generate additional information compared to the taxonomic approach. Although not significant, the trends presented by the model indicated that one species (*Scaphinotus angusticollis*) was primarily associated with undisturbed forest (i.e., closed canopy) and another species (*Pterostichus lama*) was primarily associated with the more disturbed forest canopy (i.e., semi-open and open canopies). The habitat descriptions in the literature of where these species tend to be found corroborates my findings despite their non-significance, i.e., *Scaphinotus angusticollis* (undisturbed forest species) is said to be associated with forests and shaded ground, while *Pterostichus lama* (disturbed forest canopy species) is said to be associated with decaying wood and can often be found under fallen trees (Larochelle & Larivière, 2003). The short sampling period and low diversity of carabid beetles were likely major contributors to the non-significance of the RLQ analysis. With a longer sampling period and higher diversity there is a possibility these species could lead to a significant model that could allow for distinguishing between closed canopy, gap canopy, and semi-open/open canopies. The observation of shifts in functional traits in response to habitat disturbance were also observed in carabid beetles and bees by Ribera et al. (2001) and Moretti et al. (2009), respectively. With successful studies such as these it is conceivable that with further refinement this approach could yield significant results. However,

Moretti et al. (2009) cautioned that the shifts in function that they observed in bee communities' responses to perturbation differed depending upon environmental factors that were independent of disturbance as well. It is therefore reasonable to assume that with additional habitat data, both dependent and independent of disturbance, and a higher diversity of species that compose a greater percentage of the total catch, a different result may be obtained from future studies.

With additional habitat variables, such as soil moisture, coarse woody debris presence/absence, and other variables that may affect carabid distribution included in future models, the functional approach may provide more in-depth information on how the species are affected based on their morphological traits. Additionally, an increase in species diversity would likely allow for a finer resolution to be observed that could distinguish between different harvest treatments. However, this approach required time and effort that was in addition to the taxonomic study. Whether or not additional time and effort is warranted ultimately depends on the goals of the research or monitoring.

5.2 Future Research

Future research in this area should focus on further evaluating alternative methods of bioindication. In particular, an inquiry into the accuracy of lightly trained parataxonomists using a recognizable taxonomic unit key should be conducted. In the case of this study, individuals were trained and had full access to the recognizable taxonomic key. They were also supervised, however, and in the rare instance of a misidentification they were corrected. Previous studies have examined the issue of errors

related to recognizable taxonomic units (Oliver & Beattie, 1993; Oliver & Beattie, 1996; Derraik et al., 2002), and it would be of interest to see to what extent accuracy improves with the use of a key.

Additional investigation at the Robert's Creek Study Forest is warranted. The issues of limited sample size and species diversity may be rectified with an increased sampling period. A sampling protocol that would represent species assemblages from spring to fall may prove to provide a definitive answer in terms of the utility of alternative bioindication approaches in the Robert's Creek Study Forest.

The RLQ analysis deserves further investigation as well. The RLQ analysis is capable of analyzing both quantitative and qualitative species variables and relating them to environmental variables (Doledec et al., 1996; Ribera et al., 2001). Unfortunately, there was a lack of life history information that prevented me from using qualitative variables. Such variables are of great interest because they may be more adept at differentiating habitat requirements that are affected by disturbance. For example, Ribera et al. (2001) investigated such qualitative variables as wing development, food of the adult, overwintering stage, daily activity, period of emergence, and seasonal period of activity. For further exploration in this area it would be necessary to undertake life history studies of the prominent carabid beetle species in the Robert's Creek Study Forest.

The RLQ analysis is also capable of incorporating far more sophisticated habitat variables than were used in the model in my research. Ribera et al. (2001) included such

variables as soil moisture, plant density, soil litter, canopy height, and soil pH amongst others. Additional data have been collected for the samples of this study by the Coast Region Experimental Arthropod Project so there is an opportunity to further refine the model. Inclusion of these additional variables will allow one to identify the components of treatment type that represent habitat attributes important to each carabid species. There is the potential that this would allow for further refinement of harvesting treatments that could minimize disturbance while maximizing harvest volumes.

Finally, the recognizable taxonomic unit study needs to be repeated in a variety of geographic areas and compared to parallel taxonomic studies. My study yielded very few carabid species by comparison to other studies (Chapter 2), and therefore the results from other study areas with a greater diversity of carabid beetle species could yield substantially different results. In the event that many species in a different area were grouped into significantly fewer recognizable taxonomic units, it is not unreasonable to assume that the results would be quite different from those reported here. My study included several morphologically distinct carabid beetle groupings, and these undoubtedly could lead to the necessary amount of diversity to differentiate between treatment groups. However, in a system that contained a high number of taxonomically similar species the results would likely favour the taxonomic approach as those species would simply all be grouped together in a recognizable taxonomic unit.

Literature Cited

Abildsnes, J. & B.Å. Tømmerås. 2000. Impacts of experimental habitat fragmentation on ground beetles (Coleoptera, Carabidae) in a boreal spruce forest. Annales Zoologici Fennici 37:201-212.

Allegro, G. & R. Sciaky. 2003. Assessing the potential role of ground beetles (Coleoptera, Carabidae) as biondicators in poplar stands, with a newly proposed ecological index (FAI). Forest Ecology and Management 175:275-284.

Butterfield, J., M.L. Luff, M. Baines, & M.D. Eyre. 1995. Carabid beetle communities as indicators of conservation potential in upland forests. Forest Ecology and Management 79:63-77.

Derraik, J.G.B., G.P. Closs, K.J.M. Dickinson, P. Sirvid, B.I.P. Barratt, & B.H. Patrick. 2002. Arthropod morphospecies versus taxonomic species: a case study with Araneae, Coleoptera, and Lepidoptera. Conservation Biology 16:1015-1023.

Dolédec, S., D. Chessel, C.J.F. Ter Braak, & S. Champely. 1996. Matching species traits to environmental variables: a new three table ordination method. Environmental and Ecological Statistics 3:143-166.

Grimbacher, P.S., C.P. Catterall, & R.L. Kitching. 2008. Detecting the effects of environmental change above the species level with beetles in a fragmented tropical rainforest landscape. Ecological Entomology 33:66-79.

Heliölä, J., M. Koivula, & J. Niemelä. 2001. Distribution of Carabid Beetles (Coleoptera, Carabidae) across a Boreal Forest-Clearcut Ecotone. Conservation Biology 15:370-377.

Klimaszewski, J., D.W. Langor, T.T. Work, G. Pelletier, H.E.J. Hammond, & C. Germain. 2005. The effects of patch harvesting and site preparation on ground beetles (Coleoptera, Carabidae) in yellow birch dominated forests of southeastern Quebec. Canadian Journal of Forest Research 35:2616-2628.

Koivula, M. 2002. Boreal carabid-beetle (Coleoptera, Carabidae) assemblages in thinned uneven-aged and clear-cut spruce stands. Annales Zoologici Fennici 39:131-149.

Krell, F. 2004. Parataxonomy vs. taxonomy in biodiversity studies – pitfalls and applicability of 'morphospecies' sorting. Biodiversity and Conservation 13:795-812.

Larochelle, A. & M.-C. Larivière. 2003. A natural history of the ground-beetles (Coleoptera: Carabidae) of America north of Mexico. Sofia, Bulgaria: Pensoft.

Lemieux, J. & B.S. Lindgren. 2004. Ground beetle responses to patch retention harvesting in high elevation forests of British Columbia. Ecography 27:557-566.

Moore, J., Ouimet, R., Houle, D., & Camiré, C. 2004. Effects of two silvicultural practices on ground beetles (Coleoptera, Carabidae) in a northern hardwood forest, Quebec, Canada. Canadian Journal of Forest Research. 34:959-968.

Moretti, M., F. de Bello, S.P.M. Roberts, & S.G. Potts. 2009. Taxonomical vs. functional responses of bee communities to fire in two contrasting climatic regions. Journal of Animal Ecology 78:98-108.

Oliver, I., & A.J. Beattie. 1993. A Possible Method for the Rapid Assessment of Biodiversity. Conservation Biology 7:562-568.

Oliver, I. & A.J. Beattie. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. Ecological Applications 6:594-607.

Rainio, J. & J. Niemelä. 2003. Ground beetles (Coleoptera: Carabida) as bioindicators. Biodiversity and Conservation 12:487-506.

Ribera, I., S. Dolédec, I.S. Downie, & G.N. Foster. 2001. Effect of Land Disturbance and Stress on Species Traits of Ground Beetle Assemblages. Ecology 82:1112-1129.

Stephens, S.S. & M.R. Wagner. 2006. Using ground foraging ant (Hymenoptera: Formicidae) functional groups as bioindicators of forest health in northern Arizona Ponderosa Pine forests. Environmental Entomology 35:937-949.

Todd, M.A. & J.M. Meggs. 2008. Parataxonomic key to the ground arthropods of the Roberts Creek Study Forest, British Columbia. Unpublished document available from M.A. Todd, BC Ministry of Forests and Range, Nanaimo, BC, 23 pp.

Work, T.T., M. Koivula, J. Klimaszewski, D. Langor, J. Spence, J. Sweeny, & C. Hebert. 2008. Evaluation of carabid beetles as indicators of forest change in Canada. Canadian Entomologist 140:393-414.

Appendix A- Table R: Environmental Variables 2007

	Canopy Closure %	Canopy
Site1	84.00	Closed
Site2	82.25	Closed
Site3	83.75	Closed
Site4	70.50	Semi-open
Site5	75.00	Semi-open
Site6	73.25	Semi-open
Site7	73.25	Gap
Site8	65.75	Gap
Site9	77.50	Gap
Site10	20.75	Open
Site11	3.75	Open
Site17	53.50	Open
Site18	75.50	Open
Site19	67.50	Open
Site20	79.75	Gap
Site21	78.00	Gap
Site22	76.75	Gap
Site23	75.25	Closed
Site24	66.75	Closed
Site25	88.75	Closed
Site50	41	Semi-open
Site51	42	Semi-open
Site52	58.5	Semi-open

Appendix B- Table L: Species Activity-abundance Data 2007

	Ctuberculatus	Odejeani	Palgidus	Pherculaneus	Plama	Sangusticollis
Site1	0	0	4	62	1	35
Site2	9	0	21	72	3	33
Site3	0	0	14	39	0	4
Site4	1	0	0	11	0	0
Site5	0	3	0	16	4	0
Site6	1	4	4	11	2	0
Site7	1	0	31	26	0	3
Site8	1	9	33	15	0	1
Site9	0	4	29	115	0	1
Site10	2	0	0	0	0	0
Site11	11	0	0	1	0	0
Site17	0	0	0	6	2	0
Site18	3	0	2	11	0	0
Site19	0	4	0	5	6	0
Site20	1	7	64	62	0	3
Site21	1	0	7	75	0	11
Site22	2	3	33	76	0	11
Site23	0	12	25	29	1	1
Site24	3	1	72	36	0	17
Site25	3	2	29	65	0	13
Site50	0	0	1	13	0	0
Site51	1	0	0	17	0	0
Site52	3	0	0	10	0	0

Appendix C- Table R: Environmental Variables 2008

	Canopy Closure %	Canopy
Site1	84.0	Closed
Site2	82.25	Closed
Site3	83.75	Closed
Site4	70.50	Semi-open
Site5	75.00	Semi-open
Site6	73.25	Semi-open
Site7	73.25	Gap
Site8	65.75	Gap
Site9	77.50	Gap
Site10	20.75	Open
Site11	3.75	Open
Site12	16.75	Open
Site17	53.50	Open
Site18	75.50	Open
Site19	67.50	Open
Site20	79.75	Gap
Site21	78.00	Gap
Site22	76.75	Gap
Site23	75.25	Closed
Site24	66.75	Closed
Site25	88.75	Closed
Site50	41.00	Semi-open
Site51	42.00	Semi-open
Site52	58.50	Semi-open

Appendix D- Table L: Species Activity-abundance Data 2008

	Ctuberculatus	Odejeani	Palgidus	Pherculaneus	Plama	Sangusticollis
Site1	1	2	0	54	0	70
Site2	1	1	8	32	0	18
Site3	1	0	20	30	0	17
Site4	0	1	0	4	0	0
Site5	0	15	0	10	1	0
Site6	1	16	0	8	5	0
Site7	3	3	32	40	0	11
Site8	0	2	7	27	1	1
Site9	0	6	97	25	0	3
Site10	5	0	1	7	0	0
Site11	0	0	1	3	0	0
Site12	4	0	0	7	1	0
Site17	0	1	0	2	1	0
Site18	2	6	0	13	2	0
Site19	0	11	0	4	5	0
Site20	1	16	44	32	0	19
Site21	0	8	10	5	0	12
Site22	3	7	24	17	0	5
Site23	3	11	47	31	0	8
Site24	1	0	20	2	0	17
Site25	4	2	5	14	1	5
Site50	1	7	0	11	1	1
Site51	0	3	0	16	0	0
Site52	3	3	2	16	0	0

Appendix E- Table Q: Morphological Trait Measurements

	LYW	LAL	LPW	LPH	LEW	LFL	LTR	LRL	LWF	LTL
Ctuberculatus	-0.02	-0.07	-0.05	0.03	0.03	0.00	-0.06	-0.01	-0.03	2.24
Odejeani	0.11	0.11	0.08	0.07	0.03	0.01	-0.07	0.12	-0.01	2.15
Plama	-0.04	-0.05	0.06	-0.02	-0.05	-0.07	0.14	-0.07	0.10	2.28
Palgidus	-0.02	-0.03	-0.01	0.02	-0.01	-0.03	0.00	-0.02	0.02	1.98
Pherculaneus	-0.04	-0.06	0.03	-0.04	-0.02	-0.03	0.11	-0.06	0.06	2.06
Sangusticollis	0.00	0.10	-0.12	-0.06	0.02	0.11	-0.13	0.05	-0.13	2.17