Conservation Biogeography of South African Dragonflies (Odonata)

by

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Declaration

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the owner of the copyright thereof (unless to the extent explicitly otherwise stated) and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

Date: 11 November 2008

Abstract

The great pressures on freshwaters require their conservationists and managers to develop methods to rapidly and accurately assess their condition. Dragonflies are excellent indicators of habitat integrity and are effective organisms for this purpose. However, assessment must be done at the correct spatial scale. My aim here is to optimize the spatial resolution at which species are mapped, using three different concepts and methods in freshwater invertebrate distribution mapping, with special emphasis on IUCN Red Listing. The first is the extent of occurrence (EOO) concept, using the minimum convex polygon, and the second, the area of occupancy (AOO) concept, using IUCN and quaternary catchments. The third approach uses a river layer to compare the suitability of grids as opposed to catchments in mapping.

In this study I found that area estimation based on minimum convex polygons should not be encouraged for aquatic organisms. This study also suggests that the IUCN concept of area of occupancy (AOO) should be redefined simply as occurrence, referring to known point-locality presences only and, if future data allow, to known absences. The IUCN extent of occurrence (EOO), for aquatic species, should be defined as 'the sum of the smallest hydrological units identified of presently known, inferred or projected occurrences of a taxon, excluding cases of vagrancy, that are used to estimate the threat to a taxon'. A single hydrological unit is also the conservation or management unit. Currently, that unit is the quaternary catchment.

Dragonflies have excellent potential as indicators of habitat integrity. For this purpose, my aim was to develop the Dragonfly Biotic Index (DBI) for South Africa and compare the DBI to another index, the Average Taxonomic Distinctness Index (AvTD), which was believed to have potential in assessments. The DBI and AvTD are correlated, which suggests that they could be used on a complementary basis to prioritize sites. The DBI is a low-cost, easy-to-use method and is already used for measuring habitat recovery. It has great potential for environmental assessment and monitoring freshwater biodiversity, especially as a complement to freshwater quality assessments that use macroinvertebrate scores. I thus recommend its integration into freshwater management and conservation schemes.

Opsomming

Groot druk op varswater bronne vereis hul bewaarders en bestuurders om metodes te ontwikkel vir die vinnige en akkurate meting van hul kondisie. Naaldekokers is uitstekende indikators van habitat integriteit en is organismes wat effektief gebruik kan word vir hierdie doel. Nie teenstaande, moet analieses op die korrekte ruimtelike skaal geskied. My doel is om die ruimtelike resolusie waarop spesies gekarteer word te optimaliseer, deur gebruik te maak van drie verskillende konsepte en metodes van varswater invertebrata verspreidingskartering, met spesiale klem op IUCN Rooilysing. Die eerste is die omvang van voorkoms (OVV) konsep wat gebruik maak van die minimum konveks poligoon, die tweede is die area van okkupasie (AVO) konsep, wat gebruik maak van IUCN en kwaternêre opvangsgebiede. Die derde benadering gebruik 'n rivier laag om te verlyk tussen die geskiktheid van roosters teenoor water opvangsgebiede in kartering.

In hierdie studie het ek gevind dat area skatting wat gebaseer is op minimum konvekse poligone nie aangemoedig behoort te word vir akwatiese organismes nie. Die studie stel ook voor dat die IUCN konsep van area van okkupasie (AVO) hergedefinieer behoort te word eenvoudig na voorkoms, met verwyssing na bekende punt-lokaliteit teenwoordigheid alleenlik, en, as toekomstige data dit toelaat, na bekende afwesigheid. Die IUCN omvang van voorkoms (OVV), vir akwatiese spesies, behoort gedefinieer te word as 'die som van die kleinste hidrologiese eenheid geidentifiseer, van huidiglik bekende, afgeleide of geprojekteerde voorkoms van 'n takson, uitgesluit gevalle van rondswerwing, wat gebruik is om die bedreiging van die takson te beraam'. 'n Enkele hidrologiese eenheid is ook die bewarings- of bestuurseenheid. Huidiglik is hierdie eenheid 'n kwaternêre opvangsgebied.

Naaldekokers het uitstekende potensiaal as indikators van habitat integriteit. Om hierdie rede, was my doel om die Naaldekoker Biotiese Indeks (NBI) vir Suid-Afrika te ontwikkel en om die NBI met 'n ander indeks, die Gemiddelde Taksonomiese Beslistheidsindeks (GemTB), wat glo potensiaal het vir analieses, te vergelyk. Die NBI en GemTB korreleer wat suggereer dat hulle komplimentêr gebruik kan word met die doel om liggings te prioteriseer. Die NBI is 'n lae koste, maklik gebruikbare metode en word al reeds gebruik om metings te doen van habitat herstel. Dit het baie potensiaal vir omgewingsassessering en monitering wat gebruik maak van makro-invertebrata tellings. Daarom stel ek voor dat dit met varswater bestuurs- en bewaringsplanne geintegreer word.

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Adam Johnson helped me far beyond the call of duty. We spent much good time in the field and on odd jobs, constructing all sorts of contraptions for aquatic work - novel or re-invented, they all worked.

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I would like to thank the following museums and their curators for kindly providing space and material to record their collections: Iziko Museum (Cape Town), Albany Museum (Grahamstown), and National Insect Collection (Pretoria). I would also like to thank the IUCN Odonata SSC workshop participants, particularly Jens Kipping.

Thanks also to my friends and family who have participated in my life and inevitably have influenced this thesis.

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Dedication

I dedicate this thesis to my parents, for their love and support.

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Chapter 1

GENERAL INTRODUCTION

Freshwaters are essential for sustaining human existence (Revenga et al. 2005). Ecosystem services provided by freshwater biodiversity include the provisioning of clean water, food (e.g. rice, fish), and goods to humans (e.g. reeds as building material) and resilience to anthropogenic impacts (e.g. pollution or excessive nutrient release). Other services include the suppression of water-borne diseases, flood attenuation, and delivery of sediment to coastal areas. Additionally, the recreational and spiritual value of wetlands cannot be denied (Millenium Ecosystem Assessment 2005). In terms of monetary value, global values of ecosystem goods (in the form of fishes), ecosystem services, and biodiversity yielded a value of US \$6579 X 10^9/year for all inland waters, more than two thirds of the USA's yearly gross domestic product (Constanza et al. 1997). Yet, despite our dependence on, and the importance of freshwaters, they may be the most threatened ecosystems in the world (Dudgeon et al. 2006).

Among the greatest global threats to the functioning of freshwater ecosystems are the destruction or degradation of habitat, invasion by alien species, overexploitation, water pollution, and flow modification. Superimposed upon these interacting threats are global environmental changes such as nitrogen deposition, temperature warming and shifts in precipitation and runoff patterns. Declines in biodiversity are up to five times greater in some freshwaters than in the most affected terrestrial ecosystems (Dudgeon et al. 2006, Sala et al. 2000). However, extinction rates of freshwater biodiversity are rarely monitored or are biased in terms of geography, habitat or taxonomy; global estimates remain to be made.

In a regional context, a first national conservation assessment of main river ecosystems in South Africa found that 84% of the ecosystems are threatened, with 54% critically endangered, 18% endangered, and 12% vulnerable (Nel et al. 2007). These findings highlight the need to systematically protect South Africa's freshwater biodiversity. In this study, we use dragonflies (Odonata) as flagships for the conservation of South Africa's inland waters.

Dragonflies are a well-studied group of invertebrates (Córdoba-Aguilar 2008), with their increasing recognition in conservation worldwide (Samways 2008). In a regional context, this fact is reflected in dragonflies being the only insect group that are currently being globally assessed by the World Conservation Union (IUCN). For example, in an African context, dragonflies have been the subject of a regional southern African freshwater assessment (Suhling et al. 2008), and have been assessed in South Africa for Red List status (Samways 2006). This knowledge base is continually expanded and refined, with recent discoveries of new species (Dijkstra et al. 2007), and re-discoveries of species (Samways and Tarboton 2006), as well as numerous range-extensions. The advantages of using dragonflies as bioindicators are well documented (Schmidt 1985, Chovanec & Waringer 2007) as has their use in biodiversity assessments (Samways 2008) and their potential value as service providers (Simaika and Samways 2008).

In a South African context, dragonflies are highly vagile, generalist predators and thus tend to show lower levels of endemism than many other insect taxa and little dependency on the composition of plant communities (Grant and Samways 2007). Nevertheless, threatened dragonflies occur mostly in the mountainous regions of the Cape Floristic Region, considered a centre of endemism for the group (Samways 1992, Grant and Samways 2007).

Of the 34 endemic dragonfly species in South Africa, 12 are globally Red Listed species (Samways 2006). Threats to these globally Red Listed species appear to come mainly from riverine invasive alien trees, which have dense canopies that shade out the habitat (Samways and Taylor 2004). Many of the species on the national, but also global Red List are further affected by a synergy of threats. Multiple impacts include habitat loss by urbanization, habitat disturbance by cattle trampling, possible predation by trout, detergent pollution, mine effluent and agricultural run-off, and over-abstraction of water (Samways and Taylor 2004, Samways 2004).

In light of these synergistic threats to South Africa’s freshwater biota, particularly dragonflies, it is the aim of this study to map the distribution and status of species at increasingly detailed levels of understanding (Chapters 3-4).

In Chapter 3, I optimize the spatial resolution at which species are mapped, using three different concepts and methods in freshwater invertebrate distribution mapping, with special emphasis on IUCN Red Listing. The first is the extent of occurrence (EOO) concept, using the minimum convex polygon, and the second, the area of occupancy (AOO) concept, using IUCN and quaternary catchments. The third approach uses a river layer to compare the suitability of grids as opposed to catchments in mapping.

In Chapter 4, I develop a biodiversity index for measuring ecological integrity, the Dragonfly Biotic Index (DBI), and compare this to the Average Taxonomic Distinctness index (AvTD). Furthermore, I investigate the use and value of the indices for prioritizing sites for protection and discuss their complementarity to freshwater quality indices that are based on macroinvertebrate scores.

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Chapter 2
Notes on database development

Biogeographic information from South Africa (including Lesotho and Swaziland) was used here. This area is unique in that such information is not only available to potential users worldwide via the internet (SANBI 2008), but that many taxa, including the Odonata, are well sampled. A spatial-relational database was constructed from records of odonatolgical collections and sightings. The database consists of a merger between Samways' database of collections and sightings (1988 to 2005) and an early version of a database of Pinhey's (1984, 1985) publications. The former database consisted of 5046 species records and 365 sampling locations, the latter of 1692 species records and 708 sampling locations. The merged database consisted of 6738 species records and 722 sampling locations. The database that was constructed from Pinhey's records contained a high number of omissions (absences from locations where species were recorded) and commissions (presences at locations where species were not recorded). Furthermore, many of the 722 localities were either identical or had the incorrect co-ordinates.

In the construction of the database, gazetted names were used wherever possible, replacing nicknames given for the localities by collectors. Nicknames are stored as aliases. The database was extended to include additional information on localities, including province name, alias and the source of the new locality, whether a collection or publication. Also, the species input was extended to include the actual date of collection, then, separately the month, and year, the collector's/collection's information, the original (collector's) collection number, the behaviour of the specimen when collected/recorded, association with other specimens in the collection/at the time of observation, whether or not the specimen was collected, and if so, whether it is kept dry or in alcohol. Finally, for future purposes, the habitat information input was extended. Database input extensions were made for the purpose of not only having a spatial-relational database of records, but also of keeping track of field efforts and the Stellenbosch University Museum Entomological Collection of Odonata, as well as records of other museum or personal collections, and publications.

The database was temporarily expanded to include all Odonata records of southern African species for the IUCN Freshwater Assessment of Southern Africa (Suhling et al. 2008). This allowed for testing the database for accuracy. Species maps were produced for all 162 taxa recorded in South Africa and their distributions were checked twice. For the purposes of Red Listing and keeping up-to-date with current taxonomy, species scientific names and authors were updated and IUCN Red List status recorded. From the first production, four maps required additions, seven maps needed deletions and thirteen maps changes. In the second round, records were added to six maps, deleted from four, and changed in seven. A third set of maps of South Africa's 162 taxa was produced for the fieldguide to the *Damselflies and Dragonflies of South Africa* (Samways 2008). This resulted in 60 species record changes and 66 deletions. Forty-nine erroneous localities were deleted from the database.

During this time, additional records were entered into the database from insect collections housed at the Iziko Museum* (Cape Town), Albany Museum* (Grahamstown), Northern Flagship Institution (Pretoria), National Museum (Bloemfontein) and National Insect Collection* (Pretoria). Museum visits (*) included verification of old records and identification of new specimens accessioned since Pinhey's 1984 and 1985 publications. This represents an addition of 2582 verified specimen records to the database. Additional records came from new collection effort, with special emphasis on endemic species sampling during the field seasons from 2005 to 2008 in the Western and Eastern Cape. In the Western Cape 26 locations were visited and in the Eastern Cape 25. These new sampling locations resulted in 625 new species records, extending the known geographical range of the endemic Red Listed *Ecchlorolestes peringueyi*, *E. nylephtha*, and *Syncordulia venator* (Simaika, unpublished data). First descriptions of the *Ecchlorolestes* larvae are under way (Simaika and Samways 2008a). Field collection effort has also resulted in the discoveries of two new species *S. legator,* and *S. serendipator* (Dijkstra et al. 2007) and a review of their phylogeny (Ware et al. 2008). Field effort also yielded additional specimens for a first description of *Metacnemis angusta* larvae (Simaika and Samways 2008b). The new database now consists of 9945 species records from 897 locations. From the resultant database, species distribution maps were constructed using both ArcView GIS 3.2a and ArcGIS 9.2 (Environmental Survey Research Institute 1999, 2006).

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Chapter 3

Large-scale estimators of threatened freshwater species relative to practical conservation management*

Abstract: *Freshwater ecosystems are among the most threatened in the world. It is therefore essential to map the distribution and status of species to ascertain their threat status for prioritizing conservation action. However, while there is agreement that the conservation of freshwater ecosystems depends on whole-catchment management, there are still a wide variety of large-scale mapping methods in use, the advantages and disadvantages of which have not been fully appreciated. I aim to optimize the spatial resolution at which species are mapped, using three different concepts and methods, with special emphasis on IUCN Red Listing. The first is extent of occurrence (EOO), using the minimum convex polygon, and the second, area of occupancy (AOO), using IUCN and quaternary catchments. The third approach uses rivers to compare the suitability of grids as opposed to catchments in mapping. This study shows that area estimation based on minimum convex polygons should not be encouraged for aquatic organisms. Rather I recommend that inferred distributions best be based on predictive modeling. The IUCN definition of AOO is a useful term, albeit highly scale-dependent, for casual assessment of the total approximate area over which a species occurs. However, for this aquatic fauna, assessment of occurrence should be based on point-locality presences only. The EOO, for freshwater aquatic species, should be redefined as 'the sum of the smallest hydrological units identified, of presently known, inferred or projected occurrences of a taxon, excluding cases of vagrancy, that are used to estimate the threat to a taxon'. A single hydrological unit is also the conservation or management unit. Here this unit is the quaternary catchment. In future, conservation managers and decision makers should facilitate co-operation in freshwater mapping efforts by working at the same spatial scale, i.e. the same hydrological unit.*

Keywords: Red Listing, freshwater, catchments, conservation, biodiversity, assessment, range estimation, Odonata, dragonflies.

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1. Introduction

There is general agreement that the conservation action of freshwater ecosystems depends on whole-catchment management (O'Keeffe 1989, Ward 1998, Saunders et al. 2002). For the recent assessment of freshwater biodiversity in Eastern Africa, the IUCN adopted *the catchment* as the effective conservation unit (Darwall et al. 2005). This helps identify areas of conservation concern. However, while distributions were mapped at a Level 6 resolution (not presented in the report) of the Hydro 1K system (USGS 2006), in the assessment, the authors used the very large, Level 3 catchments for analysis. According to the authors, the 'Level 3 resolution was employed in the analysis … to represent an appropriate scale for application to river basin management' (Darwall et al. 2005). However, Level 3 catchments may cover thousands of square kilometers of terrain. Furthermore, the authors compensate for the use of these large catchments by overlaying different-sized grids, depending on the resolution of the species distribution data. For example, in the recent IUCN report on freshwater biodiversity in Eastern Africa, fish, mollusc, crab and dragonfly species distributions were mapped to the boundaries of the river basins in which they were recorded (Darwall et al. 2005). With this method, Darwall et al. (2005) maintain that 'unless finer spatial detail is provided, each species must be assumed to have basin-wide distribution'. However, in the case of the crabs, although point-locality data were available, the authors chose to infer species distributions by using a two-degree grid instead. This, Darwall et al. (2005) state, better reflects the suspected wider distributions of the species.

A recent conservation assessment of freshwater crabs of southern Africa makes use of the IUCN extent of occurrence (EOO) concept (Cumberlidge and Daniels 2008). Extent of occurrence is defined in the IUCN (2006) guidelines as 'the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of present occurrence of a taxon, excluding cases of vagrancy'. Furthermore, the IUCN guidelines (IUCN 2006) recommend using the much-criticized but popular range-estimation method known as the minimum convex polygon (MCP) to map EOO (Burgman and Fox 2003). It is perhaps, understandable then that given these general guidelines and suggested methods, Cumberlidge and Daniels (2008) confuse EOO as a way to 'estimate geographic range', rather than degree of threat to species. Another recent paper by Jetz et al. (2008) on the conservation implications of overestimating species ranges severely criticizes EOO ‘range maps’. The authors present nearly two decades worth of research where the IUCN EOO concept was interpreted as a method to estimate distribution and geographic range. This worrying trend is indicative that the IUCN definition of EOO needs serious revision.

However, according to the IUCN guidelines, the intent is to use EOO as ‘a parameter that measures the spatial spread of the areas currently occupied by a taxon’, and to … ‘measure the degree from which risks to threatening factors are spread spatially across a taxon’s geographical distribution; and … [Is] not … an estimate of the amount of occupied or potential habitat, or a general measure of a taxon’s range’.

To estimate EOO, the IUCN recommends using the MCP method, a type of range estimation method that severely over-estimates the potential range of species, as it does not take into account potentially vast uninhabitable areas for a given species. New, more recent alternatives include the alpha convex hull, kernel method, and most recently, the local convex hull to generate species home ranges (Getz and Wilmers 2004). These methods are based on the same concept as the MCP and, although they are improvements, are still problematic. The IUCN (2006) guidelines recognize that the use of these estimators is problematic, but nevertheless recommends their use. I aim to show here, that for the purpose of freshwater conservation, home-range generating methods are not suitable to estimate EOO. For freshwaters, the catchment has already been identified as the logical principal conservation and management unit and I aim to stress here that its use in any analyses using freshwater taxa is more appropriate.

A recent national conservation assessment of main river ecosystems in South Africa found that 84% of the ecosystems are threatened, with 54% critically endangered, 18% endangered, and 12% vulnerable (Nel et al. 2006). These conditions put South African freshwater invertebrates at risk. An assessment by Samways (2006) of the conservation status of the South African dragonfly taxa, using current IUCN Categories and Criteria, showed that of the 34 endemic dragonfly taxa, 12 are globally threatened species. These threatened species occur mostly in the mountains of the Western Cape, a region considered to be a centre of endemism for Odonata (Samways 1992), as well as for a range of other fauna (e.g. Siegfried and Brown 1992, Lombard 1995). Threats to these globally Red Listed species appear to come mainly from invasive alien riparian trees, especially wattle (*Acacia* spp.), which have dense canopies that shade out the habitat (Samways and Taylor 2004).

In the present study, three different concepts and methods in freshwater invertebrate species analysis are explored to optimize spatial resolution for species analysis, with special emphasis on IUCN Red Listing. The first is the extent of occurrence (EOO) concept, using the minimum convex polygon and the second is the area of occupancy (AOO) concept using IUCN and quaternary catchments. The third approach uses a river layer to compare the suitability of grids as opposed to catchments in analysis. I aim to recommend a standardized method for mapping and analyzing specifically freshwater invertebrate species data, in line with IUCN guidelines and criteria.

2. Methods

Biogeographic information from South Africa (including Lesotho and Swaziland) was used here. This area is unique in that such information is not only available to potential users worldwide via the internet (SANBI 2008) but that many taxa, including the Odonata are well sampled. In this area, 162 Odonata taxa occur and all are included the in subsequent analysis. A spatial-relational database was constructed from records of odonatological collections and sightings. The database consists of a merger between Samways' database of collections and sightings (from 1988 to present) and an early version of a database from Pinhey's (1984, 1985) publications. Additional records came from insect collections housed at the South African Museum (Cape Town), Albany Museum (Grahamstown), Northern Flagship Institution (Pretoria), National Museum (Bloemfontein) and National Insect Collection (Pretoria). Museum visits included verification of old records and identification of new specimens accessioned since 1984. Additional records came from new collection efforts, with special emphasis on endemic species sampling, during the field seasons from 2005-8 in the Western and Eastern Cape. In the Western Cape 26 locations were visited, and in the Eastern Cape 25. These new sampling locations have resulted in 625 new species records. These new records extend the known geographical range of the endemic Red Listed *Ecchlorolestes peringueyi* and *E. nylephtha* (Simaika and Samways 2008), and discoveries of the two new species *Syncordulia legator* and *S.*

serendipator (Dijkstra et al. 2007). From the resultant database, species distribution maps were constructed using both ArcView GIS 3.2a and ArcGIS 9.2 (Environmental Survey Research Institute 1999, 2006).

2.1 Extent of occurrence (EOO)

The EOO, using the IUCN (2006) Categories and Criteria guidelines, is defined as the area contained within the shortest continuous boundary which can be drawn to encompass all the known, inferred, or projected sites of the present occurrence of a taxon. The EOO of all 162 taxa of South African Odonata was analyzed. Five categories of distribution were considered to represent a range of species distribution patterns of South African Odonata. For this paper a representative random sample of ten species was chosen. The categories and species are: (a) Cape or near-Cape endemics (*Ecchlorolestes nylephtha*, *E. peringueyi*, and *Syncordulia gracilis*); (b) localized species (*Chlorolestes apricans* and *C. draconicus*); (c) Afro-tropical species (*Agriocnemis pinheyi, Mesocnemis singularis*, and *Orthetrum machadoi*); (d) eastern Great Escarpment species (*Pseudagrion caffrum*); and, (e) widespread South African species (*Aeshna subpupillata*).

The IUCN uses both present and historical distributions (records before 1986) to monitor species population changes. Here, minimum convex polygons (MCPs) were constructed from point-locality distributions from both time periods using the Hawth's tools script (Beyer 2004) available for ArcGIS 9.2. The total areas (in km^2) of the MCPs were calculated. For the purpose of area calculation, I used a conical projection for all shapefiles. Analysis of a species' EOO using MCP is demonstrated for the ten randomly chosen species representative of the five distribution categories.

2.2 Area of occupancy

The IUCN Categories and Criteria define the area of occupancy (AOO) as the area within its EOO which is occupied by a taxon, excluding cases of vagrancy. The measure reflects the fact that a taxon will not usually occur throughout its EOO (IUCN 2006). The IUCN (2006) guidelines are no doubt written mainly for terrestrial taxa, and thus recommend using different-sized grids, depending on the mapping scale (IUCN 2006). However, for analysis of freshwater invertebrates, I used different freshwater catchments. Indeed, the use of grids, at least in biodiversity conservation efforts, was recently criticized by Peterson and Martínez-Meyer (2007), who suggest the use of a point-based method instead.

A first set of distribution maps of 162 taxa that occur in South Africa was made for the IUCN Southern African Freshwater Species Assessment (Suhling et al. 2008) from point-distributions that indicate the species present and historical distributions using the U.S. Geological Survey Hydro1K layer (IUCN map) (USGS 2006). The IUCN maps use very large catchments that represent a species' entire distribution. Level 3 catchments of the Hydro 1k system may cover thousands of square kilometers of terrain (USGS 2006). A second set of present and historical distribution maps was made, using point-distributions, for the field guide *Dragonflies and Damselflies of South Africa* (Samways 2008).

The original IUCN maps were then updated, and quaternary maps constructed, for comparison of different mapping methods. Quaternary catchments were developed by the Department of Water Affairs and Forestry as part of a national hierarchical drainage subdivision system. This system divides drainage regions into successively smaller hydrological units from primary catchments, through to secondary and tertiary catchments and finally to quaternary catchments (Midgley et al. 1994). This system is similar to that used to delineate the US Geological Survey (USGS) hydrological units (Seaber et al. 1987), where quaternary catchments are comparable to the USGS cataloguing units. Both layers, the IUCN and quaternary catchment layers, utilize river catchment boundaries, and are therefore potentially very effective in freshwater conservation management (e.g. Revenga et al. 2000). However, the quaternary catchment layer is much finer in detail, as its catchments are smaller than those of the IUCN layer, and is here considered fine-scale, while the IUCN layer is considered large-scale. The area occupied by a species was calculated from the IUCN and quaternary catchment maps. All shapefiles were in a conical projection.

2.3 Catchment threat status

The main river ecosystem threat status of each catchment in the quaternary layer was calculated using the complementary main stream layer to which Nel et al. (2007) have assigned threat categories largely based on the IUCN threat status categories. The catchment threat status was determined by using a two-step process. First, the catchment layer and main rivers conservation layer were intersected using the 'intersect' geoprocessing function in ArcGIS 9.2. The total lengths of streams per threat category, were calculated per catchment. The resultant stream length value per category was then divided by the total stream length in the catchment. The ratio of the category values was converted into a percentage. Second, the assignment of stream threat categories was made based on the highest percentage contribution (i.e. 51%) of the highest category. In the case where two categories contributed equal percent amounts (50%), the higher category was assigned. The ecosystem threat status assigned is based on, but is not the same as, the IUCN Categories and Criteria (IUCN 2001) used for species. Nel et al. (2007) use their own categories: critically endangered (CE), endangered (E) and vulnerable (V). Nel et al. use the additional category, termed currently not threatened (CNT). In this study, I erect an additional, non-IUCN threat category 'No Category' (NoC), necessary because certain catchments or polygons cannot be categorized into the above categories.

A quarter-degree grid was employed to compare the suitability in species mapping to that of the quaternary catchments layer. This potential of the quarter-degree grid was explored because of its popularity in freshwater species mapping (e.g. Minter et al. 2004). The same method described above was repeated using a quarter-degree grid.

2.4 Statistical Analysis

A Spearman Rank correlation was used in SPSS 13 to determine any significant relationship between the quarter degree grid and catchment threat status layer.

3. Results

3.1 Extent of occurrence (EOO) and area of occupancy (AOO)

The total km^2 area/species of IUCN catchments is, on average, 17% larger than that covered by quaternary catchments. The overall average count of localities per quaternary catchment is 1.25 localities/polygon, while for the IUCN catchment layer, it is 1.90 localities/polygon.

Ten species are compared (Table 3.1, Figure 3.1-3.5). The EOO, using minimum convex polygons (MCPs), can only be calculated for a species with at least three point localities, the minimum require to make a polygon. Due to this limitation, the method could not be applied to the historical distributions of *Agriocnemis pinheyi*, *Chlorolestes draconicus* and *Mesocnemis singularis*. *Aeshna subpupillata* is a widespread species and the MCP method estimates the present EOO of the species at 688,878 km^2, increasing its historic EOO and the site count five-fold. The species' current IUCN area is less than half of the MCP area, but less than one-tenth using quaternary catchments. The area difference between the present IUCN catchments and quaternary catchments is one-tenth. For the Afro-tropical *A. pinheyi*, the difference in current MCP area and IUCN area is even larger, but with a slightly smaller ratio to the quaternary AOO. The difference between the IUCN and quaternary catchments is less than one-tenth. The MCP of the species includes vast areas of where it is not likely to occur, and the IUCN map of *A. pinheyi* shows the southern distribution of the species extending to the coast (Figure 3.1), where it does not actually occur.

Chlorolestes apricans has been in decline with a reduction from nine to four localities. The MCP method does track this, with a considerable decline in area. The IUCN catchment AOO illustrates a more moderate decline and quaternary catchments even more moderate still. The EOO of *C. draconicus* is 175 km^2, less than 1% of the IUCN AOO, and a quarter of the estimated quaternary catchment AOO. As in the case of *A. pinheyi*, the IUCN catchments show the distribution of *C. draconicus* extending to the coast (Figure 3.2), which is not the case in reality. The South-eastern Cape endemic *Ecchlorolestes nylephtha* has declined in the number of localities, from ten to seven, and also in area. This decline is tracked by both the MCP and more so by the quaternary catchments, but not by the IUCN catchments, which actually record an increase in the number of catchments and the total present area. The IUCN catchments approach indicates that this species occurs much farther west, and also farther inland, than illustrated by the other mapping scales (Figure 3.3).

Table 3.1. Example species area of occupancy (AOO) (IUCN and quaternary catchments) and extent of occurrence (EOO) using the minimum convex polygon approach (MCP), around point distributions. Counts are given for point localities (site) and the numbers of different catchments (IUCN and quaternary). Abbreviations: NA = not applicable, NoDet = Not determinanble.

Species and area	**Site**	**AOO**				**EOO**
		IUCN		Quaternary		MCP
	Count	Area (km²)	Count	Area (km²)	Count	Area (km²)
Aeshna subpupillata						
Present area	105	290734	40	29906	79	688878
Historical area	13	51630	10	4543	13	469990
Total area	118	**342364**	**45**	**34449**	**92**	NA
Agriocnemis pinheyi						
Present area	14	77105	8	5443	11	129219
Historical area	2	14531	2	1453	2	NoDet
Total area	16	91637	10	6896	13	NA
Chlorolestes apricans						
Present area	4	12334	3	1161	3	381
Historical area	9	23005	6	1499	7	1836
Total area	13	**35339**	**7**	**2660**	**8**	NA
Chlorolestes draconicus						
Present area	3	38310	3	736	3	175
Historical area	2	6519	1	520	2	NoDet
Total area	5	**44829**	**4**	**1256**	**5**	NA
Ecchlorolestes nylephtha						
Present area	7	20322	4	803	4	3340
Historical area	10	19623	3	2018	10	4370
Total area	17	**39946**	**7**	**2821**	**14**	NA
Ecchlorolestes peringueyi						
Present area	14	24328	4	1259	8	2833
Historical area	5	23115	3	720	5	792
Total area	19	**47443**	**7**	**1978**	**13**	NA
Mesocnemis singularis						
Present area	23	103283	16	16844	20	346160
Historical area	0	0	0	0	0	0
Total area	23	**103283**	**16**	**16844**	**20**	NA
Orthetrum machadoi						
Present area	18	60134	10	9987	14	514037
Historical area	9	44788	8	4464	9	167419
Total area	27	**104921**	**18**	**14451**	**23**	NA
Pseudagrion caffrum						
Present area	12	79972	10	4078	11	57646
Historical area	7	52318	7	2685	7	89175
Total area	19	**132290**	**17**	**6763**	**18**	NA
Syncordulia gracilis						
Present area	6	40675	4	1085	5	43297
Historical area	6	35529	5	1335	6	50076
Total area	12	**76203**	**9**	**2420**	**11**	NA

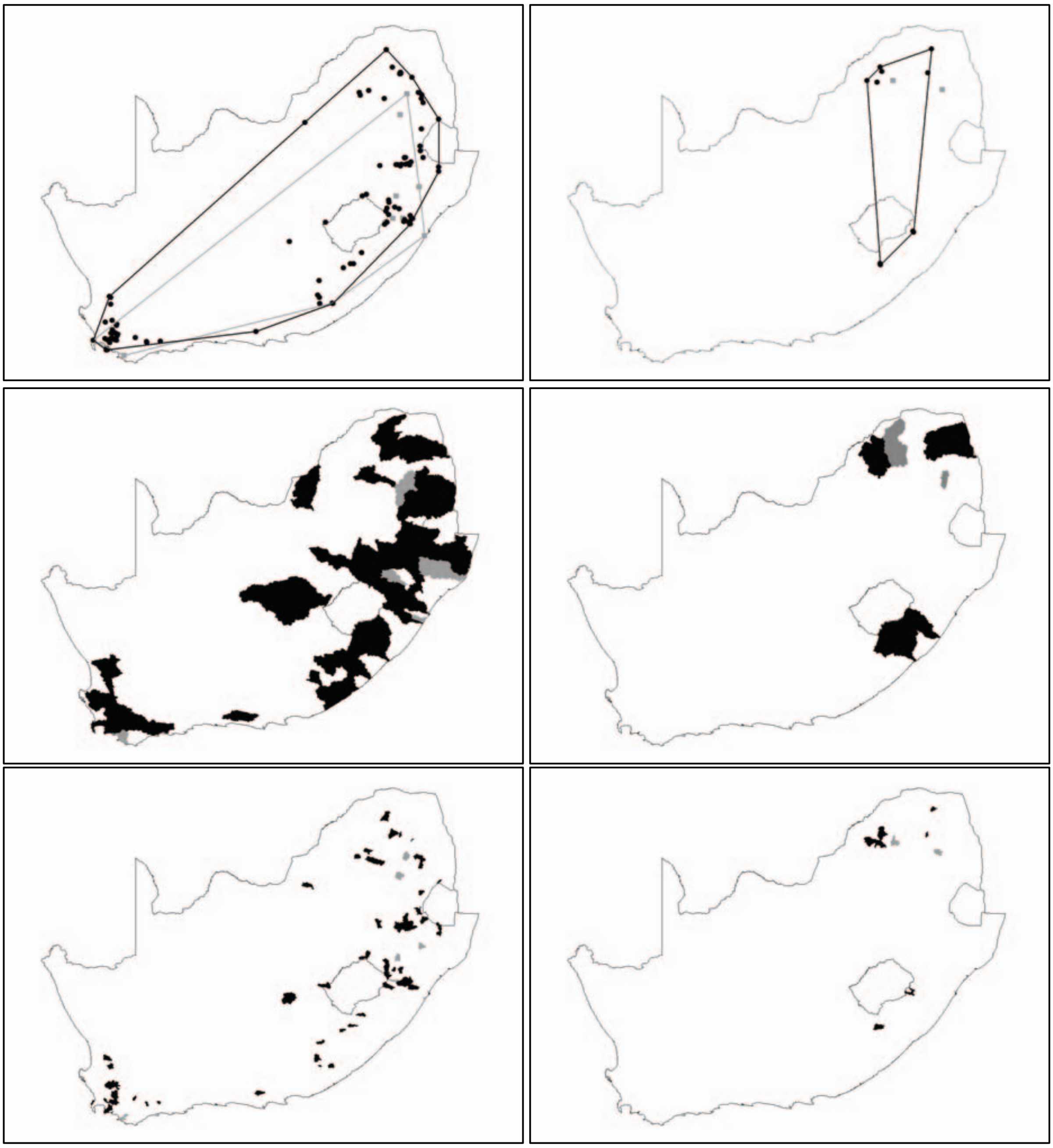

Figure 3.1. Present (black) and historical distributions (before 1986) (gray) of the widespread *Aeshna subpupillata* (left column) and the Afro-tropical, *Agriocnemis pinheyi* (right column) – using three different mapping scales. Top row, minimum convex polygons (black and gray outlines); middle row, IUCN catchments; and, bottom row, quaternary catchments.

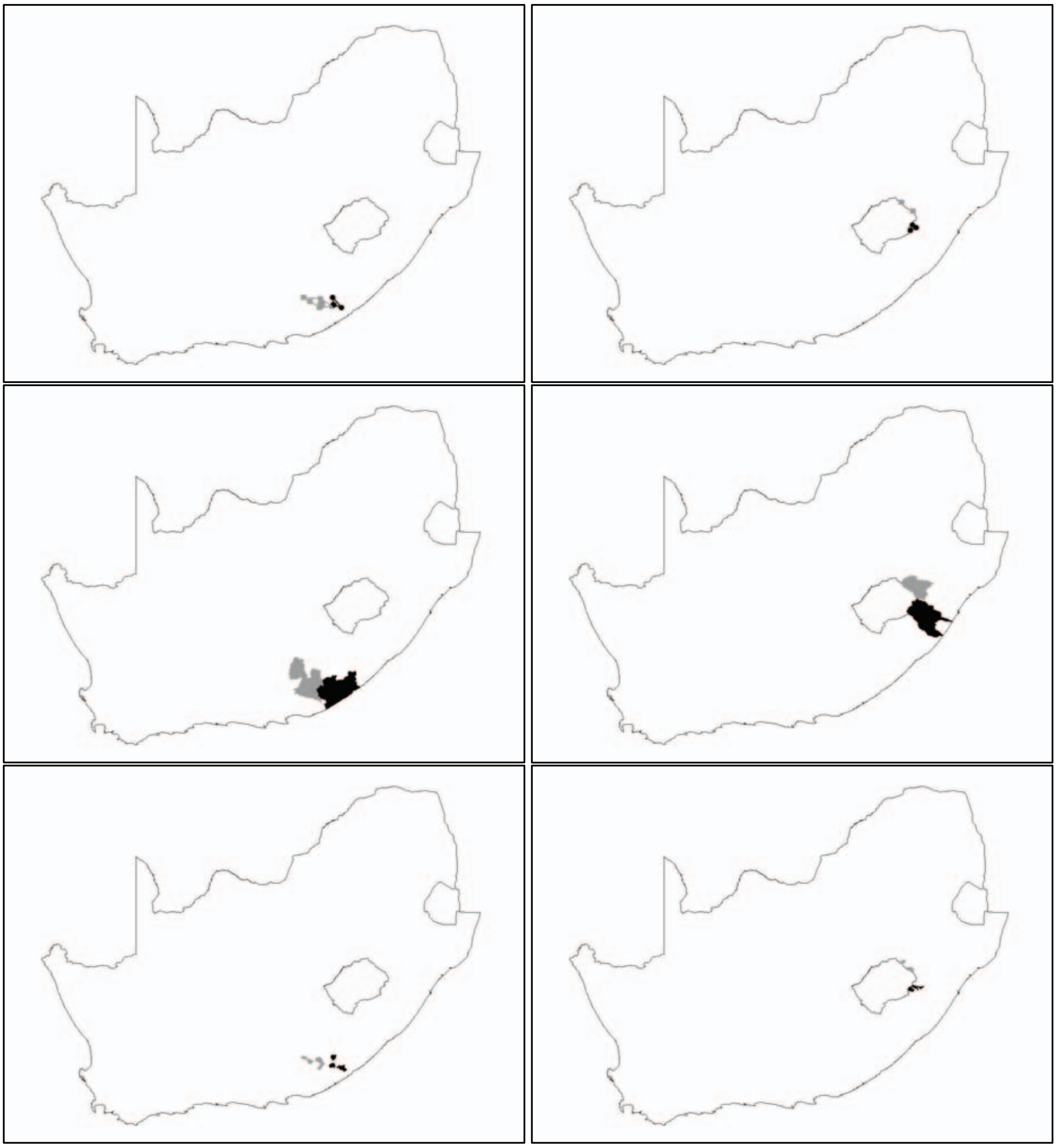

Figure 3.2. Present (black) and historical distributions (before 1986) (gray) of the localized endemic *Chlorolestes apricans* (left column) and *C. draconicus* (right column) – using three different mapping scales. Top row, minimum convex polygons (black and gray outlines); middle row, IUCN catchments; and, bottom row, quaternary catchments.

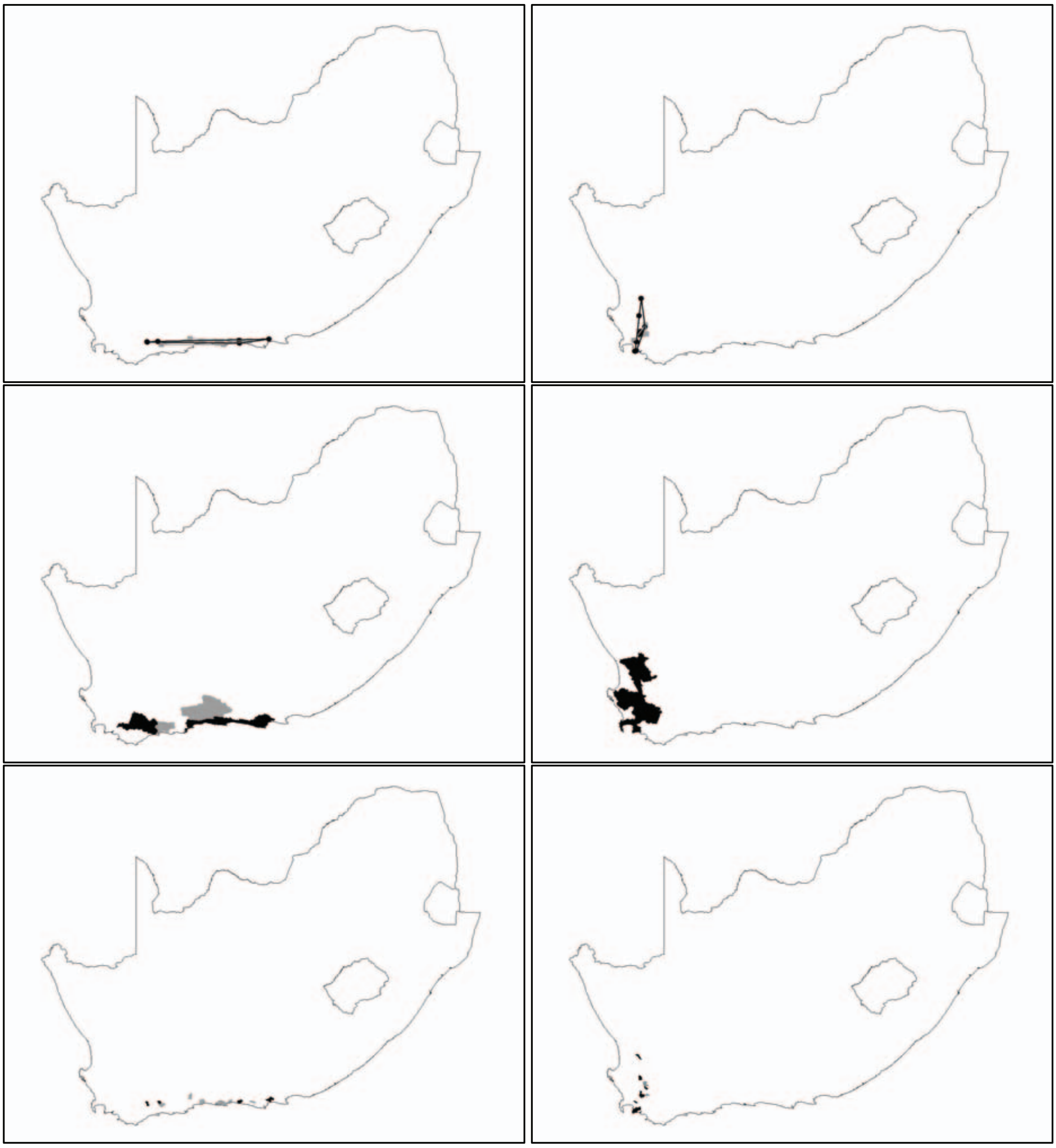

Figure 3.3. Present (black) and historical distributions (before 1986) (gray) of the South-eastern Cape and South-West Cape endemics – *Ecchlorolestes nylephtha* (left column) and *E. peringueyi* (right column) – using three different mapping scales. Top row, minimum convex polygons (black and gray outlines); middle row, IUCN catchments; and, bottom row, quaternary catchments.

The number of known localities of *E. peringueyi* has increased over time. The EOO of the South-West Cape endemic *E. peringueyi* is nine times smaller than the IUCN AOO, but in contrast is almost twice the size of the quaternary AOO. The quaternary catchment area represents less than one-tenth of the present IUCN area. The IUCN map of *E. peringueyi* misleadingly shows the species occuring farther west and north than the other mapping scales (Figure 3.3).

The EOO of *Mesocnemis singularis* is significantly larger than the present IUCN area and even larger than the quaternary catchment area (Figure 3.4). The number of localities from which the Afro-tropical *Orthetrum machadoi* is recorded has doubled. This increase is tracked by the MCP, less so by the quarternary catchments. The IUCN AOO has increased even less. The quaternary AOO represents only 2% of the EOO, but 17% of IUCN AOO. The IUCN area represents 12% of the MCP area.

The number of records of the endemic *Pseudagrion caffrum* has increased from seven to 12. However, the present MCP area is 35% smaller than the historical area, a trend not shared with the IUCN and quaternary catchments. The present IUCN area is 28% larger than the MCP area. The quaternary is 7% smaller than the MCP. The present quaternary catchment area represents 5% of the IUCN area. The IUCN distribution map of *Pseudagrion caffrum* is the only map in which the species distribution extends to the coast (Figure 3.5), although the species does not occur there.

The number of records for *Syncordulia gracilis* has remained at a steady six localities, both in the past and in the present. The MCP area has decreased, and so has the quaternary area. In contrast, the IUCN area has increased by roughly the same ratio as the MCP area. As for many of the previous species, the IUCN distribution map shows *S. gracilis* occuring farther west and north in the South-West Cape, and its eastern distribution extends to the coast (Figure 3.5), where the species is not known to occur.

Figure 3.4. Present (black) and historical distributions (before 1986) (gray) of the Afro-tropical species *Mesocnemis singularis* (left column) and *Orthetrum machadoi* (right column) – using three different mapping scales. Top row, minimum convex polygons (black and gray outlines); middle row, IUCN catchments; and, bottom row, quaternary catchments.

Figure 3.5. Present (black) and historical distributions (before 1986) (gray) of the endemics – *Pseudagrion caffrum* (left column) and *Syncordulia gracilis* (right column) – using three different mapping scales. Top row, minimum convex polygons (black and gray outlines); middle row, IUCN catchments; and, bottom row, quaternary catchments.

3.2 *Catchment threat status*

The comparison of quarter degree (QD) and HCA4 threat status layers used here is a surrogate approach for determining whether QD grids are superior for the analysis of species distributions, as the number of grid cells (2015) is roughly equivalent to the number of HCA4 polygons (1948). The threat status layers are compared in Table 3.2 and in Figure 3.6. A two-tailed Spearman-Rank correlation test found a strong, but weakly significant, correlation between the threat status layers ($r^2 = 0.900$, $n = 5$, $p = 0.037$). Visual comparison shows that a large percentage (8.64%) of the QD grid cells could not be assigned a conservation status (NoC). Furthermore, the QD grid status layer, negatively re-distributes the conservation status levels, causing an under-estimation of each non-IUCN threat category, critically endangered (CE), endangered (E), vulnerable (V), and currently not threatened (CNT) (Table 3.2). Another method of comparison, using species assemblages of the same river catchments, shows that these may be lumped or split into different grid cells. An example is shown in Figure 3.7, where various sampling locations in the Palmiet River catchment of the Kogelberg are split into QD grid cells. Some of these cells also lump other, distinct catchments together, as in the top right of Figure 3.7.

Table 3.2. Threat categories of polygons of the catchments layer and quarter degree grid layer, respectively. Abbreviations of non-IUCN threat categories are as follows: CE (critically endangered), E (endangered), V (Vulnerable) and CNT (currently not threatened). An additional category, NoC (Not Classified) was assigned to polygons that did not contain any threat category information.

	Catchments		Quarter Degree Grids	
Threat Category	Frequency	Percentage (%)	Frequency	Percentage (%)
CE	784	40.25	683	33.90
E	558	28.64	483	23.97
V	442	22.69	247	12.26
CNT	159	8.16	428	21.24
NoC	5	0.26	174	8.64
Total	1948	100.00	2015	100.00

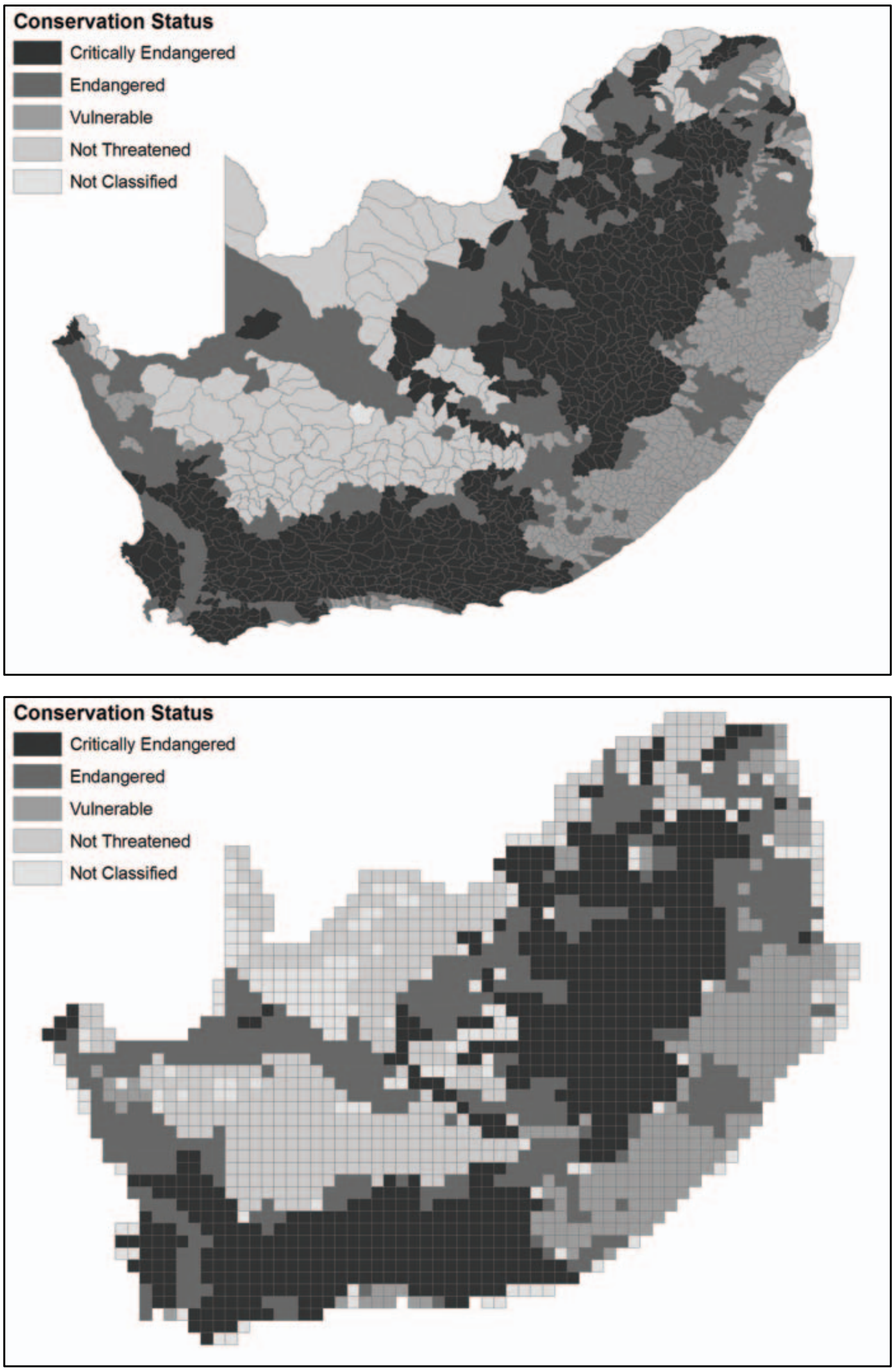

Figure 3.6. Quaternary catchments (top) and quarter degree grid map (bottom) of South Africa, displaying threat status categories (critically endangered, endangered, vulnerable, and currently not threatened). The additional category, not classified, was assigned to polygons that did not contain any threat category information.

Figure 3.7. Palmiet River catchment area (dark gray), at the South West coast of South Africa. Square lines represent the quarter-degree grid. Black dots represent sampling locations in the area. Catchments are at the quaternary scale.

4. Discussion

4.1 Extent of occurrence (EOO) and area of occupancy (AOO)

The use of minimum convex polygons (MCPs) in species extent of occurrence (EOO) estimation is recommended by IUCN guidelines (2006, 2001) and has been used for freshwater invertebrates (Cumberlidge and Daniels 2008). However, the method and its alternatives, such as α-hulls, have been severely criticized (Burgman and Fox 2003, Getz and Wilmers 2004). In this study, analysis uncovered three immediate disadvantages of mapping with MCPs: MCP range estimation is (1) a very coarse, ultra-large scale approach; (2) includes vast areas of unlikely species habitat between known occurrences; (3) requires at least three records to represent any area at all. Inevitably, large areas of unsuitable habitat are included, not taking into account the biology of species nor the biogeography of the habitats. This leads to an overestimation of a species' EOO and thus underestimates the threat they are under. Furthermore MCPs are applied subjectively, not objectively. Experts could, for example, debate whether or not to include the outliers in the distributions of *Orthetrum machadoi* and *Syncordulia gracilis*, which would inevitably change the size of the EOO (Figures 3.4 and 3.5). Indeed, which point localities to include and which to mark as outliers could be debated for several of the distributions presented, including that for the widespread *Aeshna subpupillata*.

The distribution of *Chlorolestes apricans* is an extreme example in the opposite direction. It has been in decline, with a reduction from nine to four localities. The MCP range estimate method does track this, with a significant decline of area. The IUCN catchments record the decline in less than one half of the area, while the quaternary catchments do so in one quarter area. These large percentage differences make sense when one considers that points have no area, and that the MCP is the area that has been drawn between them. In contrast, the catchment already has area. In summary, area estimation based on minimum convex polygons is not encouraged for aquatic organisms.

Catchments are a more logical unit for mapping and they have already been identified as such for conservation management (O'Keeffe, 1989, Ward 1998, Saunders et al. 2002). These are biologically defined units and are often effective in protecting the entire species' habitat and the entire system of streams that the species depend on. However, scale is important, even when using the catchment as the logical unit for conservation. Nevertheless, there are several disadvantages with using the large-scale IUCN catchments. These disadvantages share similarities with the MCP method: (1) IUCN catchments over-estimate distribution; (2) over-represent changes in distribution; (3) include habitat that does not realistically represent the species distribution. The large-scale IUCN catchments are responsive to localities that are far apart, yet sampling locations that are relatively close may not be recorded as occurring in separate catchments. This contrasts to the fine-scale quaternary catchments that are much more responsive to changes in the sampling locations. For widespread species, such as *A. subpupillata*, the large IUCN catchment area (Table 3.1, Figures 3.1-3.5) may not be an over-representation of the AOO, but can be equally well tracked by the change in the number of polygons over time. Although still sensitive to tracking changes in widespread species, this large-scale approach is not useful in tracking changes in localized species over time. However, the quaternary catchment gives a more realistic representation, not only of AOO, but also of localities sampled, especially for rare species. For *C. draconicus*, for example, the quaternary catchments decrease from seven to three polygons with less than a ¼ reduction in area. Yet, in the case of the large IUCN catchments, there is a decline from six to three polygons, and the area is reduced by almost ½. Thus, appropriate scale is also important. Furthermore, *C. draconicus* is a specialist high-montane stream species. It does not venture to the coast, not even accidentally (Figure 3.2). To name a few more examples, neither does *E. nylephtha* occur in the Karoo, nor has *E. peringueyi* been recorded north of the Cederberg nor on Table Mountain (Figure 3.3). Furthermore, the widespread *Pseudagrion caffrum* also does not occur on the coast, but is restricted to the Great Escarpment, as the point localities and the quaternary catchments show (Figure 3.5). In summary, the quaternary catchment approach is much more appropriate for species mapping. With this method, areas assigned to species occurrence are small and changes in distributions are much more likely to be recorded (average 1.25 localities/polygon) than using IUCN catchments (average 1.90 localities/polygon).

While point-localities may not have area, it is the change in the distribution and number of point-localities that is the source of the change in the polygons, whether small areas (quaternary catchments) or large areas (IUCN catchments). Therefore, when using the appropriate scale catchment - such as the quaternary catchments, identified as appropriate here- in conjunction with point-localities, it is appropriate to also change the definitions of the AOO and EOO. This change fits aquatic macroinvertebrate distributions better. Firstly, rather than speaking of an AOO, conservationists should consider actual occurrence as represented by point-localities. These point-localities are self-explanatory and unambiguous, and will of course vary over time as populations establish or go extinct. Thus, point-localities are a close reflection of reality. In other words, I suggest that the IUCN definition of area of occupancy (AOO) should be redefined simply as 'occurrence', referring to known point-locality presences only. However, doing so would mean dropping the word 'area' as, mathematically' a series of points (which have no length or breadth) cannot equal an area. 'Points of occurrence' is too cumbersome and so I recommend simply 'occurrence'.

While the EOO is already defined as including only the smallest possible area in which a species is known to occur (IUCN 2001), the method needs to be changed. Furthermore, the definition of EOO needs to be clarified to avoid propagating the worrying, nearly two-decade-long trend in the primary literature, in which EOO is equated with species distribution mapping and geographic range estimation, the most recent culmination of which is presented in Jetz et al. (2008). The IUCN extent of occurrence (EOO), for freshwater aquatic species, should be redefined as: *The sum of the smallest hydrological units identified, of presently known, inferred or projected occurrences of a taxon, excluding cases of vagrancy, that are used to estimate the threat to a taxon.* A single one of those smallest hydrological units is also a conservation or management unit, identified here as the quaternary catchment. This new definition of EOO is clear and will help to avoid confusion concerning what EOO is meant to do. It will also avoid the underestimation of the threat to species, as mapping with quaternary catchments leads to a highly responsive and objective method for monitoring species decline.

Distributions were not inferred in this study, as they are in IUCN Freshwater Assessments, including southern African Odonata (Suhling et al. 2008). Inferred distributions are based on the large-scale catchments, which severely over-estimate distributions. For example, interpretation of the distribution map of *A. subpupillata* may lead to most catchment areas of South Africa being filled in. Thus, the inferred distribution could be thought of as an extension to the estimated species' EOO. While inferring distributions in this manner may be less erroneous than the MCP when dealing with localized species, it still over-estimates EOO. Furthermore, the IUCN guidelines do not define the concept of inferred distributions (IUCN 2001). Inferred distributions should be considered as an additional or complementary measure of extent of occurrence, based on expert opinion (Maddock and Samways 2000). Perhaps, if inferred distributions were used, rather than using expert opinion alone, this additional information should be based on predictive modeling of the known distributions and the most likely scenario chosen. This application has already been successfully made using point-locality distributions of South African Odonata (Finch et al. 2006).

4.2 Catchment threat status

Many conservation studies and species atlases make use of quadratic or, less commonly, hexagonal grids. This method is commonly used for conservation of terrestrial species, but also for amphibious and fully aquatic freshwater organisms (e.g. Skelton et al. 1995, Slatyer et al. 2007). In the latter case, the grid-mapping approach ignores the natural barriers to freshwater organisms that are formed by catchment boundaries. As the present study shows, grid mapping does not fit the longitudinal, unidirectional nature of streams. This pattern emerged even though the number of quaternary catchments was roughly equivalent to the number of grids. The use of a grid approach for stream threat levels would severely distort the picture of the status of South African catchments, as demonstrated in this study by the high level of unclassifiable grids and by the negative re-distribution of the conservation status levels. This would cause an under-estimation of the threat levels of South African catchments (Figure 3.6, Table 3.2). The catchment threat status layer, which was derived here from the stream conservation status layer, was mainly used in this study to demonstrate the superiority of the catchment to the grid in conservation mapping. However, the layer has the additional value of potentially giving conservation planners a simplified overview of the overall condition of a catchment. This, in turn, helps in identifying priority areas for conservation.

5. Conclusion

This study shows conclusively that area estimation based on minimum convex polygons should not be encouraged for aquatic organisms. Likewise, the use of inferred distributions should be based on predictive modeling followed by choice of the most likely scenario by experts. This study also suggests that the IUCN definition of area of occupancy (AOO) should be redefined simply as occurrence, referring to known point-locality presences only and, if future data allow, to known absences. The IUCN extent of occurrence (EOO), for aquatic species, should be the smallest hydrological unit identified as the conservation or management unit. I suggest that this unit is the quaternary catchment.

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Chapter 4

An easy-to-use index of ecological integrity for prioritizing freshwater sites and for assessing habitat quality*

Abstract: *Prioritizing and assessing the condition of sites for conservation action requires robust and ergonomic methodological tools. I focus here on prioritizing freshwater sites using two promising biodiversity indices, the Dragonfly Biotic Index (DBI) and Average Taxonomic Distinctness (AvTD). The AvTD had no significant association with either species richness or endemism. In contrast, the DBI was highly significantly associated with species richness and endemism, although the strengths of the associations were weak. These associations are related to how the sub-indices in the DBI are weighted, and how species are distributed geographically. Additionally, the DBI was found to be very useful for site selection based on its ability to measure ecological integrity, combined with level of threat, at multiple spatial scales. The AvTD was found to be useful principally for regional use. As the DBI is a low-cost, easy-to-use method, it has the additional use as a method for assessing habitat quality and recovery in restoration programmes. The DBI operates at the species level, and is therefore highly sensitive to habitat condition and has great potential for environmental assessment and monitoring freshwater biodiversity and quality. Practical, worked examples of river restoration are given here. In view of the ease and versatility by which the DBI can be employed, I recommend its testing and possible integration into freshwater management and conservation schemes elsewhere in the world.*

Keywords: conservation, prioritization, assessment, freshwater, catchments, Odonata, taxonomic distinctness, Dragonfly Biotic Index

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1. Introduction

Site prioritization for conservation action, such as the setting aside of reserves and delineation of hotspots, is usually based on biodiversity measures such as species richness, abundance, complementarity, taxonomic and functional diversity, diversity at different scales (i.e. α, β, and γ), and indices that combine some of the above measures (Magurran 2004). The most commonly used diversity measure in ecology is species richness (Jennings et al. 2008; Fleishman et al. 2006; Magurran 2004). However, there are five problems with diversity measures based on species counts alone (Warwick and Clarke 2001; Fleishman et al. 2006). Firstly, species richness is heavily dependent on sampling effort and is therefore highly sensitive to sample size and is not comparable across studies involving unknown or differing degrees of sampling effort. Secondly, species richness does not directly reflect phylogenetic diversity. Thirdly, although species richness measures can be compared across sites, which are strictly controlled by sampling design, the values of species richness cannot be compared against an absolute standard. Fourthly, the response of species richness to environmental degradation is not monotone. Indeed, Wilkinson (1999) notes that under moderate levels of disturbance, species richness may increase. Fifthly, species richness will differ markedly with different habitat types.

An additional problem with species richness is that the measure is scale-dependent (Jennings et al. 2008). Some studies of higher taxa found that areas of high endemism do not correspond with those of high species richness at regional (Prendergast et al. 1993) nor at global scales (Orme et al. 2005). However, other studies using different resolutions at the regional (Graham and Hijmans 2006) and global (Lamoreux et al. 2006) scale did find a correlation. Given two assemblages with identical numbers of species and equivalent patterns of species abundance but differing in the diversity of taxa to which they belong, the most taxonomically varied assemblage will be the more diverse (Clarke and Warwick 2001).

In response to these findings, Average Taxonomic Distinctness (AvTD) has been proposed as a biodiversity measure (Warwick and Clarke 1995; Clarke and Warwick 1998, 2001). It calculates the average taxonomic distance between any two species chosen at random from a sample. In contrast to other diversity measures, AvTD can be used in situations where sampling is uncontrolled, unknown or unequal and where data are nominal, i.e. species are present or absent. Indeed, use of simple species lists has the advantage of ensuring that no one species can dominate contributions to the index (Clarke and Warwick 1998, 2001). Measures of taxonomic diversity can be used in conjunction with species richness and rarity scores in the context of conservation (Virolainen et al. 1998). Already, taxonomic distance has gained impetus in environmental assessment (Heino et al. 2007; Ellingsen et al. 2005; Mouillot et al. 2005).

The Dragonfly Biotic Index (DBI) is also a biodiversity measure, but based on a blend of expert knowledge of the focal species and quantitative assessment (Simaika and Samways 2008a). The DBI is based on the widely recognized potential of Odonata as indicator species (e.g. Chovanec 2000), although to date the index has been used only for measuring habitat recovery (Samways and Taylor 2004). This is an extension of the fact that odonates can be used as indicators of freshwater health (Oertli 2008), ecological integrity (Smith et al. 2007; Chovanec and Waringer 2001), and global climate change (Ott 2008).

I investigate here the value and use of the AvTD and DBI for measuring ecological integrity (i.e.: species composition of habitats), and for prioritizing sites for protection. I discuss the use of the DBI in freshwater quality assessments for purposes such as restoration.

2. Methods

2.1 Background on the Dragonfly Biotic Index

As in the case of the AvTD, the DBI relies on species presence/absence data. The DBI is comprised of three sub-indices: a species relative geographic distribution, threat status based on IUCN Categories and Criteria (IUCN 2001), and species sensitivity to habitat disturbance (Table 4.1) (Simaika and Samways 2008a). Each sub-value ranges from 0 to 3. The sum of the sub-values for any one species is the standard DBI score, which can range from 0 to 9. The standard DBI for all known South African odonate species is given in Samways (2008).

To arrive at a DBI score per site, I divided the total of all the standard DBIs by the total number of species. The range of values for the DBI per site will therefore fall between 0 and 9.

2.2 Database development

Biogeographic information from South Africa (including Lesotho and Swaziland) was used here. This area is unique in that such information is not only available to potential users worldwide, via the internet (SANBI 2008), but that many taxa, including the Odonata, are well sampled. A spatial-relational database was constructed from records of adult dragonfly and damselfly collections and sightings. The database consists of a merger between Samways' database of collections and sightings (1988 to present) and a database of Pinhey's (1984, 1985) records. Additional records came from insect collections housed at the Iziko Museum (Cape Town), Albany Museum (Grahamstown), Northern Flagship Institution (Pretoria), National Museum (Bloemfontein) and National Insect Collection (Pretoria). Museum visits included verification of old records and identification of new specimens accessioned since 1984. Additional records came from new collection effort, with special emphasis on endemic species sampling, during the field seasons from 2005to 2008 in the Western and Eastern Cape. These new records extend the known geographical range of the endemic Red Listed *Ecchlorolestes peringueyi* and *E. nylephtha* (Simaika and Samways 2008b), and discoveries of two new species *Syncordulia legator* and *S. serendipator* (Dijkstra et al. 2007). From the resultant database, species distribution maps were constructed using both ArcView GIS 3.2a and ArcGIS 9.2 (Environmental Survey Research Institute 1999, 2006). The quaternary catchments map of South Africa was used for distribution mapping (SANBI 2008).

Table 4.1. Sub-indices of the Dragonfly Biotic Index (DBI). The DBI is based on the three sub-indices relating to geographical distribution, level of threat, and sensitivity to habitat change, with particular reference to invasive alien riparian trees. The scores of the sub-indices range from 0 to 3. The total DBI per species is the sum of the scores for the three sub-indices and ranges from 0 to 9. A common, widespread, not-threatened species highly tolerant of disturbance would score 0 (0 + 0 + 0), while a highly range-restricted, threatened and sensitive species would score 9 (3 + 3 + 3). Abbreviations: IUCN species threat status (IUCN 2001): LC = Least Concern, NT = Near Threatened, VU = Vulnerable, CR = Critically Endangered, EN = Endangered, GS = Global Status, and NS = National Status.

	Sub-Indices		
Score	Distribution	Threat	Sensitivity
0	Very common throughout South Africa and southern Africa.	LC; GS	Not sensitive; little affected by habitat disturbance and may even benefit from habitat change due to alien plants; may thrive in artificial waterbodies.
1	Localized across a wide area in South Africa, and localized or common in southern Africa; or very common in 1-3 provinces and localized or common in southern Africa.	NT; GS or VU; NS	Low sensitivity to habitat change from alien plants; may occur commonly in artificial waterbodies.
2	National endemic confined to 3 or more provinces; or widespread in southern Africa but marignal and very rare in South Africa.	VU; GS or CR; NS or EN; NS	Medium sensitivity to habitat disturbance such as from alien plants and bank disturbance; may have been recorded in artificial waterbodies.
3	Endemic or near-endemic and confined to only 1 or 2 Provinces.	CR; GS or EN; GS	Extremely sensitive to habitat change from alien plants; only occurs in undisturbed natural habitat.

2.3 Statistical Analysis

To ensure that equal sampling effort was compared, and that statistical analyses could be done using the presence/absence data from the compiled South African Odonata database, a minimum of 10 species per catchment was admitted for analysis (Bob Clarke, Primer-E, *pers. comm.* 2008). This decision was made after comparison of analyses with a minimum of three and then five species. Analysis with lower species numbers (a minimum of three and five species) confirmed that a minimum sampling effort of 10 species is required for meaningful analysis.

To allow for easy comparison of Average Taxonomic Distinctness (AvTD) and the Dragonfly Biotic Index (DBI), quaternary catchments were grouped into larger primary catchment areas, called zones (Figure 4.1). A count of sampled quaternary catchments in each primary catchment zone is presented in Table 4.2. These primary catchment zones are equivalent to the existing river regions used by Schulze (2006) and earlier by Midgley et al. (1994), and their convention was not altered here. Primary areas that were under-represented were clustered into larger zones where possible. Clustering was not an arbitrary process but was made by a careful, repeated elimination in Primer 5 (Clarke and Warwick 2001). First, species occurrence in each quaternary catchment was averaged by the primary catchment, using the AVERAGE function in Primer 5. The averages were then standardized and square-root transformed in a Bray-Curtis similarity matrix.. A CLUSTER dendrogram, clustered by group average, was produced using the similarity matrix (Figure 4.2). Average taxonomic distinctness (AvTD) was calculated using PRIMER 5. Analysis of variance (ANOVA) was run both on AvTD and DBI data using SPSS 13 (SPSS Inc. 2004). The Kolmogorov-Smirnov test of normality and Levene test for homogeneity of variances were employed using SPSS 13.0. The tests determined the non-normality and un-equal variance of the index data. Therefore, the Brown-Forsythe test was used as an alternative to analysis of variance. Tamhane post-hoc test was used to determine which zones differed significantly in biodiversity. To determine whether the biodiversity indices are correlated, a Spearman Rank correlation was used in SPSS 13, as the data were non-normally distributed. Recovery scores for examples used in the application of the Dragonfly Biotic Index were calculated by dividing the value before restoration by the value after restoration and expressing this as a percentage. This was done using species richness, giving the Species Recovery Score and the DBI, giving the Dragonfly Recovery Score.

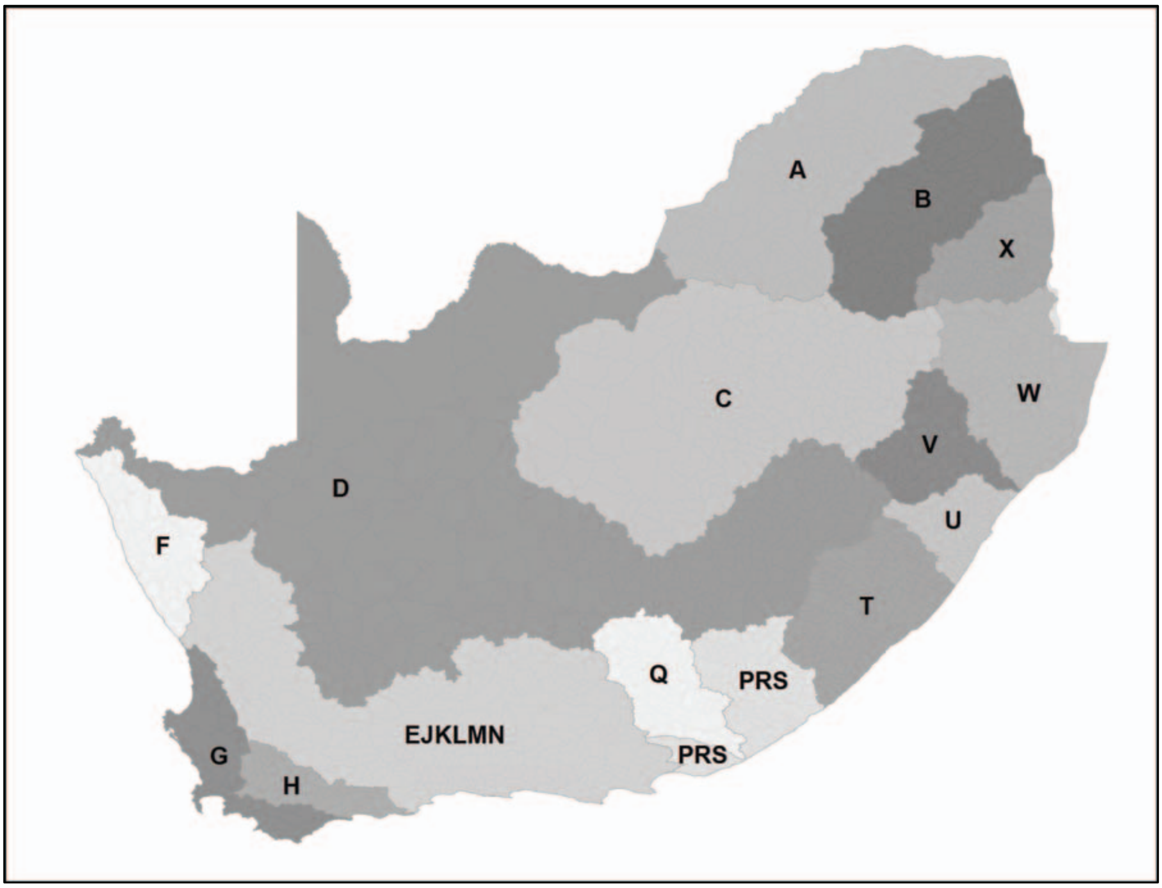

Figure 4.1. Primary catchment zones of South Africa. The Buffels and Fish river systems (F and Q) were not included in the analyses, due to insufficient sampling effort in the areas. Abbreviations are as follows: A (Limpopo); B (Olifants); C (Vaal); D (Orange); EJKLMN: E (Olifants); J (Gourits); K (Keurboom/Storm/Krom); L (Gamtoos); M (Swartkops); N (Sundays); G (Berg/Bot/Potberg); H (Breede); PRS: P (Bushmans), R (Keiskamma); S (Kei); T (Mzimvubu); U (Mkomazi); V (Tugela); W (Mfolozi/Pongola); and, X (Komati/Crocodile).

Table 4.2. Count of sampled quaternary catchments in each primary catchment zone.

Zone	Primary Catchment	Polygon Count
A	Limpopo	34
B	Olifants	23
C	Vaal	7
D	Orange	4
EJKLMN	Olifants/Gourits/Keurboom/Storms/Krom/Gamtoos/Swartkops/Sundays	15
G	Berg/Bot/Potberg	11
H	Breede	11
PRS	Bushmans/Fish/Keiskamma/Kei	9
T	Mzimvubu	8
U	Mkomazi	18
V	Tugela	16
W	Mfolozi/Pongola	31
X	Komati/Crocodile	26
Total		213

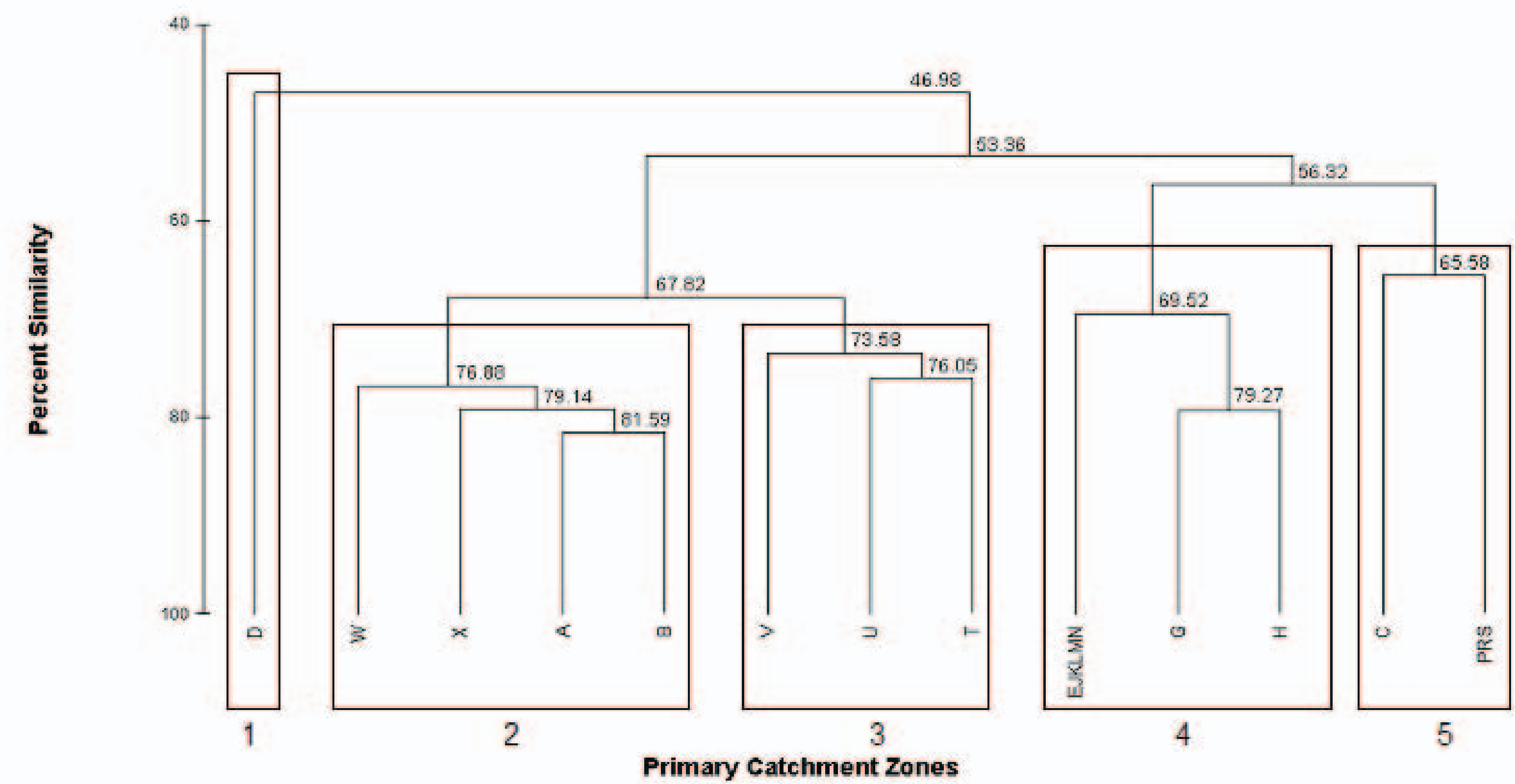

Figure 4.2. Cluster graph of the primary catchment zones. Percent similarities are given for each junction. Abbreviations for catchment zones are as follows: A (Limpopo); B (Olifants); C (Vaal); D (Orange); EJKLMN: (Olifants/Gourits/Keurboom/Storm/Krom/Gamtoos/Swartkops/Sundays); G (Berg/Bot/Potberg); H (Breede); PRS: (Bushmans/Keiskamma/Kei); T (Mzimvubu); U (Mkomazi); V (Tugela); W (Mfolozi/Pongola); and, X (Komati/Crocodile).

3. Results

The average taxonomic distinctness (AvTD) described per primary catchment zone is shown in Figure 4.3. High AvTD scores have a widespread distribution, running along the Great Escarpment of South Africa, starting with the coastal belt in the Cape, which is high in endemism, running from the West to the East Coast (G, H, EJKLMN and PRS), and extending farther inland into the Highveld (V) and KwaZulu-Natal (W, X) and northwards to the species-rich lowveld region of Mpumalanga (A, B).

The analysis of variance (ANOVA) test revealed that there are significant differences between zones ($F = 5.14$, $df = 12$, $p < 0.01$). The Tamhane post-hoc test determined which catchment zones were responsible for these differences. Catchment zone A differs significantly from EJKLMN ($p < 0.01$), G ($p < 0.00$), H ($p < 0.00$), PRS ($p < 0.00$) and V ($p < 0.00$); zone B from H ($p < 0.04$) and PRS ($p < 0.00$); zones EJKLMN and G from zone A; zone H from zones A, B, and W ($p < 0.01$); zone PRS from zone A, B, W ($p < 0.00$) and X ($p < 0.00$); zone V from A; zone W from H and PRS; zone X from zone PRS. Zones C, D, T and U did not differ significantly from any other zone.

Comparison of Figures 4.3 and 4.4 reveals that the means of the zones, while significantly different, are overall high. Thus, there are many catchments with high AvTD scores.

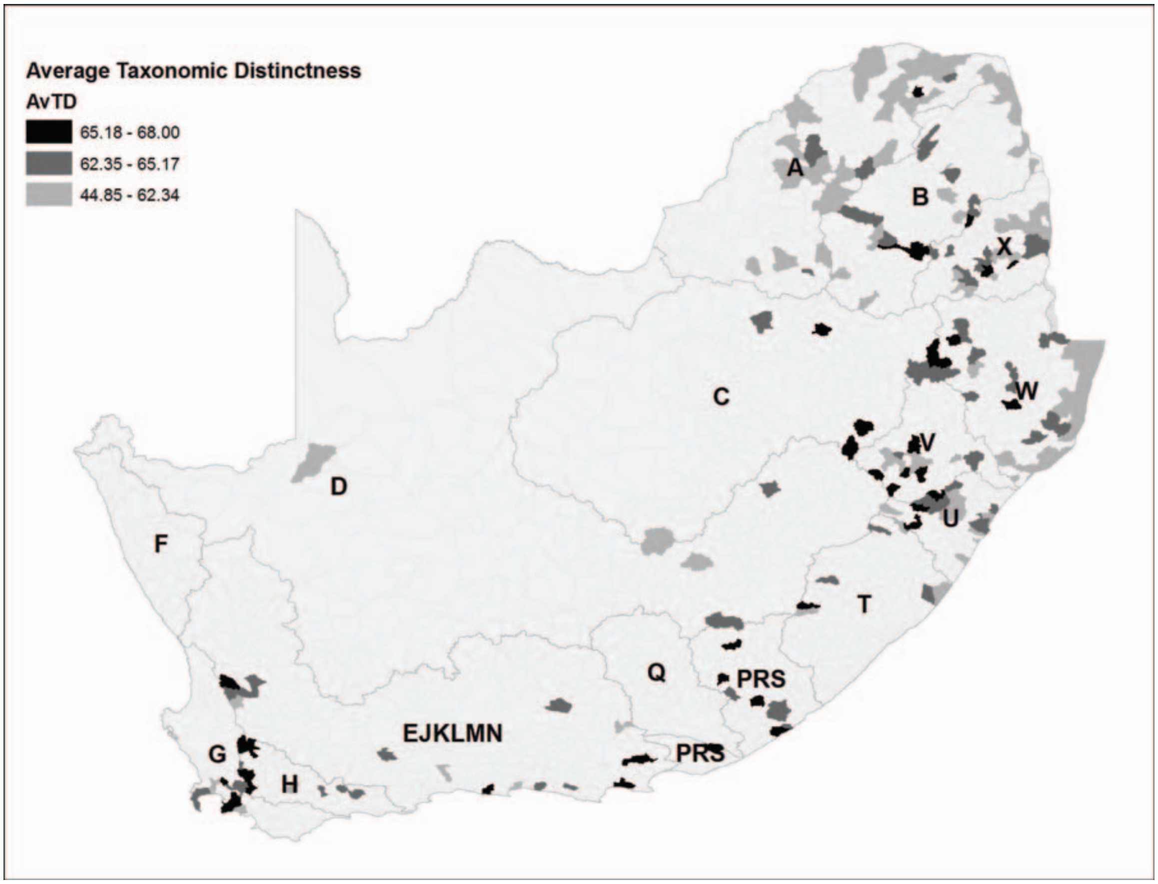

Figure 4.3. Average Taxonomic Distinctness (AvTD) of assemblages of South African Odonata per quaternary catchment. Light gray catchments indicate low AvTD value, dark gray catchments medium value, and black catchments high value.

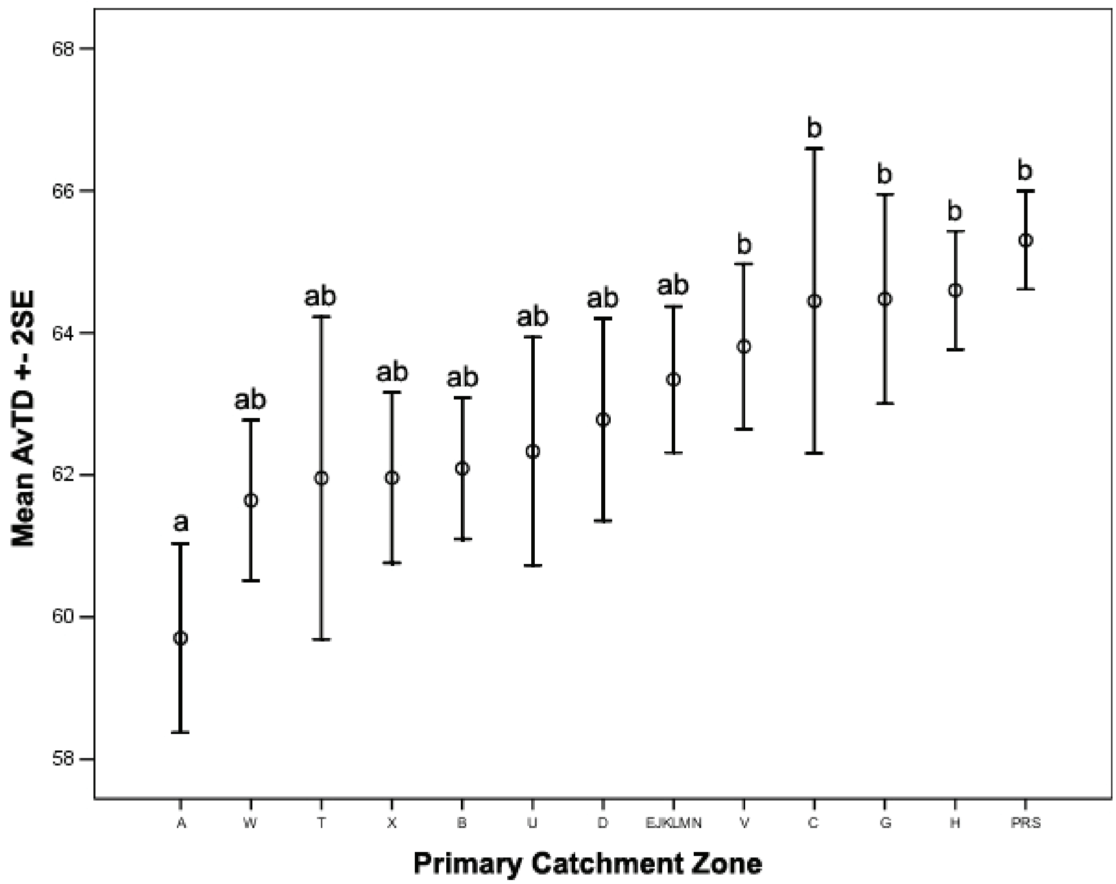

Figure 4.4. Mean Average Taxonomic Distinctness (AvTD) per primary catchment zone. An analysis of variance (ANOVA) determined that zones are significantly different ($F = 5.14$, $df = 12$, $p = 0.0001$). Catchment zones fall into three groups: a (zone A); ab (zones B, D, EJKLMN, T, U, W and X); b (zones C, G, H, PRS, and V). Error bars represent standard error (SE) ± 2.

3.1 Dragonfly Biotic Index

Visualization of the DBI scores for South African odonate assemblages is presented in Figure 4.5. A very small proportion of catchments have a high DBI score. These are all restricted to the Cape region in primary zones G and H in the South-West and EJKLMN in the South-East Cape. Most of the medium-high DBI scores are distributed south of the Great Escarpment, from the South-West Cape (G and H), along the south east coastal belt (PRS, U, W). Inland medium scores are also found in zone EJKLMN in the Cape; D in the Karoo; T in the Transkei; V in KwaZulu-Natal, and X, B and A in Mpumalanga.

The analysis of variance (ANOVA) test revealed that there are significant differences between the means of the DBIs of the primary zones ($F = 8.937$, $df = 12$, $p < 0.01$) (Figure 4.6). The Tamhane post-hoc test determined which means of the primary catchment zone were responsible for the observed differences. The mean DBIs of primary catchment zone A, B, C, H, V, and W are significantly different from at least one other catchment zone. Catchment zone A differs significantly from zone H ($p < 0.02$) and W ($p < 0.01$); zone B differs significantly from zone H ($p < 0.04$); zone C also differs significantly from zone H ($p < 0.03$); zone H differs significantly from zones A, B, C and V); zone V is significantly different from zone H ($p < 0.04$); zone W is significantly different from zone A ($p < 0.01$).

Comparison of Figures 4.5 and 4.6 confirms that the highest DBI means are in catchment zones G, H, EJKLMN and PRS. In zone EJKLMN, primary catchments K and M are most responsible for the high means. The mean of PRS is high overall.

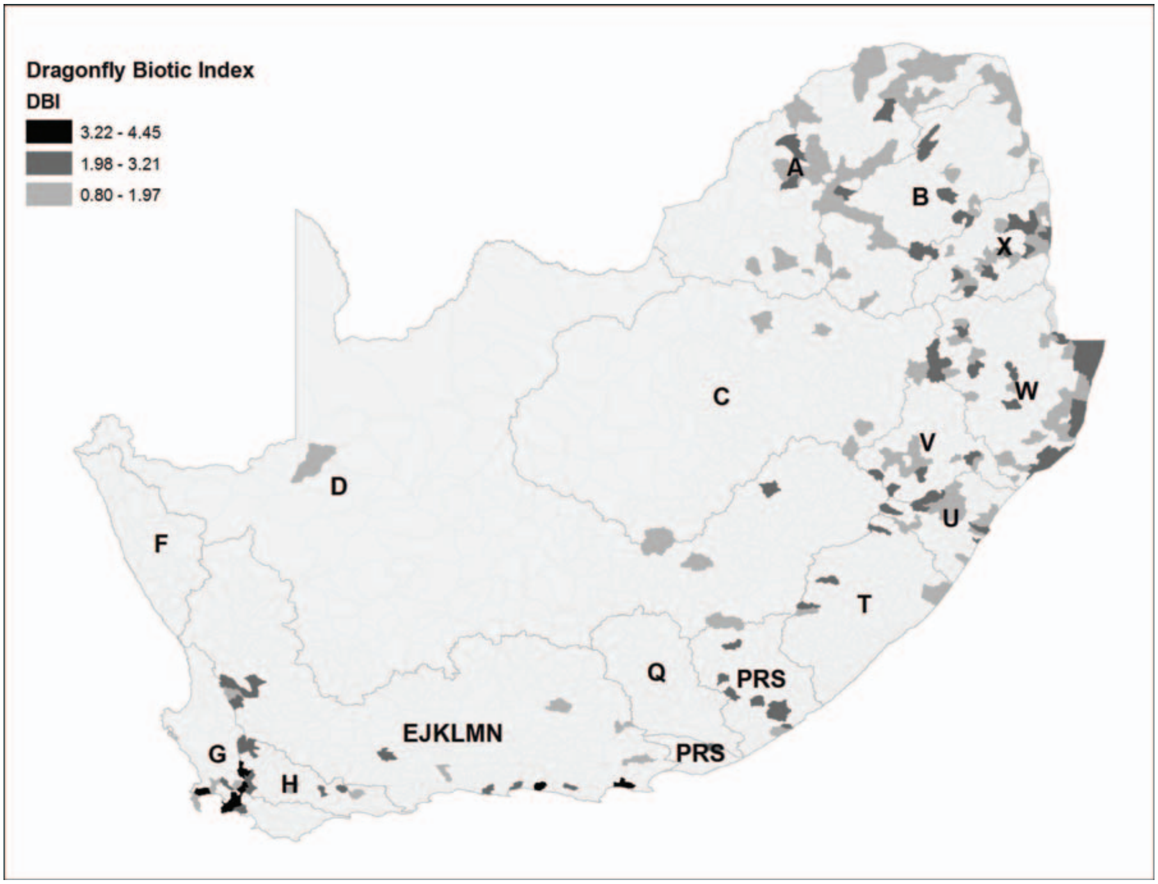

Figure 4.5. Dragonfly Biotic Index scores of assemblages of assemblages of South African Odonata per quaternary catchment. Light gray catchments indicate low DBI value, dark gray catchments medium value, and black catchments high value.

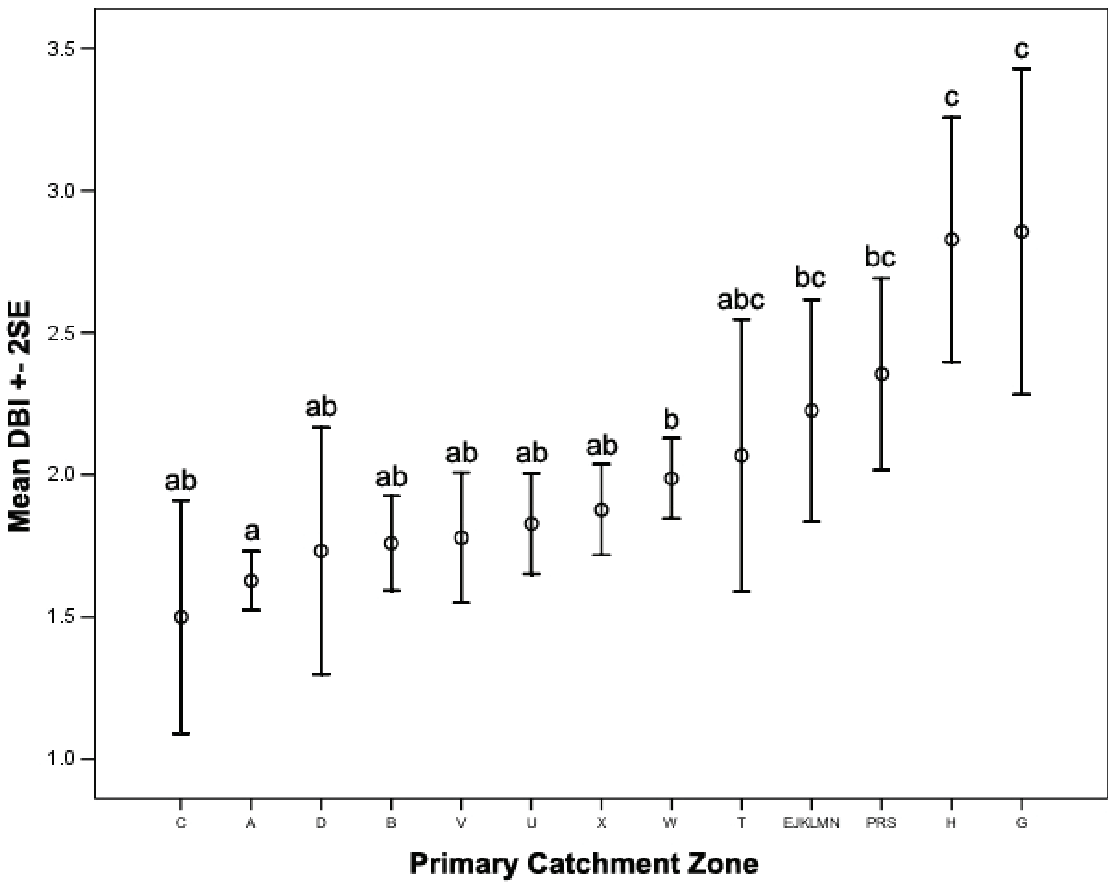

Figure 4.6. Error plot of mean Dragonfly Biotic Index (DBI) per primary catchment zone. An analysis of variance (ANOVA) determined significant differences in DBI scores between catchments ($F = 8.937$, $df = 12$, $p = 0.0001$). Primary catchment zones fall into six larger groups, a (zone A); ab (zones B, C, D, U, V, X); abc (zone T); b (zone W); bc (zones EJKLMN, PRS); and c (zones G, H). Error bars represent standard error (SE) ±2.

3.2 Comparison of AvTD to DBI

Two-tailed Spearman's rank correlation found a weak but highly significant positive correlation between AvTD and DBI ($r_s = 0.400$, $n = 213$, $p < 0.01$). The AvTD showed no association with either species richness ($r_s = -0.091$, $n = 213$, $p < 0.188$) or endemism ($r_s = 0.151$, $n = 50$, $p < 0.294$). The DBI was highly significantly correlated with species richness ($r_s = 0.209$, $n = 213$, $p < 0.01$) and with endemism ($r_s = 0.448$, $n = 50$, $p < 0.01$). However, the strength of the association for the DBI is very weak with species richness, and is weak for endemism. High DBI scores are localized in the Cape region (zones G and H). High AvTD scores have a wider distribution, particularly in catchments in zones G, H and PRS, and include zone V in the north-east region, which is poor in endemics (Figures 4.3 – 4.6). High scoring AvTD catchments are also within the species-rich zones A, B and X. Catchments in zones A, B and X score either low or medium DBI.

3.3 Practical application of the Dragonfly Biotic Index

Table 4.3 shows ten examples where dragonfly assemblage composition was recorded before and after restoration, which was achieved through removal of invasive alien trees that were shading out the naturally sunny habitats. The species are recorded as a percentage ratio (the Species Recovery Score, SRS) of the number of species after restoration compared with the number before restoration. The recovery is also given as the percentage ratio (the Dragonfly Recovery Score, DRS) of the total DBI after restoration, compared with that beforehand. In all cases, both the SRS and the DRS are above 100%, illustrating an increase in both number of species and in total DBI following restoration. Figure 4.7 shows the SRSs and the DRSs for the ten sites overlaid on a map of levels of endemism. The very high DRS values are associated with high levels of endemism, illustrating the great effectiveness of the remediation on the irreplaceable, endemic fauna. As level of endemism decreases while species richness increases, reaching the highest species richness but lowest endemism at site J, the DBI decreases in proportion to the SRS. The DBI thus has strong conservation value in that it emphasizes the threatened, narrow-range and sensitive species, and indicates their recovery when restoration is undertaken.

Table 4.3 Changes in dragonfly species richness (S) and Dragonfly Biotic Index (DBI) values following removal of invasive alien riparian trees. This recovery is expressed as a change in both percentage of species richness (Species Recovery Score) and in percentage DBI (Dragonfly Recovery Score). Scores are based on raw data, in published works, of dragonfly species changes over time.

Site	Before *S*	Before DBI	After *S*	After DBI	Recovery (%)	Recovery (%)	Co-ordinates º ' S, º ' E	Reference
A	5	8	11	48	220	600	33 59, 18 24	Simaika and Samways (2008a)
B	7	23	16	85	229	370	33 35, 19 08	Simaika and Samways (2008a)
C	7	19	15	72	214	379	33 57, 19 12	Simaika and Samways (2008a)
D	11	37	18	46	164	124	33 25, 19 17	Samways and Grant (2006b)
E	8	22	18	51	225	232	33 24, 19 17	Samways and Grant (2006b)
F	5	7	11	15	220	214	33 50, 22 26	Samways and Grant (2006b)
G	4	9	8	22	200	244	33 49, 23 50	Samways and Grant (2006b)
H	5	11	11	36	220	327	32 36, 27 25	Samways and Grant (2006b)
I	7	10	9	21	129	210	24 53, 30 45	Samways and Grant (2006b)
J	13	22	20	25	154	114	22 50, 30 36	Magoba and Samways (2009)

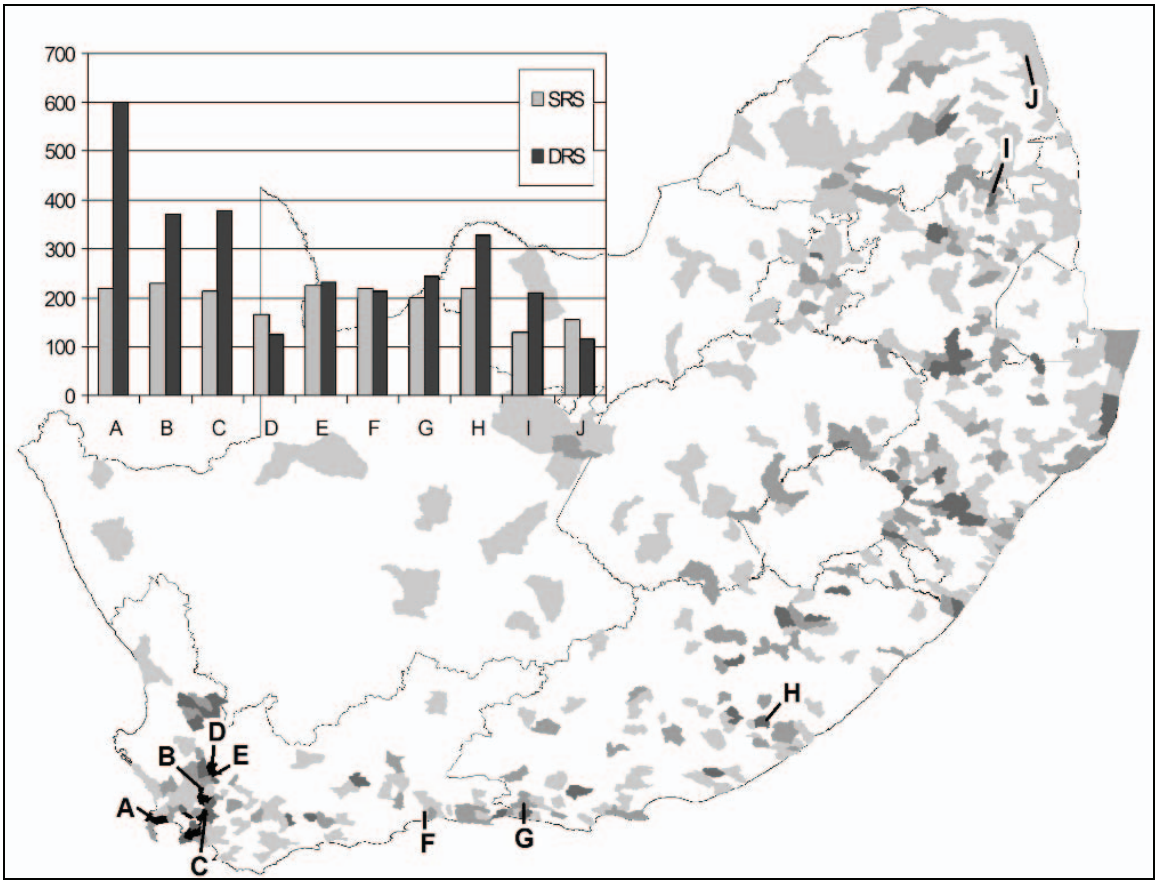

Figure 4.7 Percent recovery of dragonfly fauna at sites (A-J) following removal of alien invasive riparian trees, expressed as percent Species Recovery Score (SRS) and Dragonfly Recovery Score (DRS). Source data for sites A-J are given in Table 4.3. The recovery scores are overlaid on a map of South Africa, showing the number of national endemic dragonfly species across South Africa at the quaternary catchment scale. Light gray catchments show low levels of endemism, black ones high levels of endemism.

4. Discussion

A practical index for prioritizing sites or for assessing success of conservation action must be easy to use and provide reliable, repeatable results (McGeoch 2007). Ideally, it should also operate at the species, rather than at a higher taxonomic level, so as to be sensitive to the various subtle characteristics of, and changes in, the habitats (Smith et al. 2007).

Many biodiversity measurements have fallen short of the ideal because they have consisted of simple counts of the numbers of species (species richness), an observation voiced by many (e.g. Jennings 2008; Price et al. 1999). Researchers have thus suggested that aggregate biodiversity levels are more important in identifying priority sites (e.g. Dinerstein and Wikramanayake 1993; Pressey et al. 1993) or, alternatively, a measure of the species' identities should be used (Jennings et al. 2008; Clarke and Warwick 2001). Therefore, it was appropriate here to test the validity of two biodiversity indices for prioritizing freshwater sites: the Dragonfly Biotic Index (DBI) and the Average Taxonomic Distinctness (AvTD).

4.1 Comparison of biodiversity indices

There was a weak but significant relationship between the AvTD and the DBI. Both indices are based on presence/absence records. Yet, these indices are very different, in that the first is based solely on weighted taxonomic relatedness (Clarke and Warwick 2001), while the latter is based on weighted geographic distribution, conservation status and sensitivity to disturbance (Simaika and Samways 2008a).

The DBI is based on a mixture of objective science and expert opinion and gives more weight to geographically restricted, Red Listed and disturbance-sensitive species than to any other species. Its main thrust lies in identifying species of global conservation concern. In other words, the DBI gives priority to rare and endemic Red Listed species. In South Africa, these occur, as do many other taxa, mainly in the South-West Cape and Eastern Cape, regions characterized by endemic Corduliidae and Synlestidae (Figures 4.5 and 4.6). The remaining areas, particularly the North-East, are dominated by a species rich Afro-tropical element.

In contrast to the DBI, the AvTD is sensitive to the taxonomic relatedness of species. It is based on the intuitive principle that an assemblage of distantly related species is more diverse than an assemblage of closely related species (Warwick and Clarke 2001). In each assemblage, the AvTD tracks this principle throughout the country from the South-West to the North-East. High AvTD values have a widespread distribution, along the Great Escarpment of South Africa, starting with the coastal belt in the Cape, which is high in endemism and running from the West to the East coast, expanding farther inland into the Highveld and KwaZulu-Natal and northwards to the species-rich lowveld region of Mpumalanga.

This is where the greatest difference between the AvTD and DBI occurs. There are far fewer endemics in the North-East, and the DBI reflects this clearly. The DBI was highly significantly correlated with species richness, although the strength of the association was very weak or non-existent. The DBI was more strongly correlated with endemism than with species richness, although this was also a weak correlation. The AvTD, in contrast had no significant correlation with either species richness or endemism.

The DBI may be very weakly, although highly significantly, correlated with species richness, because it is intrinsically dependent on how the sub-indices in the DBI are weighted and distributed. For example, a species assemblage of only ten highly sensitive and threatened Cape endemic odonates at a site in the Cape Floristic Region may score an average (i.e. score per site) DBI of seven, while at a site in the species-rich region of KwaZulu-Natal, an assemblage of 25 widespread Afro-tropical species may only score an average DBI of two.

In terms of global prioritization of habitat conservation, the DBI is more readily applied than the AvTD. Conservation organizations would be interested in the results of the DBI, as the index identifies priority sites for the conservation of highly threatened and sensitive species. The AvTD can also be used to identify areas of conservation concern, but it is more relevant at a national level. For example, different provinces of South Africa may want to conserve their own hotspots of biodiversity, in a regional context, taking species representativeness into account.

4.2 Use of the Dragonfly Biotic Index for environmental monitoring

The use of the DBI and AvTD has been suggested for environmental monitoring (Simaika and Samways 2008a; Warwick and Clarke 1995). The AvTD has already been applied to tracking habitat disturbance (e.g. Warwick and Clarke 1998; Mouillot 2005), while the DBI has been employed for assessing the success of stream restoration through removal of invasive alien trees, a key threat to various aquatic organisms (Samways and Taylor 2004).

Application of the DBI to tracking habitat recovery from alien riparian plant invasion is termed here the Dragonfly Recovery Score (DRS), which is the total DBI after restoration compared with the value before restoration. The results (Table 4.3 and Figure 4.7) are clear, with restoration resulting in an increase in both species richness (the Species Recovery Score, SRS) and the total DBI (DRS) at all the sites. However, the added value of the DBI over species richness is that it weights those species that are geographically restricted, threatened and sensitive. The outcome in practical terms is that the restoration activities were highly beneficial not just to the common, widespread generalists but noticeably also to the irreplaceable, narrow-range endemics. Thus, the DBI is a very effective method for monitoring river remediation, especially for those species of conservation concern.

In terms of practicality, the individual DBIs for all species, including a species description and other essential information is given in Samways (2008). This information is therefore readily available to managers no individual species assessments need be taken. This ‘canned’ information is simply ready to plug into the total DBI (and DRS) calculations, which makes it easy to use. The DBI has the added advantage that species can be easily and rapidly identified and habitats scored while in the field. Thus for local rapid environmental impact assessments and habitat monitoring schemes, the DBI is a low-cost, easy-to-use alternative. I therefore recommend the use and integration of the DBI into management and conservation schemes.

Previous work has shown a strong correlation between adult dragonfly scores and macroinvertebrate scores (Smith et al. 2007). This suggests that the DBI, as a measure of ecological integrity, could be used alongside macroinvertebrate scores (e.g. Dickens and

Graham 2002) for freshwater health assessments. However, the exact relationship between the DBI and macroinvertebrate scores requires further, detailed exploration.

Despite the obvious and very positive advantages of the indices presented here. I emphasize that all the various elements of biodiversity cannot be encapsulated within a single index (Warwick and Clarke 1995; Price et al. 1999). Furthermore, species presence-absence data, whether using taxonomic distinctness or a combined index based on geographic distribution, threat and sensitivity, are not the only facets of diversity. The distribution of individuals among species (evenness), for example, is another very important element (Price et al. 1999) and the particular abundances of species may be important for maintaining significant functions and services (Luck et al. 2003). Finally, study of a single taxon, including the Odonata, should not be taken simply at face value to represent overall biodiversity (Price et al. 1999; Oertli 2008), a situation easily remedied by concordance studies with other taxa.

4.3 Practicality and general applicability of the Dragonfly Biotic Index

The DBI requires a good record of dragonfly species in an area under investigation (e.g., a 100 m stretch of stream, subsection of marshland or portion of catchment). As found in other studies, five visits to a site incorporating slow walking of the banks of the waterbodies is usually sufficient (Schmidt 1985). It is often necessary to supplement this activity with searches of dense vegetation for crepuscular species (for example *Gynacantha* and *Zyxomma* species). The only equipment required is an aerial net for confirmation of species identity and a 10X+20X hand lens for close examination of diagnostic characters (e.g. genitalia). A good field guide of the local odonate fauna, including species habitat tolerances, geographical distributions and some indication of level of threat, is also necessary. When more knowledge becomes available, this can be built into a more comprehensive field guide, as has been done for South African dragonflies (Samways 2008). Thus the method initially will have some challenges where the dragonfly fauna is poorly known. However, it is not out of the question to establish some preliminary values for individual species DBIs, refining them as more information becomes available. Also, there needs to be some knowledge of the flight periods to ensure all species are accounted for (Samways and Grant 2006a, 2006b).

Employing the DBI will inevitably bring the criticism that adults may not represent the larvae, and larvae should also be used in the index. This can be countered on various points. Firstly, a comparable sample of larvae requires far more sampling effort, because sampling in water is awkward and larvae can be very cryptic and live in inaccessible places (Niba and Samways 2006a). Secondly, if a good sample is obtained, only final-instar larvae can be identified to the species level. Thirdly, in many countries, including South Africa, a large proportion of dragonfly larvae remain undescribed and their identification requires more effort than that of adults. Fourth, adults typically mate and oviposit only in suitable freshwater habitats, thus residency of most species collected in mating habitat can be assumed. Should there still be skeptics, one could argue that the only true record of residency is not the larvae but the exuviae, left behind after emergence (Ott et al. 2007). This is the only true demonstration that the habitat in question is suitable to odonates in both the aquatic and aerial parts of the life cycle.

The total DBI records the ‘core resident species’ (Niba and Samways 2006b). Some vagrant species will of course also be recorded, particularly when more intensive searches over longer periods of time are done. The occasional, additional records, however, tend not to affect the total DBI to any great extent. Thus, the overall score of the DBI is the contribution by core resident species.

While I have presented the results here for one country, the concept of the DBI could be easily adapted elsewhere. However, this depends on the number of species in the odonate fauna, its breadth of geographic distribution, Red List status and sensitivity to disturbance. Where more or alternate information is available, the index could be expanded to include sub-indices such as habitat tolerance and relative abundance. The limit to the DBI is that odonates may not be good surrogate species for other taxa, owing to lack of concordance (Prendergast et al. 1993), although they have potential use as umbrellas for wetland plant species (Bried et al. 2007). Nevertheless, the easy use of the DBI and its sensitivity mean that it is a useful tool in stimulating conservation action.

5. Conclusion

The DBI is very useful for site selection as well as for measuring ecological integrity at multiple scales, while the AvTD is useful principally for regional use. The DBI and AvTD are correlated, which suggests that they could be used on a complementary basis for to prioritize sites. The DBI is a low-cost, easy-to-use method and is already used for measuring habitat recovery. It has great potential for environmental assessment and monitoring freshwater biodiversity, especially as a complement to freshwater quality assessments that use macroinvertebrate scores. I thus recommend its integration into freshwater management and conservation schemes.

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Chapter 5
GENERAL DISCUSSION

The threats to South Africa's freshwater systems are synergistic, as evidenced by the pressures exhibited on its biota (Davies and Day 1998), particularly the dragonflies (Samways 2004, 2006). This is of great concern, as stress on aquatic biota threatens the survival of species and may even cause freshwaters to cease functioning normally (Millennium Ecosystem Assessment 2005). This could lead to even greater challenges in a country already facing enormous demands on clean drinking water (Davies and Day 1998). To monitor the condition of wetlands, the present study employed dragonflies as an indicator taxon for habitat integrity (Córdoba-Aguilar 2008). The study's findings and their implications for further study are listed below:

1. Area estimation based on minimum convex polygons should not be encouraged for aquatic organisms. Likewise, the use of inferred distributions should be based on predictive modeling followed by a choice of the most likely scenario by experts.

2. This study also suggests that the IUCN definition of area of occupancy (AOO) should be redefined simply as occurrence, referring to known point-locality presences only and, if future data allow, to known absences.

3. The IUCN extent of occurrence (EOO), for aquatic species, should be defined as 'the sum of the smallest hydrological units identified, of presently known, inferred or projected occurrences of a taxon, excluding cases of vagrancy, that are used to estimate the threat to a taxon'. A single hydrological unit is also the conservation or management unit. Currently, that unit is the quaternary catchment.

4. The Dragonfly Biotic Index (DBI) is very useful for site selection as well as measuring ecological integrity at multiple scales, while the Average Taxonomic Distinctness Index (AvTD) is useful principally for regional use.

5. The DBI and AvTD are correlated, which suggests that they could be used on a complementary basis to prioritize sites.

6. The DBI is a low-cost, easy-to-use method and already in use for measuring habitat recovery (Simaika and Samways 2008). It has great potential for environmental assessment and monitoring freshwater biodiversity, especially as a complement to freshwater quality assessments that use macroinvertebrate scores. I thus recommend its integration into freshwater management and conservation schemes.

7. The DBI is sufficiently sensitive, easy-to-use and robust to be of great value to conservation managers interested in wetland assessment, monitoring and restoration (Simaika and Samways 2008).

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