

Summer 7-22-2011

Discontinuities: Predicting Invasions and Extinctions

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DISCONTINUITIES: PREDICTING INVASIONS AND EXTINCTIONS

by

Aaron Lotz

A DISSERTATION

Presented to the Faculty of

The Graduate College at the University of Nebraska

In Partial Fulfillment of Requirements

For the Degree of Doctor of Philosophy

Major: Natural Resource Sciences

(Applied Ecology)

Under the Supervision of Professor Craig R. Allen

Lincoln, Nebraska

August, 2011

DISCONTINUITIES: PREDICTING INVASIONS AND EXTINCTIONS

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University of Nebraska, 2011

Advisor: Craig R. Allen

In this dissertation, I explore multiple tenets of the textural discontinuity hypothesis, which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape. Landscapes form hierarchies that are structured by vegetative, geomorphological and contagious disturbance processes. The spatial and temporal patterns inherent in landscapes reflect numerous processes, interacting on distinct scales, which shape the assembly of animal communities. Analysis of body mass patterns and functional group distributions has been suggested as methods to provide insight about these underlying hierarchical processes. Scientists have posited that species at the edges of body mass aggregations may be exposed to highly variable resources. This dissertation focuses on the distribution of biological diversity in space and time and socio-ecological factors that are contributing to the worldwide increase in invasive and endangered species.

I analyzed invasions and extinctions of birds and mammals across five Mediterranean-climate ecosystems and in 100 countries using an information-theoretic approach. All body mass distribution data analyzed were discontinuous. This work provided further support for Holling's textural discontinuity hypothesis. Alpha diversity of function increased in 9 out of the 10 Mediterranean-climate ecosystems analyzed when NIS were introduced into the community. After the introduction of NIS, I observed a decrease in cross-scale redundancy of functional groups in mammals and when both

taxonomic groups were combined. In Eocene Epoch mammal data, speciation events were not detected near body mass aggregation edges. Only 64% of the biomes in mammals had ecoregions with similar structure and only 50% of the biomes in birds had ecoregions with similar structure, which may be a result of coarse landscape classification schemes. GDP per capita was positively correlated with the proportion of NIS bird and mammal species within a country. Resilience of a country was correlated to life expectancy. As life expectancy increased, resilience of a country decreased. Results may help us make proper management decisions in monitoring particular non-indigenous species and focus conservation efforts on those native species.

DEDICATION

For Barry and Susan

ACKNOWLEDGMENTS

I wish to express sincere appreciation to Dr. Craig R. Allen for his assistance in the preparation of this manuscript. I would like to thank Dr. Pablo A. Marquet, Dr. Johannes Knops, and Dr. Sandra B. Zellmer for their challenging comprehensive exam questions and their helpful comments on my work. I also wish to thank Dr. Andrew J. Tyre for his statistical expertise and his ability to fit me into his hectic schedule. In addition, special thanks to Dr. Tala Awada whose open door policy, concern for my progress, invaluable encouragement, advice, friendship and laughter helped me get through this doctoral program. Thanks also to Dr. David Kitchen, who persuaded me to stay in school and not go to Africa to kill poachers, for his lifetime of friendship, and staunch support throughout my academic career. Thank you to Dr. Jim Brandle for giving me the opportunity to teach children about science and for always wanting to know how I was doing. I would like to thank Millard Baker for reminding me to fight the power and Kristen Zellmer for being there to dish out reality and give me a good kick in the pants when I needed it. I would also like to thank, Brenda & Chris Pracheil, Dustin Martin, Ingrid Barcelo, Luis Ramirez, Dr. John Quinn, Cassie Manhart, Timothy Weber, Christopher Lowrey, Nate Gosch, Ryan Bjerke, Justin Williams, Donald Wardwell, Carla Knight, Christina Amarillas, Andres Baeza, Fernando D. Alfaro, and Daniela Rivera for their valuable input, supportive laughter and keeping me focused throughout my degree. Thank you to my brother, Jonathan Lotz, for providing comic relief and brotherly support throughout this endeavor. Finally, I wish to thank my parents Dr. Barry Lotz and Susan Lotz for being there to support me, both mentally and financially, as I went through the trials and tribulations of this journey.

GRANT INFORMATION

The James S. McDonnell Foundation-Studying Complex Systems and the Resilience Alliance.

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CHAPTER 1: INTRODUCTION

One of the major unresolved problems, at the forefront of worldwide environmental concerns, is the increase in invasive and endangered species. Changes in biodiversity due to human actions have been more profound in the last 50 years than in the previous entirety of recorded human history. An astonishing 52% of cycads, 32% of amphibians, 25% of conifers, 23% of mammals, and 12% of bird species are currently threatened with extinction (Millennium Ecosystem Assessment 2005). As native species decline, introduced non-indigenous species may become established and these can affect ecosystem processes and can potentially lead to the further extinction of native species (Fors & Allen 2002). This potential loss of ecological processes inherently affects landscape structure and dynamics, including predator-prey interactions, dispersal, foraging behavior and functional group composition.

In this dissertation, I explore multiple tenets of the textural discontinuity hypothesis, which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape (Holling 1992). Landscapes form hierarchies that are structured by vegetative, geomorphological and contagious disturbance processes (Holling 1992). The spatial and temporal patterns inherent in landscapes reflect numerous processes, interacting on distinct scales, which shape the assembly of animal communities (Turner 1990). In order to understand these patterns, one must understand the interactions between organisms and between organisms and their environment.

Analysis of body mass patterns and functional group distributions have been suggested as methods to provide insight about these underlying hierarchical processes, as

both are important in ecosystem-level biota structure (Forys & Allen 2002). Studies have reported a discontinuous body mass distribution reflecting the architecture of the landscape (Havlicek & Carpenter 2001, Allen & Holling 2008). These studies support Holling's (1992) textural discontinuity hypothesis, which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape. Other studies have demonstrated the influence of community interactions on body mass distributions (Oksanen et al. 1979, Stubblefield et al. 1993, and Nummi et al. 2000). These studies support Hutchinson's (1959) community interaction hypothesis, which states that in the process of community formation species may be displaced, unfilled niches may be occupied, and niches may be partitioned. Community interactions may be most important at local scales, important only after community entrainment due to landscape patterns (under the premise of the textural discontinuity hypothesis) (Allen et al. 2006).

Mediterranean-climate ecosystems (in distinct regions of Chile, Africa, California, Australia, and Spain) provide replicated examples of ecological convergence and are similar in ecological structure and function, yet contain fauna that has been evolutionarily isolated (Peterson et al. 1998). Thus, Mediterranean-climate ecosystems enable the comparison of similarities in scale-specific vertebrate community structure. If systems with similar ecological structure provide similar opportunities for animals, then these geographically disparate systems should have similar body mass distributions despite geographic and evolutionary isolation. If the vertebrate body mass structures are similar among these systems, the entrainment hypothesis of Holling's textural discontinuity hypothesis is supported.

Mediterranean-climate regions are also densely populated by humans and have been altered by human habitation and landscape transformation. These areas have been invaded by numerous non-indigenous vertebrate species and declines and extinctions of native species have transformed faunal community composition. Analysis of body-mass aggregations has been suggested as a predictor of invasiveness, endangerment and nomadism (Allen et al. 1999, Allen & Saunders 2002). The mix of native and non-indigenous species in Mediterranean-climate ecosystems provides excellent data to investigate the effects of invasive species on functional group composition and changes in body mass distributions, as well as how these changes affect biodiversity and functional representation at various scales. Invasive species may alter: 1) alpha diversity, which refers to the diversity within a particular ecosystem, 2) beta diversity, which refers to the diversity between two ecosystems and 3) gamma diversity, which is the overall diversity across all ecosystems (Balée 2006, Fridley et al. 2007).

There is very little information on the effects of paleoecological time on changes in body mass distributions. Smith et al. (2004) suggest that, over evolutionary time as lineages speciate and diversify, species do not occupy a greater range of body sizes. Holling et al. (2002) suggest that it takes extreme disturbances over paleoecological time and space to substantially change body mass distributions. Lambert and Holling (1998) analyzed body mass distributions of the Pleistocene North American mammal extinctions and suggested that they reflected changes in key mesoscale aspects of the landscape. The Siberia megafauna extinction at end of Pleistocene (Folke et al. 2004) and great mammal faunal crash in North America at the end of the Miocene (Lambert 2006) may have

triggered irreversible regime shifts and a closer examination of body mass distributions may prove insightful.

Organisms of different body sizes have different requirements for resources and operate at different spatiotemporal scales (West et al. 1997). Therefore, an animal cannot simultaneously interact with multiple scales, but has to specialize at a single scale or shift between two scales (Allen & Saunders 2002). Peterson et al. (1998) suggest functional diversity within body mass aggregations and redundancy of functional groups across scales increases resilience. Little is known with respect to ecological function and whether a species in the same guild or functional group is dispersed across scales. If body mass structure reflects landscape structure at these different scales, this will support the textural discontinuity hypothesis and create a fundamental link between landscape and community ecology.

There is a critical need for integrated concepts and research capable of uniting the natural and social sciences (Pickett et al. 1997). Studies have recognized the need to couple human systems with environment systems (Turner et al. 2003), the convergence of environmental and financial markets (Sandor et al. 2002), the importance of socio-cultural dynamics in natural resource management (Stratford & Davidson 2002) and the tremendous impact of humans on the environment in comparison to other species (Fowler & Hobbs 2003). Numerous studies have focused on only one aspect of the socio-ecological relationship such as carbon emissions (Kratena 2004), water (Postel 2003) or human population growth (Struglia & Winter 2002).

At present, only three projects have attempted to focus on and integrate multiple socio-ecological factors at a national scale, with an emphasis on their roles in an

ecologically sustainable society, into an index of values that can be ranked and compared. In 2003, the Global Footprint Network was established in an effort to establish and maintain a sustainable future. As part of that effort, the Ecological Footprint was created. This metric, comprised of 5 levels and 6 sub-categories, calculates how much natural resources we have, how much we use and who uses it in order to track human demands on the biosphere (Ewing et al. 2008). The U.S. National Aeronautics and Space Administration's (NASA) Socioeconomic Data and Applications Center (SEDAC) published three indexes, the 2005 Environmental Sustainability Index (ESI) (Esty et al. 2005), the 2006 Environmental Performance Index (EPI) (Esty et al. 2006) and the 2008 EPI (Esty et al. 2008). Each index was developed in order to explore the relationships, at a national scale, between multiple socio-ecological factors and their effect on a country's environmental performance and sustainability. These reports demonstrated that as humans, we are an integral part of the environment and have a tremendous impact on the environment around us. These indicators represent another layer of the ecosystem, previously unaccounted for in ecological research. The economic, demographic, environmental and societal metabolism facets of humanity are not isolated, but highly integrated and have profound impacts on the world's ecosystem and fauna. I suggest these socio-ecological landscape factors may be able to provide clues as to the invasiveness and endangerment of species around the world.

This dissertation focuses on the distribution of biological diversity in space and time and socio-ecological factors that are contributing to the worldwide increase in invasive and endangered species. I begin this document by comparing the body mass distributions among ecosystems with similar ecological structure and examining the

effects of invasions and extinctions (Chapter 2) and changes in alpha, beta and gamma diversity (Chapter 3). I then explore changes in body mass distributions over paleoecological time to examine speciation events (Chapter 4). I then examine the effects of spatial scale on body mass distributions to determine the extent to which the textural discontinuity hypothesis remains valid (Chapter 5). The final research chapter of this dissertation focuses on the socio-ecological factors that may be contributing to the rising number of invasive and endangered birds and mammals (Chapter 6). Finally, I conclude with the significance of this work (Chapter 7).

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CHAPTER 2: BODY MASS DISCONTINUITIES IN MEDITERRANEAN-CLIMATE ECOSYSTEMS

Over the last 35 years the population status of 1,686 vertebrate species has declined by 30% (Loh et al. 2008). Climate change, habitat loss, habitat degradation and the introduction of non-indigenous species (NIS) are just some of the factors that have contributed to an estimated three orders of magnitude increase in the rate of species extinction over the past few hundred years and significant changes in biodiversity worldwide (Millennium Ecosystem Assessment 2005). Invasive species have contributed to the decline of vertebrate species. The economic costs of invasive species exceed \$120 billion a year in environmental damages and are the primary cause for decline for approximately 42% of the United States' endangered species (Pimentel et al. 2005). Identifying the cause of invasions and the ability to reliably predict invasions are very poor (see Gallien et al. 2010). A better understanding would improve management and lessen the potential impacts on native species.

Landscapes form hierarchies that are structured by vegetative, geomorphologic and contagious disturbance processes (Holling 1992). The spatial and temporal patterns inherent in landscapes reflect numerous processes, interacting on distinct scales, which potentially shape the assembly of animal communities. Allen et al. 2006 reviewed five non-mutually exclusive hypotheses for observed body mass patterns, each operating at different scales; specifically energetic, phylogenetic, biogeographical, textural discontinuity and community interaction hypothesis. The authors suggest that their relevance varies with the scale at which each is applied. Community interactions produce

structure at local scales, the energetic and textural discontinuity at intermediate scales and the phylogenetic and biogeographical at broad scales. Many studies have reported a discontinuous body mass distribution reflecting the ecological structure of the landscape (Restrepo et al. 1997, Raffaelli et al. 2000, Havlicek & Carpenter 2001, see Allen & Holling 2008). These studies support the textural discontinuity hypothesis (TDH), which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape (Holling 1992).

Organisms of different body sizes have different requirements for resources and operate at different spatiotemporal scales (West et al. 1997). Therefore, an animal cannot simultaneously interact with multiple scales, but has to specialize at a single scale or shift between two scales (Allen & Saunders 2002). Allen et al. (1999) suggest that the gaps between body mass aggregations represent scale breaks and that these points in the body mass distribution are highly susceptible to change in structure and ecological processes. Analysis of body-mass aggregations has been suggested as a predictor of invasiveness and endangerment (Allen et al. 1999). Allen & Saunders (2002) reported nomadic bird species in southcentral Australia were significantly associated with the edge of body mass aggregations. Allen et al. (1999) reported in four different south Florida taxa (birds, mammals and herpetofauna), that more invasive and declining species exist at the edge of body mass aggregations than could be expected by chance.

Here I examined whether invasive and endangered species were nonrandomly distributed in vertebrate body mass structures among five Mediterranean-climate ecosystems. Mediterranean-climate ecosystems are projected to incur the highest loss of biodiversity of all terrestrial ecosystems by the year 2100 (Sala et al. 2000) with South

Africa and southwestern Australia expected to undergo the most significant impact (Klausmeyer & Shaw 2009). These regions are interesting to ecologists and conservation biologists for a variety of reasons, including the ecological convergence mentioned above. They are biologically rich, with a relatively high level of endemism (Mittermeier et al. 2005). Mediterranean-climate regions support large human populations, resulting in extensive, rapid, anthropogenic transformation. In a study examining key threats and trends in Mediterranean-climate ecosystems, the California-Baja California region had the highest population density and highest percent urban area of any of the five Mediterranean regions examined, while southwestern Australia had the highest percentage of high intensity agriculture (Underwood et al. 2009). Compared to other continental areas, Mediterranean regions have been invaded by a large number of non-indigenous organisms, including vertebrates. Concomitant with invasions, declines and extinctions have transformed the faunas of Mediterranean ecoregions (Di Castri 1991).

Mediterranean-climate regions provide replicated examples of ecosystems that are similar in ecological structure and function, yet contain fauna that has been evolutionarily isolated (Cody and Mooney 1978, Peterson et al. 1998, Abbott and Le Maitre 2010). Thus, Mediterranean-climate ecosystems enable the comparison of scale specific vertebrate community structure and should exhibit similar body mass patterns. In order to test Holling's (1992) textural discontinuity hypothesis, I determined whether discontinuous body mass patterns existed in each of the five Mediterranean-climate ecosystems and examined whether the vertebrate body mass structures are similar among these systems. A lack of discontinuous body mass patterns and a lack of similarity among

these systems would provide strong evidence against the textural discontinuity hypothesis.

METHODS

Study Areas

Mediterranean-climate ecosystems occur in central coastal Chile, the Western Cape Province of South Africa, San Diego County, California, the South West Botanical Province in southwestern Australia, and Spain. Delineation of Mediterranean-climate regions varies by author, especially whether one considers primarily climatic or vegetative boundaries, therefore multiple sources were used to gain a consensus on the extent of Mediterranean-climate habitat in each region (Akin 1991, Di Castri 1991, Hobbs et al. 1995). In general, these ecosystems are characterized by winter rains with a pronounced dry-season, and frequent fires (Hobbs et al. 1995). Vegetation is mainly grassland and brush including Chaparral (California), Mattoral (Chile), Maquis (Spain), Fynbos (South Africa) and Mallee or Kwongan (Australia). Although the entire Mediterranean-climate ecosystem in California and Australia were not sampled, body mass distribution analysis is robust to approximately 18% (mammals) and 15% (birds) omission error (Sendzimir 1998). Therefore, my results are limited in scope to the areas that I selected.

Native species

I examined two terrestrial vertebrate groups, birds and mammals. Species lists for Mediterranean-climate ecosystems were obtained from published sources. Californian mammals were determined from a specific study of their habitat preferences and

distribution (Quinn 1990). Unitt (2004) and Vuilleumier (1991) provided a thorough list of the birds of San Diego County. Vuilleumier (1991) provided a list of Chilean birds and Chilean mammals were provided by Miller (1981). Spanish mammals were determined from Cheylan (1991). Spanish birds were compiled from the comprehensive works of Cramp (1978 – 1994). Mammals residing in the Mediterranean-climate Fynbos of the southern cape of South Africa were determined from Smithers (1983). Winterbottom (1966) provided data on the avifauna of Mediterranean shrublands in South Africa. Mammal data for Australia were compiled from Strahan (1995). Australian birds were determined from Saunders and Ingram (1995).

Body-mass estimates

In most cases, body mass estimates for birds were compiled from Dunning (1993) and estimates for mammals from Silva and Downing (1995). Where possible, body mass estimates from the region of interest were used, otherwise estimates were taken from the nearest geographic location. Male and female body mass estimates were averaged when weights for both sexes were provided. The body sizes were recorded in grams and subsequently logarithmically transformed.

Species Criteria

A species was considered a non-indigenous species (NIS) to an ecosystem of interest if it was introduced, intentionally or inadvertently. Species introduced after the year 2006 and only species with established breeding populations were included in analyses. Endangered species were classified as those species listed as either extinct, endangered, a species of special concern, vulnerable or threatened. A list of NIS and endangered species was created utilizing data from the 2006 International Union for

Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species (IUCN 2006), but data quality varies by region and taxa.

Discontinuity Analysis

Body mass pattern or structure refers to the distribution of body mass aggregations and gaps along the body mass axis. Body mass distributions were analyzed for discontinuities using simulations of actual data compared to a null distribution (a continuous unimodal kernel distribution of the log-transformed data (Restrepo et al. 1997). A body-mass aggregation consisted of three or more species with body masses that did not exceed the expectation of the null distribution and was defined by the upper and lower extremes of the aggregation (Allen et al. 1999). Gaps in body-mass aggregations were defined as significantly large areas between adjacent body masses that exceeded the expectation of the null distribution (Forys and Allen 2002). Significance of discontinuities in the data was determined by calculating the probability that the observed discontinuities were chance events (compared observed values with output of 1,000 simulations run against the null set (Restrepo et al. 1997).

The number of species in my data sets vary from < 30 to > 150, therefore I maintained a constant statistical power of approximately 0.50 when setting alpha for detecting discontinuities (Lipsey 1990). Two other methods were also used to confirm the location of discontinuities. I used Bayesian Classification and Regression Trees (Chipman et al. 1998) and hierarchical cluster analysis (SAS Institute 1999) to further validate any patterns detected in body mass distributions. A multiple method approach in detecting significance in body mass patterns has been suggested as the best protocol (Allen et al. 2006, Stow et al. 2007). Visual examination of juxtaposition of pairs of

Mediterranean-climate ecosystem body mass distributions was also used to determine similarities in body mass aggregation patterns. Abundance data for each individual species was not integrated into my analyses. Changes in dominance of species (greater numbers of individuals) could change aggregation and gap locations, but body mass distribution patterns will be conserved despite changes in species composition or number (Havlicek & Carpenter 2001).

Chi-square and Phi correlation analyses were used to determine differences in body mass patterns between Mediterranean-climate ecosystems. The null hypothesis of the Chi-square analyses is that there are equal distributions among each of the four cells in a 2 x 2 table (one in the gap condition while the other is in the lump condition and vice versa, both in the lump condition and both in the gap condition), which would indicate that the two groups are unrelated. A Bonferroni correction was applied to account for multiple comparisons in the same analysis (Miller 1981). In order to provide additional information, a correlation of binary variables, using Phi correlation, was also utilized in order to elucidate more information and to determine the relationship between the Mediterranean-climate ecosystems. Species within each Mediterranean-climate ecosystems were assigned a binary variable (where 1 represented a body mass aggregation and 0 represented a gap), based on the log₁₀ body mass axis, and divided into a 0.01 gram increment bin. The body mass axis range was limited by the shared data between the two groups. The Phi coefficient ranges from -1 to 1. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Fleiss et al. (2003) suggested that values greater than ± 0.30 indicate a strong relationship.

NIS & Endangered Species Analysis

After completing these analyses, the location of NIS body masses within these distributions was determined. NIS and endangered species could be distributed in the body-mass patterns in various ways; randomly, within a limited range of body masses, at the edge of aggregations, invasives only within body mass aggregations or invasives only in the gaps (Figure 2.1). Both gaps and edges of body mass distributions represent changes in scale and areas of high variability (see Allen et al. 1999). I specifically tested whether NIS and endangered species occurred in gaps between body mass aggregations in the observed distribution more often than expected values if their distribution were random using a chi-square goodness of fit test.

T-tests were used to determine the distance to body mass aggregation edge of native endangered species compared to native non-endangered species and the distribution of NIS in terms of distance to the nearest edge in the observed distribution when compared to the native species distances from edges. Taxonomic groups were also analyzed separately for both comparisons. T-tests were also used to compare the distance to body mass aggregation edge of native and NIS, with NIS distance to body mass aggregation edge when gap spaces are set to zero. Gap space was set to zero in order to observe the results when you assume all NIS are actually on the edge.

RESULTS

Discontinuities

All ten Mediterranean-climate bird and mammal body-mass distributions were discontinuous. Unique aggregations of species were detected in each taxa, by all methods. I observed a range of 4-12 discontinuities in my datasets. There were a total of

57 endangered bird species and 52 endangered mammal species and a total of 54 invasive bird species (74.1% within body mass aggregations) and 41 invasive mammal species (48.8% within body mass aggregations) (Table 2.1). Bird communities had between 6 – 12 body mass aggregations and mammals had between 4 – 8 body mass aggregations in each community. Bird communities had a smaller average gap size ($x = 0.097 \pm 0.09$) than mammals ($x = 0.322 \pm 0.166$). Visual examination of juxtaposition of pairs of Mediterranean-climate ecosystem body mass distributions revealed similar overlapping patterns at some scales, but not at every scale, in either birds (Figure 2.2) or mammals (Figure 2.3).

Lump/Gap Structure

Mammals

Five of 10 pairwise comparisons between mammal body mass aggregations in Mediterranean-climate ecosystems were significant in the chi-square analysis and 5 of 10 pairwise comparisons were significant in the Phi correlation analysis (Tables 2.2 & 2.3). The pairwise comparisons between California and South Africa, Chile and southwestern Australia, Spain and South Africa, Spain and southwestern Australia, and South Africa and southwestern Australia indicated no similarity in body mass patterns. All other Mediterranean-climate ecosystem body mass patterns were similar to each other. The Phi coefficients between California and Chile, California and Spain, California and southwestern Australia, Chile and Spain, and Chile and South Africa indicated strong positive relationships. The Phi coefficients between Chile and southwestern Australia and between Spain and southwestern Australia indicated positive relationships, but the relationships were weak. The Phi coefficients between the remaining pairwise

comparisons of Mediterranean climate ecosystems indicated negative relationships, but the relationships were weak.

Birds

Two of 10 pairwise comparisons between bird body mass aggregations in Mediterranean-climate ecosystems were significant in the chi-square analysis and one of 10 pairwise comparisons were significant in the Phi correlation analysis (Tables 2.4 & 2.5). The pairwise comparisons between California and Chile and California and South Africa indicated similar body mass patterns. All other pairwise comparisons of Mediterranean-climate ecosystem body mass patterns were not similar to each other. The Phi coefficient between California and South Africa was the only comparison that indicated a strong positive relationship. The Phi coefficients between California and Chile, California and Spain, and Chile and southwestern Australia indicated positive relationships, but the relationships were weak. The Phi coefficients between the remaining pairwise comparisons of Mediterranean climate ecosystems indicated negative relationships, but the relationships were weak.

NIS and Endangered Species

The distance to edge of a body mass aggregation for native endangered species was significantly less than for non-endangered species in Australia mammals ($z=1.345$, $P=0.09$) and South Africa birds ($z=1.56$, $P=0.06$). The distance to edge of a body mass aggregation for native non-endangered species was significantly less than for endangered species in Spain mammals ($z=2.66$, $P=0.004$) and Australia birds ($z=1.820$, $P=0.03$) (Table 2.6). The distance to edge of a body mass aggregation for natives was significantly less than for NIS in California birds ($z=1.3$, $P=0.10$), Chile birds ($z=1.13$, $P=0.10$) and

Spain mammals ($z=1.269$, $P=0.10$). The distance to edge of a body mass aggregation for NIS was significantly less than for natives in Spain birds ($z=-1.745$, $P=0.04$) (Table 2.7). Pooling data across ecosystems, the distance to edge of a body mass aggregation for all native non-endangered was significantly less than for endangered species in birds ($z=1.565$, $P=0.06$) and the distance to edge of a body mass aggregation for all native was significantly less than for NIS, in mammals ($z=1.869$, $P=0.03$) (Table 2.8).

T-tests comparing distance to edge of a body mass aggregation for native and non-native species with NIS species in gaps set to zero were significant in California mammals ($z=-1.41$, $P=0.08$), Spain birds ($z=-2.199$, $P=0.01$) and mammals ($z=-1.085$, $P=0.10$), South Africa birds ($z=-1.352$, $P=0.09$), Australia mammals ($z=-1.332$, $P=0.09$) and Chile mammals ($z=-1.915$, $P=0.03$). Pooling data across ecosystems, t-tests comparing distance to edge of a body mass aggregation for all native and NIS, with NIS species in gaps set to zero, were significant in both taxonomic groups (birds, $P = 0.06$; mammals, $P = 0.004$) (Table 2.9).

NIS birds ($X^2=29.91$, 4 df; $P = 5.11 \times 10^{-6}$) and mammals ($X^2=34.25$, 4 df; $P = 6.63 \times 10^{-7}$) occurred in gaps between body mass aggregations in the observed distribution more often than expected (Table 2.10). Based on calculated gap space, NIS birds ($X^2=2.68$, 4 df; $P = 0.61$) and mammals ($X^2=3.67$, 4 df; $P = 0.45$) did not occur in gaps between body mass aggregations in the observed distribution more often than expected. Declining species of birds ($X^2=31.32$, 4 df; $P = 2.63 \times 10^{-6}$) and mammals ($X^2=8.80$, 4 df; $P = 0.07$) occurred at body mass aggregation edges in the observed distribution more often than expected (Table 2.11). Based on calculated gap space, declining species of birds

($X^2=6.88$, 4 *df*; $P = 0.14$) and mammals ($X^2=1.28$, 4 *df*; $P = 0.86$) did not occur at body mass aggregation edges in the observed distribution more often than expected.

DISCUSSION

Discontinuous body-mass distributions were found in all Mediterranean-climate ecosystems and taxa examined. Discontinuities in body mass distributions have also been shown in North American birds (Skillen and Maurer 2008), south Florida herpetofauna, birds, and mammals (Allen et al. 1999, Allen 2006), Pleistocene and Miocene mammals (Lambert and Holling 1998, Lambert 2006), tropical forest birds (Restrepo et al. 1997), and boreal region birds and mammals (Holling 1992) and in various other taxa (reviewed in Sendzimir et al. 2003). This analysis extends these conclusions to include animal communities across Mediterranean-climate ecosystems of the world.

Body mass distribution patterns observed using the statistical methods employed in this dissertation are real and are not an apparent effect of random noise. Sendzimir (1998) ran simulations to determine differences in observed body mass distributions compared to simulated body mass distribution patterns. Aggregation pattern recognition did decline below robust levels in the 10 to 15 percent (mammals) and 5 percent level (birds) of error due to random variation in body size in mammals. This decline in aggregation pattern recognition was evident across all body sizes in both mammals and birds.

Bird communities had more aggregations than mammal communities and mammal communities had larger gap sizes than birds. These differences may be due to how each taxa perceive and interact with landscape structure. It has been suggested that

birds live in a more three dimensional world and mammals live in a one dimensional world (Holling 1992). If this were the case, birds would be able to access a greater number of resources enabling the use of a more complex landscape structure and allow birds to occupy more niches. More niches could account for a greater number of aggregations, which would contradict the textural discontinuity hypothesis. TDH would posit that as a landscape becomes more structurally complex, the opportunity to fill new niches increases, not vice versa. Birds and mammals also have different locomotory modes, affording birds the ability to utilize both terrestrial and aerial resources in any given landscape (Sendzimir 1998).

Mediterranean-climate ecosystems are geographically isolated regions that have long been considered to be structurally similar and have similar climatic regimes. However, more in-depth research of ecological processes within these ecosystems suggests that these regions are examples of both convergence and divergence (reviewed in Rundel 2011). Only 35% of all pairwise comparisons between the structures of body mass distributions in five Mediterranean-climate ecosystems were determined to be similar. These results may be due to inherent differences in soil fertility, geology, differential rainfall, topographic heterogeneity, climatic heterogeneity, and different disturbance regimes (i.e., fire frequency) among these Mediterranean-climate ecosystems (Di Castri 1991, reviewed in Cowling et al. 1996).

Differences in human transformation of these Mediterranean-climate ecosystems might also explain the lack of similarity in all pairwise comparisons between ecosystems. The Mediterranean Basin (i.e., Spain in this analysis) has the oldest history of human occupation (i.e., agriculture and animal husbandry) dating back 10,000 years ago (Naveh

& Kutiel 1990). The other regions were inhabited by hunter-gatherers and pastoralists until colonization which subsequently brought European grains and livestock to these regions (Aschmann 1973). Differences in time since settlement, cultural differences between the colonizing countries, and other regional differences (i.e., availability of resources and relationships between colonizers and indigenous people) may have contributed to differences in current patterns of human transformation between Mediterranean-climate ecosystems, thus resulting in differences in landscape structure between ecosystems (Hobbs et al. 1995). Differences in present and projected human population sizes, growth rates, and densities between Mediterranean-climate ecosystems have major impacts on land use, resulting in different patterns of landscape transformation (Mooney et al. 2001). Thus, body mass distributions within these ecosystems will never be the same in all pairwise comparisons. My results may also be an artifact of the type of statistical analysis applied to the data. Chi-square and Phi correlation analyses may not have been appropriate; however, no other statistical analyses for this type of comparison have ever been documented.

More NIS and endangered species were found to occur at the edges of body mass aggregations than could be expected by chance alone in 30% of datasets, thus supporting similar analyses which examined the distribution of NIS and endangered species in relation to body mass aggregations (Allen et al. 1999). Skillen and Maurer (2008) reported an average of 52% of declining species were closer to body mass aggregation edges than the median distance to the nearest body mass aggregation edge and showed an average of 72% of NIS were further away from body mass aggregation edges than the median distance (i.e., located in gaps). In those regions where invasive and endangered

species were not found near the edges of body mass aggregations, plausible factors may be degree of human alteration or influence, species assemblages transitioning from a recent environmental change, or a delayed response to a prior change in the environment (Skillen and Maurer 2008). Body mass aggregation edges and gaps between body mass aggregations or scale breaks have been referred to as “zones of crisis and opportunity” (Allen et al. 1999) depending on whether the species is an endangered species or an invader.

The results of this study clearly have significant implications for the field of landscape ecology and conservation biology. Humans continually altering the landscape at an increasing rate, current patterns of global change (i.e., climate change and globalization), and an increased rate of invasions (Vitousek et al. 1996, Mooney and Cleland 2001, Lodge and Shrader-Frechette 2003), will continue to change the composition of animal communities both locally, regionally and globally. My results may help our ability to maintain ecosystem resilience by making the proper management decisions in monitoring particular non-indigenous species (those closest to body mass aggregation edges and in gaps) and focus conservation efforts on those native species nearest body mass aggregation edges (i.e., the endangered ones).

Table 2.1. Overall number of native species in five Mediterranean-climate ecosystems, the overall percentage of endangered and non-indigenous species (NIS), the percentage of NIS species in each region after a hypothetical removal of threatened species, the number of discontinuities and the percentage of NIS within body mass aggregations.

Region	Number of			% NIS After Extinction of Threatened Species	Number of Discontinuities (birds, mammals)	% of NIS within body mass aggregations
	Native Bird and Mammal Species	% Endangered	% NIS			
California	148	12.8	13.5	15.5	9,5	70
Chile	150	3.3	13.3	13.8	9,4	65
Spain	165	20	10.3	12.9	6,8	59
South Africa	157	8.3	11.5	12.5	7,7	61
Southwestern Australia	188	16.5	10.6	12.7	12,5	60

Table 2.2. Chi-Square probability levels for each pairwise comparison of Mediterranean-climate ecosystem mammal body mass aggregations. Probability levels less than or equal to 0.005 were considered significant matches between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	<.0001*	<.0001*	0.0326	<.0001*
Chile	---	<.0001*	<.0001*	0.4484
Spain		---	0.5239	0.4445
South Africa			---	0.3378

Table 2.3. Phi coefficients for each pairwise comparison of Mediterranean-climate ecosystem mammal body mass aggregations. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	0.2895*	0.3932*	-0.1016	0.4488*
Chile	---	0.3790*	0.4681*	0.0486
Spain		---	-0.0301	0.0455
South Africa			---	-0.0519

Table 2.4. Chi-Square probability levels for each pairwise comparison of Mediterranean-climate ecosystem bird body mass aggregations. Probability levels less than or equal to 0.005 were considered significant matches between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	0.0023*	0.1059	<.0001*	0.1184
Chile	---	0.0566	0.9393	0.9634
Spain		---	0.2529	0.2389
South Africa			---	0.0235

Table 2.5. Phi coefficients for each pairwise comparison of Mediterranean-climate ecosystem bird body mass aggregations. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared ecosystems.

	South			Southwestern
	Chile	Spain	Africa	Australia
California	0.1771	0.0940	0.5017*	-0.0860
Chile	---	0.1143	-0.0046	0.0027
Spain		---	-0.0692	-0.0682
South Africa			---	-0.1366

Table 2.6. Results of t-tests comparing mean (x) and standard deviation (SD) distance to edge of body mass aggregation of endangered and non-endangered bird and mammal species in five Mediterranean-climate ecosystems.

Region	Non-Endangered	Endangered	Birds		Non-Endangered	Endangered	Mammals	
	x(SD)	x(SD)	z	P	x(SD)	x(SD)	z	P
California	0.071(.069)	0.083(.113)	0.985	0.160	0.099(.106)	0.12(.085)	0.540	0.290
Chile	0.054(.044)	0.075(.030)	0.774	0.220	0.15(.151)	0.078(.135)	-1.050	0.150
Spain	0.144(.132)	0.143(.115)	0.290	0.390	0.045(.056)	0.080(.054)	2.660	0.004*
South Africa	0.061(.063)	0.040(.080)	1.560	0.060*	0.118(.121)	0.130(.119)	0.389	0.350
SWAustralia	0.046(.053)	0.134(.214)	1.820	0.030*	0.171(.135)	0.119(.100)	1.345	0.090*

Table 2.7. Results of t-tests comparing mean (x) and standard deviation (SD) distance to edge of body mass aggregation of native and non-indigenous (NIS) bird and mammal species in five Mediterranean-climate ecosystems.

Region	Native	NIS	Birds		Native	NIS	Mammals	
	x(SD)	x(SD)	z	P	x(SD)	x(SD)	z	P
California	0.073(.077)	0.087(.067)	1.300	0.100*	0.101(.103)	0.118(.074)	0.515	0.300
Chile	0.054(.043)	0.081(.062)	1.130	0.100*	0.143(.149)	0.115(.089)	0.244	0.400
Spain	0.144(.128)	0.072(.105)	1.745	0.040*	0.051(.057)	0.073(.051)	1.269	0.100*
South Africa	0.059(.064)	0.048(.034)	0.222	0.410	0.120(.120)	0.136(.093)	0.830	0.200
SWAustralia	0.051(.076)	0.053(.069)	0.180	0.430	0.140(.116)	0.206(.171)	0.993	0.160

Table 2.8. Results of t-tests comparing mean (x) and standard deviation (SD) distance to edge of body mass aggregation of all endangered and non-endangered bird and mammal species and native compared to non-indigenous (NIS) bird and mammal species, pooling five Mediterranean-climate ecosystems.

Taxonomic Group	Non-Endangered							
	Non-Endangered	Endangered	Endangered	Native	NIS	Native/NIS		
	x(SD)	x(SD)	z	P	x(SD)	x(SD)	z	P
Birds	0.072(.083)	0.113(.131)	1.565	0.1	0.076(.090)	0.069(.070)	0.398	0.350
Mammals	0.109(.120)	0.113(.097)	0.729	0.5	0.11(.115)	0.13(.108)	1.869	0.030*

Table 2.9. Results of t-tests comparing mean (x) and standard deviation (SD) distance to edge of body mass aggregation of native and non-indigenous (NIS) bird and mammal species with NIS species in gaps set to zero in five Mediterranean-climate ecosystems.

Region	Native	NIS	Birds		Native	NIS	Mammals	
	x(SD)	x(SD)	z	P	x(SD)	x(SD)	z	P
California	0.073(.077)	0.079(.073)	0.478	0.316	0.101(.103)	0.053(.091)	-1.41	0.080*
Chile	0.054(.043)	0.077(.067)	0.759	0.224	0.143(.149)	0.046(.066)	1.915	0.030*
Spain	0.144(.128)	0.065(.109)	2.199	0.010*	0.051(.057)	0.030(.062)	1.085	0.100*
South Africa	0.059(.064)	0.033(.043)	1.352	0.090*	0.120(.120)	0.108(.117)	0.333	0.370
SWAustralia	0.051(.076)	0.046(.072)	0.919	0.180	0.140(.116)	0.079(.107)	1.332	0.090*
Combined	0.076(.090)	0.060(.075)	1.523	0.060*	0.11(.115)	0.065(.091)	2.695	0.004*

Table 2.10. Results of Chi-Square analyses comparing the observed distribution of non-indigenous birds and mammals in gaps between body mass aggregations in five Mediterranean-climate ecosystems.

Region	Birds		Mammals	
	Expected	Observed	Expected	Observed
California	1.218	2	1.03	4
Chile	0.656	1	2	6
Spain	0.583	3	1.34	4
South Africa	0.87	4	0.955	3
SW Australia	1.076	4	1.07	4
X ²	29.91(5.11E-06)*		34.25(6.63E-07)*	

Table 2.11. Results of Chi-Square analyses comparing the observed distribution of endangered birds and mammals in gaps between body mass aggregations in five Mediterranean-climate ecosystems.

	Birds		Mammals	
Region	Expected	Observed	Expected	Observed
California	1.73	7	0.69	1
Chile	0.18	0	0.50	2
Spain	1.55	1	1.53	2
South Africa	0.29	2	1.31	3
SW Australia	0.89	0	3.47	6
X^2	31.32 (2.63E-06)*		8.80(0.07)*	

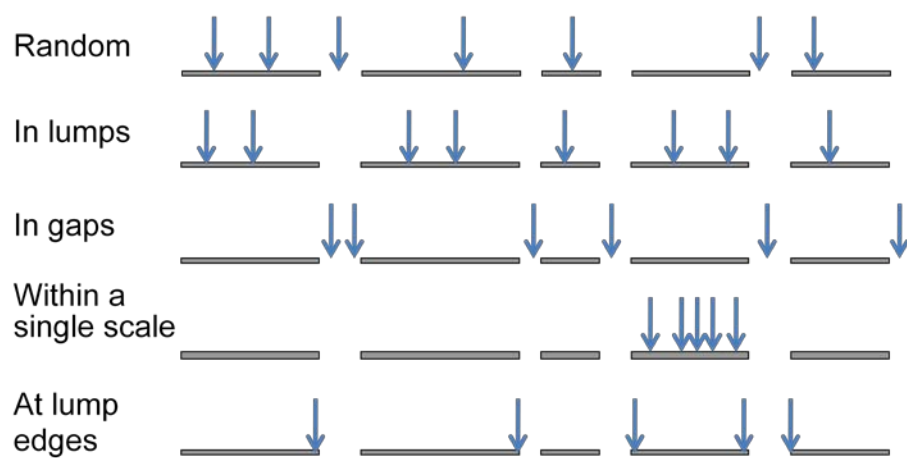


Figure 2.1. Potential patterns of where invasive and endangered species could be distributed in the context of vertebrate body mass distributions. Each pattern supports a different hypothesis.

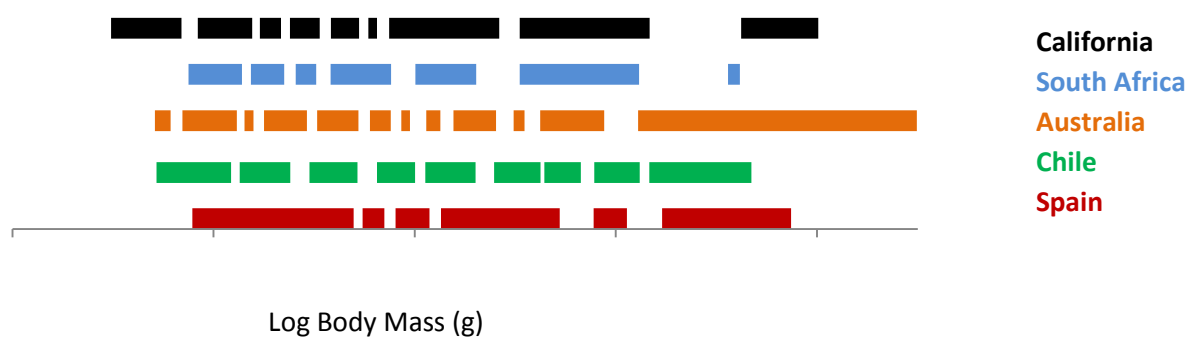


Figure 2.2. Juxtaposition of bird body mass aggregations across five Mediterranean-climate ecosystems.

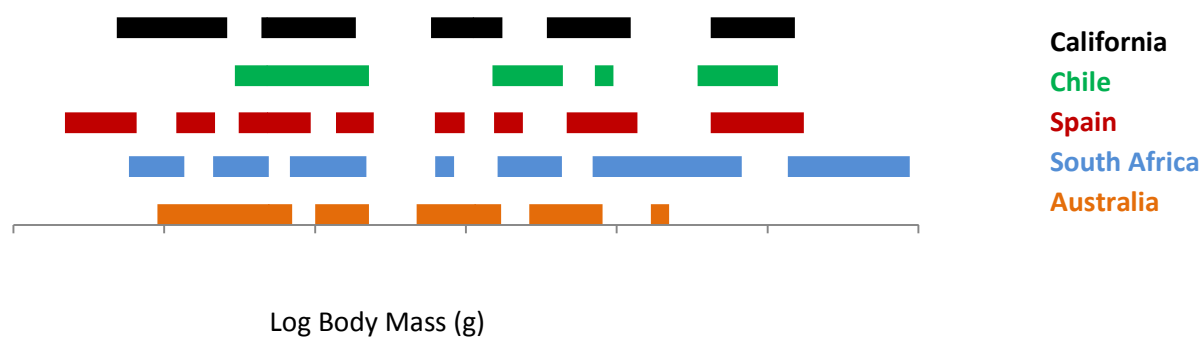


Figure 2.3. Juxtaposition of mammal body mass aggregations across five Mediterranean-climate ecosystems.

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CHAPTER 3: CHANGES IN ALPHA, BETA & GAMMA DIVERSITY OF FUNCTION IN MEDITERRANEAN-CLIMATE ECOSYSTEMS

Climate change, habitat loss, habitat degradation and the introduction of non-indigenous species (NIS) are just some of the factors that have contributed to an estimated three orders of magnitude increase in the rate of species extinction over the past few hundred years and significant changes in biodiversity worldwide (Millennium Ecosystem Assessment 2005). Biodiversity is the composition, structure and function of an ecosystem (Noss 1990). Therefore, biological diversity must be treated more seriously as a global resource to be indexed, used and preserved (Wilson 1988).

Landscapes form hierarchies (which contain breaks in object proximities, and textures at particular scales and sizes of objects) that are structured by vegetative, geomorphologic and contagious disturbance processes (Holling 1992). The spatial and temporal patterns inherent in landscapes reflect numerous processes, interacting on distinct scales, which potentially shape the assembly of animal communities. Many studies have reported a discontinuous body mass distribution reflecting the ecological structure of the landscape (see Allen & Holling 2008). These studies support the textural discontinuity hypothesis (TDH), which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape (Holling 1992).

Organisms of different body sizes have different requirements for resources and operate at different spatiotemporal scales (West et al. 1997). Therefore, an animal cannot simultaneously interact with multiple scales, but has to specialize at a single scale or shift

between two scales (Allen & Saunders 2002). Allen et al. (1999) suggest that the gaps between body mass aggregations represent scale breaks and that these points in the body mass distribution are highly susceptible to change in structure and ecological processes. Analysis of body-mass aggregations may provide insight into functional group change within and across scales (i.e., cross-scale resilience) following invasions and extinctions (Forys & Allen 2002). In order to test Holling's (1992) textural discontinuity hypothesis, I determined whether discontinuous body mass patterns existed in each of the five Mediterranean-climate ecosystems. A lack of discontinuous body mass patterns would provide strong evidence against the textural discontinuity hypothesis.

Following landscape change and native species decline, NIS may become established and can affect ecosystem processes at varying scales and may lead to the further extinction of native species (Williamson 1996, Vitousek et al. 1997). Thus, NIS may alter; 1) alpha diversity, which refers to the diversity within a particular ecosystem and is expressed by the number of species in that ecosystem, 2) beta diversity, which refers to the diversity between two ecosystems and is expressed by the total number of species unique to each of the ecosystems being compared and 3) gamma diversity, which is the overall diversity in all ecosystems within a particular region (Whittaker 1972, Balée 2006, Fridley et al. 2007).

In a recent review of the Convention on Biological Diversity's 2020 goals, authors suggest that an assessment of functional diversity is missing. In order to realize the benefits of ecosystem services, which are derived from ecosystem functions, the species that perform those functions must be conserved (Perrings et al. 2010). However, species are not equal in their effects on ecosystem functioning (Mouchet et al. 2010).

Function typically refers to ecological functions, such as pollination, seed dispersal, predation or grazing. Functional diversity is based on what organisms do, not on their evolutionary history (reviewed in Petchey & Gaston 2006). Functional diversity is an important determinant of ecosystem processes and is defined as the number of functional groups represented by the species in a community (reviewed in Petchey & Gaston 2002).

The potential loss of native species richness may disrupt ecological processes that inherently shape landscape structure, such as predator-prey dynamics, seed dispersal, hydrology, nutrient cycling and fire regime. Most studies have examined the effects of invasive species on species richness, not functional group diversity (reviewed in Vitousek et al. 1996). Few studies analyze functional diversity in relation to invasions (Forys & Allen 2002, Hooper and Dukes 2010). Peterson et al. (1998) suggest ecological resilience is generated more by functional diversity than by species richness and that redundancy of functional groups across scales increases cross-scale resilience.

Mediterranean-climate ecosystems are among the most diverse and threatened in the world (Malcolm et al. 2006). Mediterranean-climate ecosystems are projected to incur the highest loss of biodiversity of all terrestrial ecosystems by the year 2100 (Sala et al. 2000) with South Africa and southwestern Australia expected to undergo the most significant impact (Klausmeyer & Shaw 2009). These regions are also densely populated by humans and have been altered by human habitation and landscape transformation. Mediterranean-climate regions support large human populations, resulting in extensive, rapid, anthropogenic transformation. In a study examining key threats and trends in Mediterranean-climate ecosystems, the California-Baja California region had the highest population density and highest percent urban area of any of the five Mediterranean

regions examined, while southwestern Australia had the highest percentage of high intensity agriculture (Underwood et al. 2009). Mediterranean-climate ecosystems provide a unique data set because they are highly invaded communities whose non-indigenous fauna has been well-documented. They have been invaded by numerous non-indigenous vertebrate species and declines and extinctions of native species have transformed the faunal community composition in these regions (Malcolm et al. 2006). Here, I investigate the effects of invasive species and the loss of endangered species on functional group composition. Specifically, how these changes affect biodiversity and functional group representation within and across body mass aggregations at various scales.

METHODS

Study Areas

I focused on central coastal Chile, the Western Cape Province of South Africa, San Diego County, California, the South West Botanical Province in western Australia, and Spain. Delineation of Mediterranean-climate regions varies by author, especially whether one considers primarily climatic or vegetative boundaries, therefore multiple sources were used to gain a consensus on the extent of Mediterranean-climate habitat in each region (Akin 1991, Di Castri 1991, Hobbs et al. 1995). In general, these ecosystems are characterized by winter rains with a pronounced dry-season and frequent fires (Hobbs et al. 1995). Vegetation is mainly grassland and brush including Chaparral (California), Mattoral (Chile), Maquis (Spain), Fynbos (South Africa) and Mallee or Kwongan (Australia). Although the entire Mediterranean-climate ecosystem in California and Australia were not sampled, body mass distribution analysis is robust to approximately

18% (mammals) and 15% (birds) omission error (Sendzimir 1998). Therefore, my results are limited in scope to the areas that I selected.

Native species

I examined two terrestrial vertebrate groups, birds and mammals. Species lists for Mediterranean-climate ecosystems were obtained from published sources. Californian mammals were determined from a specific study of their habitat preferences and distribution (Quinn 1990). Unitt (2004) provided a thorough list of the birds of San Diego County. Vuilleumier (1991) provided a list of Chilean birds and Chilean mammals were provided by Miller (1981). Spanish mammals were determined from Cheylan (1991). Spanish birds were compiled from the comprehensive works of Cramp (1978 – 1994). Mammals residing in the Mediterranean-climate Fynbos of the southern cape of South Africa were determined from Smithers (1983). Winterbottom (1966) provided data on the avifauna of Mediterranean shrublands in South Africa. Mammal data for Australia were compiled from Strahan (1995). Australian birds were determined from Saunders and Ingram (1995).

Body-mass estimates

In most cases, body mass estimates for birds were compiled from Dunning (1993) and estimates for mammals from Silva and Downing (1995). Where possible, body mass estimates from the region of interest were used, otherwise estimates were taken from the nearest geographic location. Male and female body mass estimates were averaged when weights for both sexes were provided. The body sizes were recorded in grams and subsequently logarithmically transformed.

Species Criteria

A species was considered NIS to an ecosystem of interest if it was introduced, intentionally or inadvertently. Species introduced after the year 2006 and only species with established breeding populations were included in analyses. Endangered species were classified as those species listed as either extinct, endangered, a species of special concern, vulnerable or threatened. A list of invasive and endangered species was created utilizing data from the 2006 International Union for Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species (IUCN 2006), but data quality varies by region and taxa.

Discontinuity Analysis

Body mass pattern or structure refers to the distribution of body mass aggregations and gaps along the body mass axis. Body mass distributions were analyzed for discontinuities using simulations of actual data compared to a null distribution (a continuous unimodal kernel distribution of the log-transformed data (Restrepo et al. 1997). A body-mass aggregation consisted of three or more species with body masses that did not exceed the expectation of the null distribution and was defined by the upper and lower extremes of the aggregation (Allen et al. 1999). Gaps in body-mass aggregations were defined as significantly large areas between adjacent body masses that exceeded the expectation of the null distribution (Forys and Allen 2002). Significance of discontinuities in the data was determined by calculating the probability that the observed discontinuities were chance events (compared observed values with output of 1,000 simulations run against the null set (Restrepo et al. 1997).

The number of species in my data sets vary from < 30 to > 150 , therefore I maintained a constant statistical power of approximately 0.50 when setting alpha for

detecting discontinuities (Lipsey 1990). Two other methods were also used to confirm the location of discontinuities. I used Bayesian Classification and Regression Trees (Chipman et al. 1998) and hierarchical cluster analysis (SAS Institute 1999) to further validate any patterns detected in body mass distributions. A multiple method approach in detecting significance in body mass patterns has been suggested as the best protocol (Allen et al. 2006, Stow et al. 2007). Abundance data for each individual species was not integrated into my analyses. Changes in dominance of species (greater numbers of individuals) could change aggregation and gap locations, but body mass distribution patterns will be conserved despite changes in species composition or number (Havlicek & Carpenter 2001).

Chi-square and Phi correlation analyses were used to determine differences in body mass patterns between Mediterranean-climate ecosystems. The null hypothesis of the Chi-square analyses is that there are equal distributions among each of the four cells in a 2 x 2 table (one in the gap condition while the other is in the lump condition and vice versa, both in the lump condition and both in the gap condition), which would indicate that the two groups are unrelated. A Bonferroni correction was applied to account for multiple comparisons in the same analysis (Miller 1981). In order to provide additional information, a correlation of binary variables, using Phi correlation, was also utilized in order to elucidate more information and to determine the relationship between the Mediterranean-climate ecosystems. Species within each Mediterranean-climate ecosystems were assigned a binary variable (where 1 represented a body mass aggregation and 0 represented a gap), based on the log₁₀ body mass axis, and divided into a 0.01 gram increment bin. The body mass axis range was limited by the shared data

between the two groups. The Phi coefficient ranges from -1 to 1. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Fleiss et al. (2003) suggested that values greater than ± 0.30 indicate a strong relationship.

Functional Diversity Analysis

Species richness has low explanatory power (i.e., does not identify the unique ecological role of each species) and should not be used as a measure of functional diversity (Petchey et al. 2004). However, there is a positive correlation between species richness and functional diversity (Tilman et al. 1996). I constructed functional group classifications using both diet and foraging strata, based on published species accounts (Table 3.1). Functional diversity, or functional group richness, was calculated as the number of diet/foraging strata present in each body mass aggregation. Functional diversity change across scales (i.e., alpha, beta and gamma diversity of function) was determined using the Shannon Index H' (Shannon and Weaver 1949). In order to isolate the effects of NIS on an ecosystem and predict their future impact, a “preinvasion” species list included all native species, all species listed as endangered and native species that are extinct. A “postinvasion” species list included native species and NIS with established breeding populations. This list assumed that all endangered species will go extinct in the near future; therefore currently endangered species were omitted.

Shannon Index values were used to compare alpha, beta and gamma diversity of function in each taxa; 1) original community composition contrasted with the addition of NIS, 2) original community composition contrasted with a postinvasion community, and 3) original community composition with NIS contrasted with a postinvasion community. T-tests were used to compare functional group richness preinvasion versus postinvasion,

by taxa and when combined. In order to determine any changes in the redundancy in functional groups across scales, I used t-tests to compare the number of body mass aggregations where at least one member of a functional group was present preinvasion to the number of aggregations where it occurred postinvasion (Forys and Allen 2002). Taxonomic groups were analyzed separately.

RESULTS

Discontinuities

All ten Mediterranean-climate bird and mammal body-mass distributions were discontinuous. Unique aggregations of species were detected in each taxa, by all methods. I observed a range of 4-12 discontinuities in my datasets. There were a total of 57 endangered bird species and 52 endangered mammal species and a total of 54 invasive bird species and 41 invasive mammal species (Table 3.2). Visual examination of juxtaposition of pairs of Mediterranean-climate ecosystem body mass distributions revealed similar overlapping patterns at some scales, but not at every scale, in either birds (Figure 3.1) or mammals (Figure 3.2).

Lump/Gap Structure

Mammals

Five of 10 pairwise comparisons between mammal body mass aggregations in Mediterranean-climate ecosystems were significant in the chi-square analysis and 5 of 10 pairwise comparisons were significant in the Phi correlation analysis (Tables 3.3 & 3.4). The pairwise comparisons between California and South Africa, Chile and southwestern Australia, Spain and South Africa, Spain and southwestern Australia, and South Africa

and southwestern Australia indicated no similarity in body mass patterns. All other Mediterranean-climate ecosystem body mass patterns were similar to each other. The Phi coefficients between California and Chile, California and Spain, California and southwestern Australia, Chile and Spain, and Chile and South Africa indicated strong positive relationships. The Phi coefficients between Chile and southwestern Australia and between Spain and southwestern Australia indicated positive relationships, but the relationships were weak. The Phi coefficients between the remaining pairwise comparisons of Mediterranean climate ecosystems indicated negative relationships, but the relationships were weak.

Birds

Two of 10 pairwise comparisons between bird body mass aggregations in Mediterranean-climate ecosystems were significant in the chi-square analysis and one of 10 pairwise comparisons were significant in the Phi correlation analysis (Tables 3.5 & 3.6). The pairwise comparisons between California and Chile and California and South Africa indicated similar body mass patterns. All other pairwise comparisons of Mediterranean-climate ecosystem body mass patterns were not similar to each other. The Phi coefficient between California and South Africa was the only comparison that indicated a strong positive relationship. The Phi coefficients between California and Chile, California and Spain, and Chile and southwestern Australia indicated positive relationships, but the relationships were weak. The Phi coefficients between the remaining pairwise comparisons of Mediterranean climate ecosystems indicated negative relationships, but the relationships were weak.

Functional Diversity

California - Historically, there were 144 native bird and mammal species in the Mediterranean-climate ecosystem of California. Currently, approximately 13% of these species are endangered and 14% are NIS. If all currently listed species go extinct and all of the currently established NIS persist, 16% of the fauna will be non-native. Alpha diversity of function in birds and mammals increased with the inclusion of NIS and increased further after endangered species were removed (Table 3.7). Postinvasion, the bird community gained three new functional groups (terrestrial herbivore, aquatic omnivore, and foliage omnivore) and lost one (aquatic herbivore). The mammal community gained one new functional group (aquatic herbivore).

Chile - Historically, there were 150 bird and mammal species in the Mediterranean-climate ecosystem of Chile. Currently, approximately 3% of these species are endangered and 13% are NIS. If all currently listed species go extinct and all of the currently established NIS persist, 14% of the fauna will be non-native. Alpha diversity of function in birds increased with the inclusion of NIS and decreased below the diversity level of the original community after endangered species were removed. Alpha diversity of function in mammals increased with the inclusion of NIS and increased further after endangered species were removed (Table 3.7). Postinvasion, the bird community did not gain or lose any functional groups. The mammal community gained one new functional group (terrestrial omnivore).

Spain - Historically, there were 170 bird and mammal species in the Mediterranean-climate ecosystem of Spain. Currently, approximately 19% of these species are endangered and 10% are NIS. If all currently listed species go extinct and all of the currently established NIS persist, 7% of the fauna will be non-native. Alpha

diversity of function in birds increased with the inclusion of NIS and increased further after endangered species were removed. Alpha diversity of function in mammals decreased with the inclusion of NIS and decreased further after endangered species were removed (Table 3.7). Postinvasion, the bird community gained three new functional groups (aquatic granivore, foliage herbivore and aquatic omnivore). The mammal community lost two functional groups (aquatic carnivore and aquatic herbivore).

South Africa - Historically, there were 156 bird and mammal species in the Mediterranean-climate ecosystem of South Africa. Currently, approximately 8% of these species are endangered and 12% are NIS. If all currently listed species go extinct and all of the currently established NIS persist, 13% of the fauna will be non-native. Alpha functional diversity in birds and mammals increased with the inclusion of NIS and increased further after endangered species were removed (Table 3.7). Postinvasion, the bird community gained two new functional groups (aquatic omnivore and terrestrial omnivore). The mammal community gained two new functional groups (arboreal granivore and terrestrial omnivore).

Southwestern Australia - Historically, there were 184 bird and mammal species in the Mediterranean-climate ecosystem of Southwestern Australia. Currently, approximately 17% of these species are endangered and 11% are NIS. If all currently listed species go extinct and all of the currently established NIS persist, 13% of the fauna will be non-native. Alpha diversity of function in birds increased with the inclusion of NIS and increased further after endangered species were removed. Alpha diversity of function in mammals increased with the inclusion of NIS and increased further after endangered species were removed (Table 3.7). Postinvasion, the bird community gained

two new functional groups (aquatic insectivore and aquatic piscivore). The mammal community gained one new functional group (terrestrial omnivore).

Alpha diversity of function increased in 9 out of the 10 Mediterranean-climate ecosystems analyzed when NIS were introduced into the community. Alpha diversity of function increased in eight ecosystems and decreased in two with the removal of endangered species from the community after introducing NIS (Table 3.7). Beta diversity of function in birds decreased in 80% of the Mediterranean-climate pair-wise comparisons analyzed when NIS were introduced and remained the same or decreased in 60% of the Mediterranean-climate pair-wise comparisons with the removal of endangered species from the communities (Table 3.8). Beta diversity of function in mammals increased in 70% of the Mediterranean-climate pair-wise comparisons analyzed when NIS were introduced and remained the same or increased in 80% of comparisons when endangered species were subsequently removed from the communities (Table 3.9). Gamma diversity of function in birds and mammals increased with the inclusion of NIS and after endangered species were removed (Table 3.10).

Functional Redundancy

Mammals

California - Historically, the mammal community within the California Mediterranean-climate ecosystem consisted of 34 species from 8 functional groups (Appendix A). The within-aggregation functional diversity ranged from two to five functional groups. The aggregations of smaller-bodied mammals had more species and higher diversity of functional groups than larger-bodied mammals. Of the 8 functional groups, two functional groups occurred in three or more aggregations, one functional

group had one representative and occurred in only one aggregation, and the remaining five functional groups occurred in two aggregations. In 23% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, only one aggregation lost one functional group, therefore the loss of functional diversity within aggregations was minimal. Membership in each functional group within each aggregation had no overall net loss of species. In 28% of the cases only one species remained to represent a functional group within an aggregation. The mammal community within the California Mediterranean-climate ecosystem gained one new functional group in four aggregations and lost one functional group in one aggregation. Overall redundancy of function across aggregations (i.e., cross-scale redundancy) was increased in two functional groups (Table 3.11).

Chile - Historically, the mammal community within the Chile Mediterranean-climate ecosystem consisted of 29 species from 10 functional groups (Appendix B). The within-aggregation functional diversity ranged from two to six functional groups. The aggregations of smaller-bodied mammals had more species and higher diversity of functional groups than larger-bodied mammals. Of the 10 functional groups, two functional groups occurred in three or more aggregations, and five functional groups had one representative and occurred in only one aggregation. In 25% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, only one aggregation lost one functional group, therefore the loss of functional diversity within aggregations was minimal. Membership in each functional group within

each aggregation had an overall net gain of six species. In 27% of the cases only one species remained to represent a functional group within an aggregation. The mammal community within the Chile Mediterranean-climate ecosystem gained three new functional groups in one aggregation and two new functional groups in another aggregation and lost one functional group in one aggregation. Cross-scale redundancy increased in three functional groups and decreased in one functional group (Table 3.12).

Spain - Historically, the mammal community within the Spain Mediterranean-climate ecosystem consisted of 46 species from 10 functional groups (Appendix C). The within-aggregation functional diversity ranged from two to five functional groups. The aggregations of smaller-bodied mammals had more species and higher diversity of functional groups than larger-bodied mammals. Of the 10 functional groups, three functional groups occurred in four or more aggregations, and three functional groups had one representative and occurred in only one aggregation. In 14% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, one aggregation lost one functional group and two functional groups were lost in another aggregation. Membership in each functional group within each aggregation had an overall net loss of one species. In 10% of the cases only one species remained to represent a functional group within an aggregation. The mammal community within the Spain Mediterranean-climate ecosystem gained one functional group in one aggregation, lost one functional group in one aggregation, and lost two functional groups in another aggregation. Cross-scale redundancy increased in one functional group and decreased in three functional groups (Table 3.13).

South Africa - Historically, the mammal community within the South Africa Mediterranean-climate ecosystem consisted of 66 species from 7 functional groups (Appendix D). The within-aggregation functional diversity ranged from two to five functional groups. The aggregations of smaller-bodied mammals had a similar number of species and diversity of functional groups compared to larger-bodied mammals. Of the 7 functional groups, three functional groups occurred in four or more aggregations, and two functional groups had one representative and occurred in only one aggregation. In 10% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, one aggregation lost two functional groups. Membership in each functional group within each aggregation had no overall net loss of species. In 11% of the cases only one species remained to represent a functional group within an aggregation. The mammal community within the South Africa Mediterranean-climate ecosystem gained one functional group in three aggregations and lost one functional group in two aggregations. Cross-scale redundancy increased in one functional group and decreased in two functional groups (Table 3.14).

Southwestern Australia – Historically, the mammal community within the southwestern Australia Mediterranean-climate ecosystem consisted of 44 species from 7 functional groups (Appendix E). The within-aggregation functional diversity ranged from two to five functional groups. The aggregations of smaller-bodied mammals had more species and higher diversity of functional groups than larger-bodied mammals. Of the 7 functional groups, two functional groups occurred in four or more aggregations, and one

functional group had one representative and occurred in only one aggregation. In 29% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, one aggregation lost three functional groups, one aggregation lost two functional groups and one aggregation lost one functional group. Membership in each functional group within each aggregation had an overall net loss of seven species. In 23% of the cases only one species remained to represent a functional group within an aggregation. The mammal community within the southwestern Australia Mediterranean-climate ecosystem gained one functional group in three aggregations, lost three functional groups in one aggregation, lost two functional groups in another aggregation, and lost one functional group in an aggregation. Cross-scale redundancy increased in one functional group and decreased in four functional groups (Table 3.15).

Birds

California - Historically, the bird community within the California Mediterranean-climate ecosystem consisted of 114 species from 16 functional groups (Appendix F). The within-aggregation functional diversity ranged from two to eight functional groups. The aggregations of smaller-bodied birds had the same number of species, but a lower diversity of functional groups compared to larger-bodied birds. Of the 16 functional groups, three functional groups occurred in five or more aggregations, and three functional groups had one representative and occurred in only one aggregation. In 8% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, two aggregations lost one functional group. Membership in each functional group

within each aggregation had an overall net loss of one species. In 13% of the cases only one species remained to represent a functional group within an aggregation. The bird community within the California Mediterranean-climate ecosystem gained one functional group in three aggregations, gained three functional groups in two aggregations, and lost one functional group in two aggregations. Cross-scale redundancy increased in three functional groups and decreased in one functional group (Table 3.16).

Chile - Historically, the bird community within the Chile Mediterranean-climate ecosystem consisted of 121 species from 19 functional groups (Appendix G). The within-aggregation functional diversity ranged from three to eight functional groups. The aggregations of smaller-bodied birds had more species, but a lower diversity of functional groups compared to larger-bodied birds. Of the 19 functional groups, four functional groups occurred in five or more aggregations, and three functional groups had one representative and occurred in only one aggregation. In 18% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, two aggregations lost one functional group. Membership in each functional group within each aggregation had an overall net gain of six species. In 17% of the cases only one species remained to represent a functional group within an aggregation. The bird community within the Chile Mediterranean-climate ecosystem gained one functional group in two aggregations and lost one functional group in two aggregations. Cross-scale redundancy increased in two functional groups and decreased in two functional groups (Table 3.17).

Spain - Historically, the bird community within the Spain Mediterranean-climate ecosystem consisted of 119 species from 14 functional groups (Appendix H). The within-aggregation functional diversity ranged from three to eight functional groups. The aggregations of smaller-bodied birds had more species and a higher diversity of functional groups compared to larger-bodied birds. Of the 14 functional groups, five functional groups occurred in four or more aggregations, and three functional groups had one representative and occurred in only one aggregation. In 19% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, four aggregations lost one functional group. Membership in each functional group within each aggregation had an overall net loss of six species. In 21% of the cases only one species remained to represent a functional group within an aggregation. The bird community within the Spain Mediterranean-climate ecosystem gained one functional group in one aggregation, gained two functional groups in two aggregations, and lost one functional group in four aggregations. Cross-scale redundancy increased in two functional groups and decreased in four functional groups (Table 3.18).

South Africa - Historically, the bird community within the South Africa Mediterranean-climate ecosystem consisted of 91 species from 12 functional groups (Appendix I). The within-aggregation functional diversity ranged from two to seven functional groups. The aggregations of smaller-bodied birds had more species and a higher diversity of functional groups compared to larger-bodied birds. Of the 12 functional groups, three functional groups occurred in four or more aggregations, and two

functional groups had one representative and occurred in only one aggregation. In 14% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, one aggregation lost one functional group. Membership in each functional group within each aggregation had an overall net gain of nine species. In 14% of the cases only one species remained to represent a functional group within an aggregation. The bird community within the South Africa Mediterranean-climate ecosystem gained one functional group in three aggregations and lost one functional group in one aggregation. Cross-scale redundancy increased in one functional group and decreased in one functional group (Table 3.19).

Southwestern Australia - Historically, the bird community within the southwestern Australia Mediterranean-climate ecosystem consisted of 144 species from 16 functional groups (Appendix J). The within-aggregation functional diversity ranged from two to nine functional groups. The aggregations of smaller-bodied birds had more species and a similar diversity of functional groups compared to larger-bodied birds. Of the 16 functional groups, six functional groups occurred in five or more aggregations, and two functional groups had one representative and occurred in only one aggregation. In 15% of the cases only one species represented a functional group within an aggregation.

After all currently listed species go extinct and all of the currently established NIS persist, three aggregations lost one functional group. Membership in each functional group within each aggregation had no overall loss of species. In 14% of the cases only one species remained to represent a functional group within an aggregation. The bird community within the southwestern Australia Mediterranean-climate ecosystem gained

one functional group in three aggregations and lost one functional group in four aggregations. Cross-scale redundancy increased in one functional group and decreased in three functional groups (Table 3.20).

Across body mass aggregations, there were changes in the number of species belonging to each functional group (Figures 3.3 & 3.4). Postinvasion, on average, birds gained two new functional groups and mammals gained one in each Mediterranean-climate ecosystem. Birds in the functional group terrestrial granivores and those that were herbivorous comprised the majority of additional memberships, while insectivores (terrestrial, aquatic, foliage, bark, and aerial) comprised the majority of declines in functional group membership. Carnivorous and herbivorous mammals comprised the majority of additional functional group memberships, while herbivores and granivores comprised the majority of declines in functional group membership. In birds, NIS increased membership in terrestrial carnivores, terrestrial granivores, foliage herbivores, terrestrial herbivores, and terrestrial omnivores. In mammals, NIS increased membership in terrestrial carnivores, terrestrial herbivores, and terrestrial omnivores.

Most functional group membership declines were in insectivorous birds and mammals. Within each individual body mass aggregation, there were changes in species composition and functional groups present, however, small sample size and extremely low statistical power did not make it possible to detect differences in functional group richness preinvasion and post invasion within each ecosystem. There were no differences in functional group richness preinvasion and post invasion for each terrestrial vertebrate group or for both taxonomic groups combined (Table 3.21). There was a decrease in

cross-scale redundancy of functional groups in mammals and when both taxonomic groups were combined (Table 3.21).

DISCUSSION

Discontinuous body-mass distributions were found in all Mediterranean-climate ecosystems and taxa examined. Discontinuities in body mass distributions have also been shown in North American birds (Skillen and Maurer 2008), south Florida herpetofauna, birds, and mammals (Allen et al. 1999, Allen 2006), Pleistocene and Miocene mammals (Lambert and Holling 1998, Lambert 2006), tropical forest birds (Restrepo et al. 1997), and boreal region birds and mammals (Holling 1992) and in various other taxa (reviewed in Sendzimir et al. 2003). This analysis extends these conclusions to include animal communities across Mediterranean-climate ecosystems of the world.

Body mass distribution patterns observed using the statistical methods employed in this dissertation are real and are not an apparent effect of random noise. Sendzimir (1998) ran simulations to determine differences in observed body mass distributions compared to simulated body mass distribution patterns. Aggregation pattern recognition did decline below robust levels in the 10 to 15 percent (mammals) and 5 percent level (birds) of error due to random variation in body size in mammals. This decline in aggregation pattern recognition was evident across all body sizes in both mammals and birds.

Mediterranean-climate ecosystems are geographically isolated regions that have long been considered to be structurally similar and have similar climatic regimes. However, more in-depth research of ecological processes within these ecosystems

suggests that these regions are examples of both convergence and divergence (reviewed in Rundel 2011). Only 35% of all pairwise comparisons between the structures of body mass distributions in five Mediterranean-climate ecosystems were determined to be similar. These results may be due to inherent differences in soil fertility, differential rainfall, topographic heterogeneity, climatic heterogeneity, and different disturbance regimes (i.e., fire frequency) among these Mediterranean-climate ecosystems (reviewed in Cowling et al. 1996).

Differences in human transformation of these Mediterranean-climate ecosystems might also explain the lack of similarity in all pairwise comparisons between ecosystems. The Mediterranean Basin (i.e., Spain in this analysis) has the oldest history of human occupation (i.e., agriculture and animal husbandry) dating back 10,000 years ago (Naveh & Kutiel 1990). The other regions were inhabited by hunter-gatherers and pastoralists until colonization which subsequently brought European grains and livestock to these regions (Aschmann 1973). Differences in time since settlement, cultural differences between the colonizing countries, and other regional differences (i.e., availability of resources and relationships between colonizers and indigenous people) may have contributed to differences in current patterns of human transformation between Mediterranean-climate ecosystems, thus resulting in differences in landscape structure between ecosystems (Hobbs et al. 1995). Differences in present and projected human population sizes, growth rates, and densities between Mediterranean-climate ecosystems have major impacts on land use, resulting in different patterns of landscape transformation (Mooney et al. 2001). Thus, body mass distributions within these ecosystems will never be the same in all pairwise comparisons. My results may also be an

artifact of the type of statistical analysis applied to the data. Chi-square and Phi correlation analyses may not have been appropriate; however, no other statistical analyses for this type of comparison have ever been documented.

The introduction of invasive species led to an increase in alpha diversity of function in 90% of the Mediterranean-climate ecosystems analyzed and an increase in alpha diversity of function in 80% of the Mediterranean-climate ecosystems analyzed after the removal of endangered species. These results provide further support for the trends reported in numerous studies regarding an increase in alpha diversity after the establishment of non-indigenous species (Olden and Rooney 2006, reviewed in Sax and Gaines 2003). An increase in alpha diversity of functional supports the notion that, at a local scale, invasive species richness and native species richness are positively correlated and invasive species do not reduce native species richness (Houlahan and Findlay 2004, Sax 2002, Sax et al. 2002, and Ellis et al. 2000). While Fridley et al. (2007) agree that species rich ecosystems are hotspots for invasive species, they contend that a decrease in local species richness can increase the success of invasion.

Beta diversity of function in birds decreased in 80% (when non-indigenous species were introduced) and remained the same or decreased in 60% (when endangered species were subsequently removed) of the Mediterranean-climate pair-wise comparisons. The opposite was seen in mammals, where beta diversity of function increased in 70% (when non-indigenous species were introduced) and remained the same or increased in 80% (when endangered species were subsequently removed) of the Mediterranean-climate pair-wise comparisons. A decrease in beta diversity of function supports studies reporting a regional scale decline in species richness following the

introduction of invasive species (Newmark 1995, Vitousek et al. 1996). A decrease in beta diversity represents the change from a regionally distinct community to a more homogenous community and this process has been termed biotic homogenization (McKinney and Lockwood 1999). Biotic homogenization has been suggested as an influential process which could negatively impact ecosystem function and resilience (reviewed in Olden et al. 2004).

An increase in beta diversity of function supports studies which state no apparent loss in species richness at a regional scale from invasive species and no subsequent biotic homogenization (Davis 2003, Rosenzweig 2001). These mixed results at a regional scale have been described elsewhere (reviewed in Sax and Gaines 2003). Contrary to Sax and Gaines (2003) review of species richness, gamma diversity of function (i.e., global scale) increased in both taxonomic groups and supports the theory that as scale increases, species richness increases (Rosenzweig 1999). This supports studies that have concluded invasive species may not threaten overall richness (Rosenzweig 2001).

Parker et al. (1999) and Vitousek et al. (1996) describe many kinds of environmental effects associated with non-indigenous species: including genetic (loss of genetic diversity and evolutionary pressure due to hybridization), ecosystem (alteration of nutrient cycling and productivity), and population or community (species richness may increase, but abundance of native species declines). Although there were no significant differences between pre- and post- invasion Shannon Index values, results of this study demonstrated a trend of non-indigenous species positively affecting alpha diversity of function in both birds and mammals, negatively affecting beta diversity of function in birds, and positively affecting beta diversity of function in mammals.

Despite differences in pre- and post- invasion species communities, the number of functional groups in a given body mass aggregation (functional richness w/n scales) remained similar pre- and postinvasion. However, changes in the distribution of species within functional groups across different body mass aggregations in mammals and when both taxonomic groups were combined, further validate an apparent decrease in functional redundancy and cross-scale resilience. Land-use intensification in Mediterranean-climate ecosystems has most likely resulted in loss of functional redundancy, which ultimately increases the chances of successful invasions (Laliberte et al. 2010, Hooper and Dukes 2010). The loss of cross-scale resilience and simplification of these communities due to human influences is well documented and may also have unforeseen consequences (Regier and Baskerville 1986, Peterson et al. 1998, Fors and Allen 2002). In recent years, one such consequence is the hotly debated subject of global homogenization, or the loss of specialist species (Clavel et al. 2010).

With regard to cross-scale redundancy in birds remaining the same, it may be that the overall net effect of the establishment of NIS birds in these ecosystems has offset losses of species due to other anthropogenic impacts (e.g., habitat fragmentation, degradation or destruction) (Fors and Allen 2002). Birds had higher species diversity than mammals, which some authors suggest would decrease the probability of a whole functional group becoming extinct (Fonseca & Ganade 2001). This may also be a result of no niche replacement, but conservation of broad functional patterns via functional complementarity (Rosenfeld 2002). Sundstrom et al. (2011) postulated that the loss of one or more species within a functional group may decrease competition between other

species within that group, thus inherently increasing the remaining species' ability to survive.

The results of this study clearly have significant implications for the field of landscape ecology and conservation biology. Humans continually altering the landscape at an increasing rate, current patterns of global change (i.e., climate change and globalization), and an increased rate of invasions (Vitousek et al. 1996, Mooney and Cleland 2001, Lodge and Shrader-Frechette 2003), will continue to change the composition of animal communities both locally, regionally and globally. Although there is no overwhelming evidence that invasive species cause a disproportionate share of incipient and actual extinctions on mainlands (reviewed in Gurevitch and Padilla 2004), this study suggests invasive species negatively impact key components of ecosystems (i.e., diversity of functional groups, functional redundancy and cross-scale resilience).

Davis and Thompson (2001) state traits between native and non-indigenous species are effectively indistinguishable; therefore, future analyses should include abundance data for all species in order to ascertain the level of functional extinctions (Carlton et al. 1999). Future analyses of functional diversity must also account for the potential variation in functional roles of species in heterogeneous environments (see Wellnitz and Poff 2001) and the potentially disproportionate effect of both a particular functional group and/or one of its members (Hoey and Bellwood 2009). Sustaining ecosystem resilience will require the conservation of both species diversity and diversity of function across multiple scales within an ecosystem.

Table 3.1. Functional groups used to classify the vertebrate fauna of California, Chile, South Africa, Spain and Southwestern Australia based on foraging strata and diet.

Diet	Foraging Strata	Abbreviation
Carnivore	Aerial	CaAe
Carnivore	Aquatic	CaAq
Carnivore	Arboreal	CaAr
Carnivore	Terrestrial	CaTe
Granivore	Aquatic	GrAq
Granivore	Arboreal	GrAr
Granivore	Foliage	GrFo
Granivore	Terrestrial	GrTe
Herbivore	Aerial	HeAe
Herbivore	Aquatic	HeAq
Herbivore	Arboreal	HeAr
Herbivore	Foliage	HeFo
Herbivore	Fossorial	HeFs
Herbivore	Terrestrial	HeTe
Insectivore	Aerial	InAe
Insectivore	Aquatic	InAq
Insectivore	Arboreal	InAr
Insectivore	Bark	InBa
Insectivore	Foliage	InFo
Insectivore	Fossorial	InFs

Table 3.1. Continued

Diet	Foraging Strata	Abbreviation
Insectivore	Terrestrial	InTe
Omnivore	Aquatic	OmAq
Omnivore	Bark	OmBa
Omnivore	Foliage	OmFo
Omnivore	Terrestrial	OmTe
Nectivore	Aerial	NeAe
Nectivore	Foliage	NeFo
Piscivore	Aerial	PiAe
Piscivore	Aquatic	PiAq
Piscivore	Terrestrial	PiTe

Table 3.2. Overall number of native species in five Mediterranean-climate ecosystems, the overall percentage of endangered and non-indigenous species (NIS) and the percentage of NIS species in each region after a hypothetical removal of threatened species.

Region	Number of Native Bird and Mammal Species		% NIS After Extinction of Threatened Species	
		% Endangered	% NIS	
California	148	12.8	13.5	15.5
Chile	150	3.3	13.3	13.8
Spain	165	20	10.3	12.9
South Africa	157	8.3	11.5	12.5
Southwestern Australia	188	16.5	10.6	12.7

Table 3.3. Chi-Square probability levels for each pairwise comparison of Mediterranean-climate ecosystem mammal body mass aggregations. Probability levels less than or equal to 0.005 were considered significant matches between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	<.0001*	<.0001*	0.0326	<.0001*
Chile	---	<.0001*	<.0001*	0.4484
Spain		---	0.5239	0.4445
South Africa			---	0.3378

Table 3.4. Phi coefficients for each pairwise comparison of Mediterranean-climate ecosystem mammal body mass aggregations using. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	0.2895*	0.3932*	-0.1016	0.4488*
Chile	---	0.3790*	0.4681*	0.0486
Spain		---	-0.0301	0.0455
South Africa			---	-0.0519

Table 3.5. Chi-Square probability levels for each pairwise comparison of Mediterranean-climate ecosystem bird body mass aggregations using. Probability levels less than or equal to 0.005 were considered significant matches between the lump and gap architecture of the compared ecosystems.

			South	Southwestern
	Chile	Spain	Africa	Australia
California	0.0023*	0.1059	<.0001*	0.1184
Chile	---	0.0566	0.9393	0.9634
Spain		---	0.2529	0.2389
South Africa			---	0.0235

Table 3.6. Phi coefficients for each pairwise comparison of Mediterranean-climate ecosystem bird body mass aggregations. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared ecosystems.

	South			Southwestern
	Chile	Spain	Africa	Australia
California	0.1771	0.0940	0.5017*	-0.0860
Chile	---	0.1143	-0.0046	0.0027
Spain		---	-0.0692	-0.0682
South Africa			---	-0.1366

Table 3.7. Alpha diversity of function for the vertebrate fauna of California, Chile, South Africa, Spain, and Southwestern Australia. (Shannon Diversity Index (H1), divergence from equiprobability (D1), equitability (Evenness), Non-indigenous species (NIS), and Endangered species (E). A high H1 value indicates a high degree of diversity.

	Birds			Mammals		
California	H1	D1	Evenness	H1	D1	Evenness
Functional Groups w/o NIS	3.096	0.712	81.305	2.449	0.358	87.241
Functional Groups w/NIS	3.280	0.808	80.241	2.743	0.427	86.545
Functional Groups w/o E w/NIS	3.283	0.717	82.064	2.790	0.380	88.016
Chile						
Functional Groups w/o NIS	3.623	0.625	85.279	2.803	0.519	84.390
Functional Groups w/NIS	3.638	0.610	85.635	2.910	0.550	84.098
Functional Groups w/o E w/NIS	3.607	0.641	84.911	3.017	0.443	87.197
South Africa						
Functional Groups w/o NIS	2.724	0.735	78.749	1.845	0.740	71.366
Functional Groups w/NIS	2.822	0.878	76.266	2.064	0.936	68.794
Functional Groups w/o E w/NIS	2.836	0.864	76.639	2.187	0.813	72.915
Spain						
Functional Groups w/o NIS	2.853	0.732	79.577	2.672	0.650	80.432
Functional Groups w/NIS	3.030	0.877	77.547	2.616	0.706	78.744
Functional Groups w/o E w/NIS	3.116	0.791	79.750	2.420	0.580	80.680
Southwestern Australia						
Functional Groups w/o NIS	3.223	0.581	84.743	2.187	0.621	77.890
Functional Groups w/NIS	3.322	0.678	83.051	2.386	0.614	79.530
Functional Groups w/o E w/NIS	3.323	0.677	83.071	2.608	0.392	86.930

Table 3.8. Beta diversity of function for the bird fauna of each Mediterranean-climate ecosystem. (Shannon Diversity Index (H1), divergence from equiprobability (D1), equitability (Evenness), Non-indigenous species (NIS), and Endangered species (E). A high H1 value indicates a high degree of diversity.

	San Diego v Chile			Chile v South Africa			South Africa v Southwestern Australia		
	H1	D1	Evenness	H1	D1	Evenness	H1	D1	Evenness
Functional Groups w/o NIS	2.997	0.173	94.533	3.017	0.304	90.835	2.236	0.086	96.296
Functional Groups w/NIS	2.774	0.226	92.483	2.924	0.398	88.001	2.663	0.144	94.877
Functional Groups w/o E w/NIS	2.774	0.226	92.483	2.943	0.379	88.596	2.663	0.144	94.877
	San Diego v South Africa			Chile v Spain			Spain v Southwestern Australia		
Functional Groups w/o NIS	2.936	0.234	92.629	3.051	0.119	96.241	1.485	0.515	74.234
Functional Groups w/NIS	2.703	0.297	90.094	2.407	0.178	93.103	1.122	1.200	48.312
Functional Groups w/o E w/NIS	2.520	0.284	89.879	2.407	0.178	93.103	1.122	1.200	48.312
	San Diego v Spain			Chile v Southwestern Australia					
Functional Groups w/o NIS	2.379	0.206	92.024	2.674	0.648	80.486			
Functional Groups w/NIS	2.264	0.321	87.576	2.514	0.656	79.297			
Functional Groups w/o E w/NIS	2.466	0.341	87.950	2.453	0.717	77.378			
	San Diego v Southwestern Australia			South Africa v Spain					
Functional Groups w/o NIS	1.923	0.662	74.400	2.604	0.204	92.746			
Functional Groups w/NIS	1.912	0.895	68.111	2.749	0.251	91.631			
Functional Groups w/o E w/NIS	2.142	0.858	71.410	2.781	0.219	92.688			

Table 3.9. Beta diversity of function for the mammal fauna of California, Chile, South Africa, Spain, and Southwestern Australia. (Shannon Diversity Index (H1), divergence from equiprobability (D1), equitability (Evenness), Non-indigenous species (NIS), and Endangered species (E). A high H1 value indicates a high degree of diversity.

	San Diego v Chile			Chile v South Africa			South Africa v Southwestern Australia		
	H1	D1	Evenness	H1	D1	Evenness	H1	D1	Evenness
Functional Groups w/o NIS	2.252	0.070	96.972	2.369	0.216	91.627	2.656	0.152	94.596
Functional Groups w/NIS	1.585	0.085	94.639	2.611	0.197	92.990	2.822	0.178	94.073
Functional Groups w/o E w/NIS	1.585	0.085	94.639	2.611	0.197	92.990	2.918	0.082	97.277
	San Diego v South Africa			Chile v Spain			Spain v Southwestern Australia		
Functional Groups w/o NIS	1.784	0.538	76.846	0.811	0.189	81.128	2.113	0.209	91.014
Functional Groups w/NIS	1.890	0.695	73.099	1.449	0.136	91.410	2.396	0.189	92.700
Functional Groups w/o E w/NIS	1.949	0.636	75.403	2.230	0.092	96.023	1.189	0.189	90.564
	San Diego v Spain			Chile v Southwestern Australia					
Functional Groups w/o NIS	2.198	0.125	94.627	2.128	0.194	91.652			
Functional Groups w/NIS	1.730	0.270	86.479	2.171	0.151	93.498			
Functional Groups w/o E w/NIS	2.055	0.267	88.486	2.156	0.166	92.838			
	San Diego v Southwestern Australia			South Africa v Spain					
Functional Groups w/o NIS	1.842	0.158	92.119	1.906	0.094	95.282			
Functional Groups w/NIS	1.793	0.208	89.624	2.411	0.173	93.294			
Functional Groups w/o E w/NIS	2.000	0.000	100.000	2.413	0.172	93.351			

Table 3.10. Gamma diversity of function for the vertebrate fauna of California, Chile, South Africa, Spain, and Southwestern Australia. (Shannon Diversity Index (H1), divergence from equiprobability (D1), equitability (Evenness), Non-indigenous species (NIS), and Endangered species (E). A high H1 value indicates a high degree of diversity.

	H1	D1	Evenness
Birds			
Gamma Diversity without NIS	3.409	1.176	74.343
Gamma Diversity with NIS	3.477	1.108	75.838
Functional Groups without E and with NIS	3.509	1.076	76.530
Mammals			
Gamma Diversity without NIS	2.630	1.177	69.086
Gamma Diversity with NIS	2.758	1.050	72.433
Functional Groups without E with NIS	2.849	0.959	74.825

Table 3.11. Each body mass aggregation represents a unique scale within the California Mediterranean-climate ecosystem. The total number of mammal species within each functional group and the number of functional groups lost at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. Membership is defined as the number of functional groups that gained or lost species after the introduction of invasive species and the extinction of endangered species.

Body Mass Aggregation	CaTe	GrTe	HeAr	HeAq	HeFs	HeTe	InTe	OmTe	Membership Gained	Membership Lost	Functional Groups Lost
1		4	1			(1)	2		1	0	-
2	1	7(6)	2		1	1		(2)	1	1	-
3	2(1)					3	1	(2)	1	1	-
4	4			(1)				1	1	-	-
5	2(1)					1		1(0)	0	2	1

Table 3.12. Each body mass aggregation represents a unique scale within the Chile Mediterranean-climate ecosystem. The total number of mammal species within each functional group and the number of functional groups lost at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. Membership is defined as the number of functional groups that gained or lost species after the introduction of invasive species and the extinction of endangered species.

Body Mass Aggregation	CaAq	CaAr	CaTe	GrTe	HeAq	HeAr	HeFs	HeTe	InAq	InTe	OmTe	Membership Gained	Membership Lost	Functional Groups Lost
1	(1)		(1)	4	(1)	2	1	6		2		3	0	-
2		1	3(4)		1(2)			1(3)	1	1		3	0	-
3	1		1					1(0)				0	1	1
4			1		(1)			2			(1)	2	1	-

Table 3.13. Each body mass aggregation represents a unique scale within the Spain Mediterranean-climate ecosystem. The total number of mammal species within each functional group and the number of functional groups lost at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. Membership is defined as the number of functional groups that gained or lost species after the introduction of invasive species and the extinction of endangered species.

Body Mass Aggregation	CaAq	CaTe	GrTe	HeAq	HeAr	HeFs	HeTe	InAq	InFs	InTe	Membership Gained	Membership Lost	Functional Groups Lost
1			2							4	-	-	-
2			1(2)			1	4	1		2	1	0	-
3							3(1)	1	3		0	1	-
4		1		1(0)	2(1)		(2)				1	2	1
5		2								2(3)	1	0	-
6		1(3)					3				1	0	-
7	1(0)	1(0)					3			1	0	2	2
8		1					4				-	-	-

Table 3.14. Each body mass aggregation represents a unique scale within the South Africa Mediterranean-climate ecosystem. The total number of mammal species within each functional group and the number of functional groups lost at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. Membership is defined as the number of functional groups that gained or lost species after the introduction of invasive species and the extinction of endangered species.

Body Mass Aggregation	CaAq	CaTe	GrAr	HeFs	HeTe	InFs	InTe	OmTe	Membership Gained	Membership Lost	Functional Groups Lost
1					2(3)		5		1	0	-
2					6	1	3		-	-	-
3		1			8(7)	2	2	(2)	1	1	-
4			(1)	1			3		1	0	-
5		5(6)			4				1	0	-
6	1	6(5)			8(7)		2	(1)	1	2	-
7		1(0)			5(0)				0	2	2

Table 3.15. Each body mass aggregation represents a unique scale within the southwestern Australia Mediterranean-climate ecosystem. The total number of mammal species within each functional group and the number of functional groups lost at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. Membership is defined as the number of functional groups that gained or lost species after the introduction of invasive species and the extinction of endangered species.

Body Mass Aggregation	CaTe	GrTe	HeAr	HeTe	InAq	InAr	InTe	OmTe	Membership Gained	Membership Lost	Functional Groups Lost
1		3(2)	1	5(3)		2(1)	7(5)		0	4	-
2		1(0)		2(0)		1(0)	1	(2)	1	3	3
3	1(0)		1(0)	6(1)	1		2		0	3	2
4	(2)		1	5(0)			1		1	1	1
5	1			2(3)				(1)	2	0	-

Table 3.16. Each body mass aggregation (BMA) represents a unique scale within the California Mediterranean-climate ecosystem. The total number of bird species within each functional group and the number of functional groups lost (FL) at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. The number of functional groups that gained (G) or lost (L) species after the introduction of invasive species and the extinction of endangered species.

BMA	Ca Ae	Ca Aq	Gr Fo	Gr Te	He Ae	He Aq	He Fo	He Te	In Ae	In Aq	In Ba	In Fo	In Te	Om Aq	Om Ba	Om Fo	Om Te	G	L	FL
1					4							4						-	-	-
2			3(2)	(2)					5(4)			10(8)	7					1	3	-
3				1					4		1		4(3)					0	1	-
4				3(2)		1		3		1	4	3					0	1	-	
5				(1)					6		1	1	4					1	0	-
6												1(0)	3(2)					0	2	1
7	2(1)	2	(1)	5			(2)		2(1)	1			4(5)		1	(1)	2	4	2	-
8	12(8)	1		(1)		1(0)	1	(1)					2(3)	(1)			2	4	2	1
9	2(1)																(1)	1	1	-

Table 3.17. Each body mass aggregation (BMA) represents a unique scale within the Chile Mediterranean-climate ecosystem.

The total number of bird species within each functional group and the number of functional groups lost (FL) at each scale.

Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. The number of functional groups that gained (G) or lost (L) species after the introduction of invasive species and the extinction of endangered species.

BMA	Ca Ae	Ca Aq	Gr Aq	Gr Fo	Gr Te	He Ae	He Aq	He Fo	He Te	In Ae	In Aq	In Ba	In Fo	In Te	Ne Ae	Om Aq	Pi Ae	Pi Aq	Pi Te	G	L	FL
1										1		1	6	2	1					-	-	-
2					5					3		1	3	2	1					-	-	-
3					4(1)	1		1		2	1(0)	1	1	8(9)						2	1	1
4	1									1				8				1		-	-	-
5	2				2(3)		1	1(2)	1		4			3			2			2	0	-
6	6			1	(1)		3	1		1	2			2(3)					1	2	0	-
7	1	1	3		1		2				2					2		1		-	-	-
8	6	1					1		(1)							1(0)		3	1	1	1	1
9	3						1(2)							1						1	0	-

Table 3.18. Each body mass aggregation (BMA) represents a unique scale within the Spain Mediterranean-climate ecosystem. The total number of bird species within each functional group and the number of functional groups lost (FL) at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. The number of functional groups that gained (G) or lost (L) species after the introduction of invasive species and the extinction of endangered species.

BMA	Ca Ae	Ca Aq	Ca Te	Gr Aq	Gr Fo	Gr Te	He Aq	He Fo	He Te	In Ae	In Aq	In Ba	In Fo	In Te	Om Aq	G	L	FL
1	1(0)				(1)	10(12)				9(8)		1	14(13)	20(15)		2	4	1
2					1	1(0)				4	1		1	2		0	1	1
3						1				4(3)	1	1	1	3		0	1	-
4	8(6)	1		(1)		5	1	(1)	1	2(0)	2(1)		2	3		2	3	1
5	6(4)	2	1						(1)						(1)	2	1	-
6	7(3)		1											1(0)		0	2	1

Table 3.19. Each body mass aggregation (BMA) represents a unique scale within the South Africa Mediterranean-climate ecosystem. The total number of bird species within each functional group and the number of functional groups lost (FL) at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. The number of functional groups that gained (G) or lost (L) species after the introduction of invasive species and the extinction of endangered species.

BMA	CaAe	GrFo	GrTe	HeFo	HeTe	InAe	InBa	InFo	InTe	NeAe	NeFo	OmAq	OmTe	G	L	FL
1		1	2			1(2)		6	4	1	2			1	0	-
2		1	6(7)			5	1	1	4		1			1	0	-
3			1(2)	1		2(3)			5					2	0	-
4			2	4		3		(1)	11(12)		1			2	0	-
5	1		2(3)						4					1	0	-
6	4(3)		2		3				6			(1)	(1)	3	0	-
7	1								2(0)					0	1	1

Table 3.20. Each body mass aggregation (BMA) represents a unique scale within the southwestern Australia Mediterranean-climate ecosystem. The total number of bird species within each functional group and the number of functional groups lost (FL) at each scale. Numbers in parentheses represent the total number of mammal species within each functional group after the introduction of invasive species and the extinction of endangered species. The number of functional groups that gained (G) or lost (L) species after the introduction of invasive species and the extinction of endangered species.

BMA	Ca Ae	Ca Aq	Ca Te	Gr Fo	Gr Te	He Aq	He Fo	He Te	In Ae	In Aq	In Ba	In Fo	In Te	Ne Fo	Om Te	Pi Aq	G	L	FL
1				1								2					-	-	-
2				1	1(3)			1	3		1	10(9)	8(7)	3			1	2	-
3									4				1	2			-	-	-
4					3(2)		2		4		1(0)	5	5	7			0	2	1
5					3		1		4		1		8(7)				0	1	-
6				1(0)	2							1	1	2			0	1	1
7					3(4)			1	1			1	3				1	0	-
8				1	1(2)								1	2			1	0	-
9	3				3(4)				2				2		1		1	0	-
10				1	1(2)		1						2(0)			(1)	2	1	-
11	8	2	2	1(2)	1		1	2	1				1(0)			(1)	2	1	1
12	1					1(2)		2(1)		(1)			2(1)				2	2	-

Table 3.21. Results of Wilcoxon signed rank tests comparing functional group richness preinvasion and post invasion for each terrestrial vertebrate group and for both taxa combined.

Taxonomic Group	<i>z</i>	<i>P</i>
Birds	1.539	0.156
Mammals	0.108	0.940
Combined	1.156	0.251

Table 3.22. Results of Wilcoxon signed ranked tests comparing the average number of body mass aggregations and standard deviation (SD) in California, Chile, South Africa, Spain, and Southwestern Australia that had at least one member of a functional group pre- and postinvasion.

Taxonomic Group	Average	Average	z	P
	Number (SD) of	Number (SD) of		
	body mass	body mass		
	aggregations	aggregations		
	preinvasion	postinvasion		
Birds	3.129 (0.26)	3.11 (0.244)	-0.408	0.697
Mammals	2.375 (0.240)	1.75 (0.228)	-3.501	<0.001
Combined	2.880 (0.190)	2.778 (0.183)	-1.930	0.055

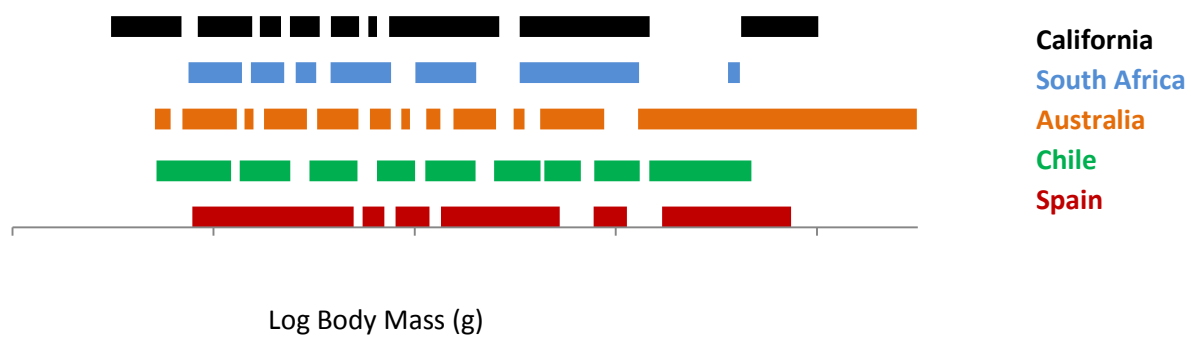


Figure 3.1. Juxtaposition of bird body mass aggregations across five Mediterranean-climate ecosystems.

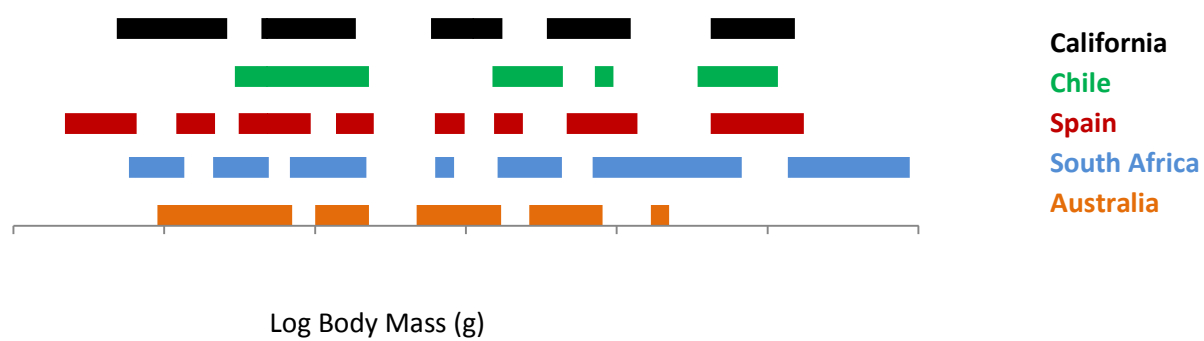


Figure 3.2. Juxtaposition of mammal body mass aggregations across five Mediterranean-climate ecosystems.

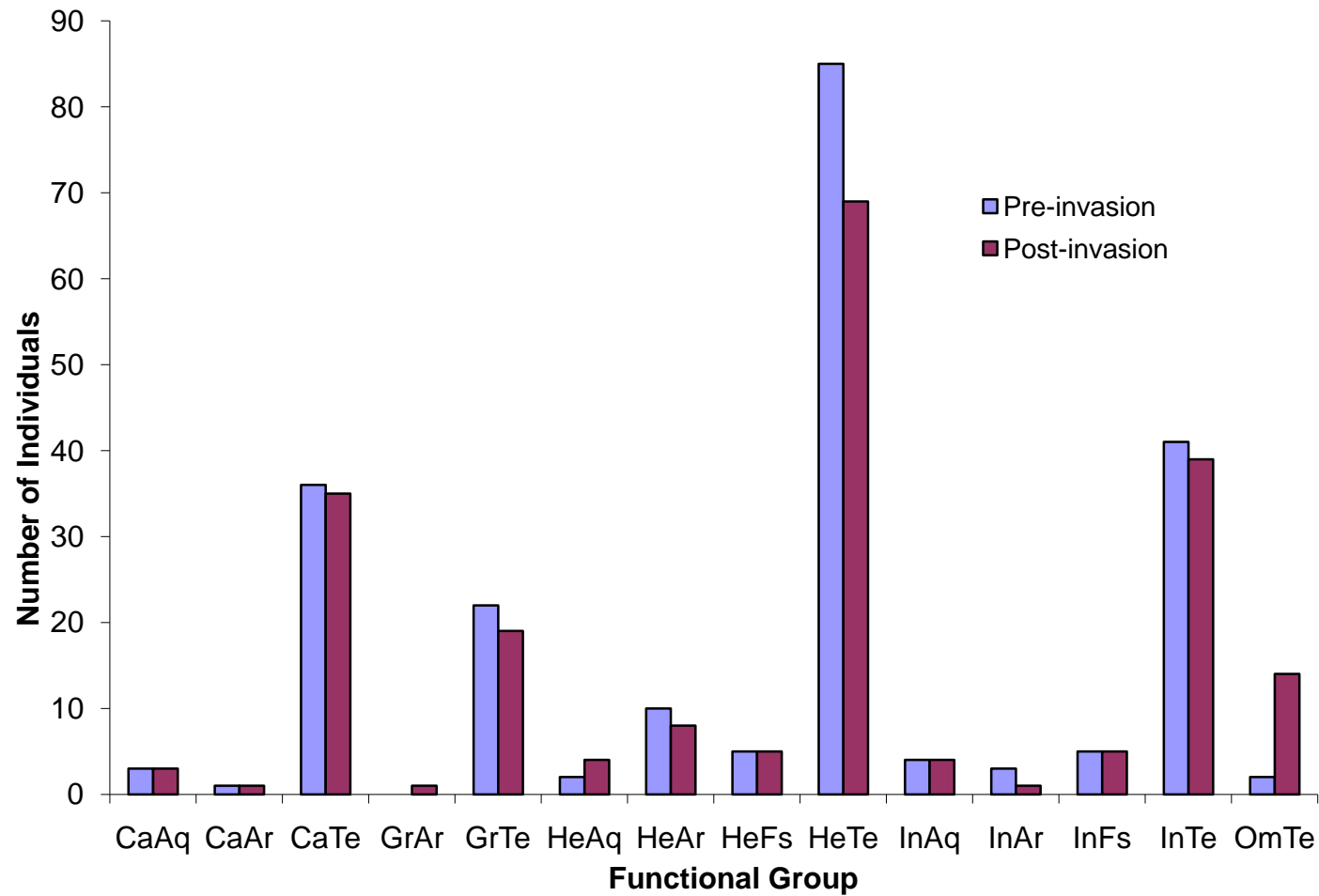


Figure 3.3. Comparison of the number of species belonging to each functional group across all body mass aggregations pre- and postinvasion for all the birds of Mediterranean-climate ecosystems. Explanations for the functional group abbreviations are provided in Table 3.1.

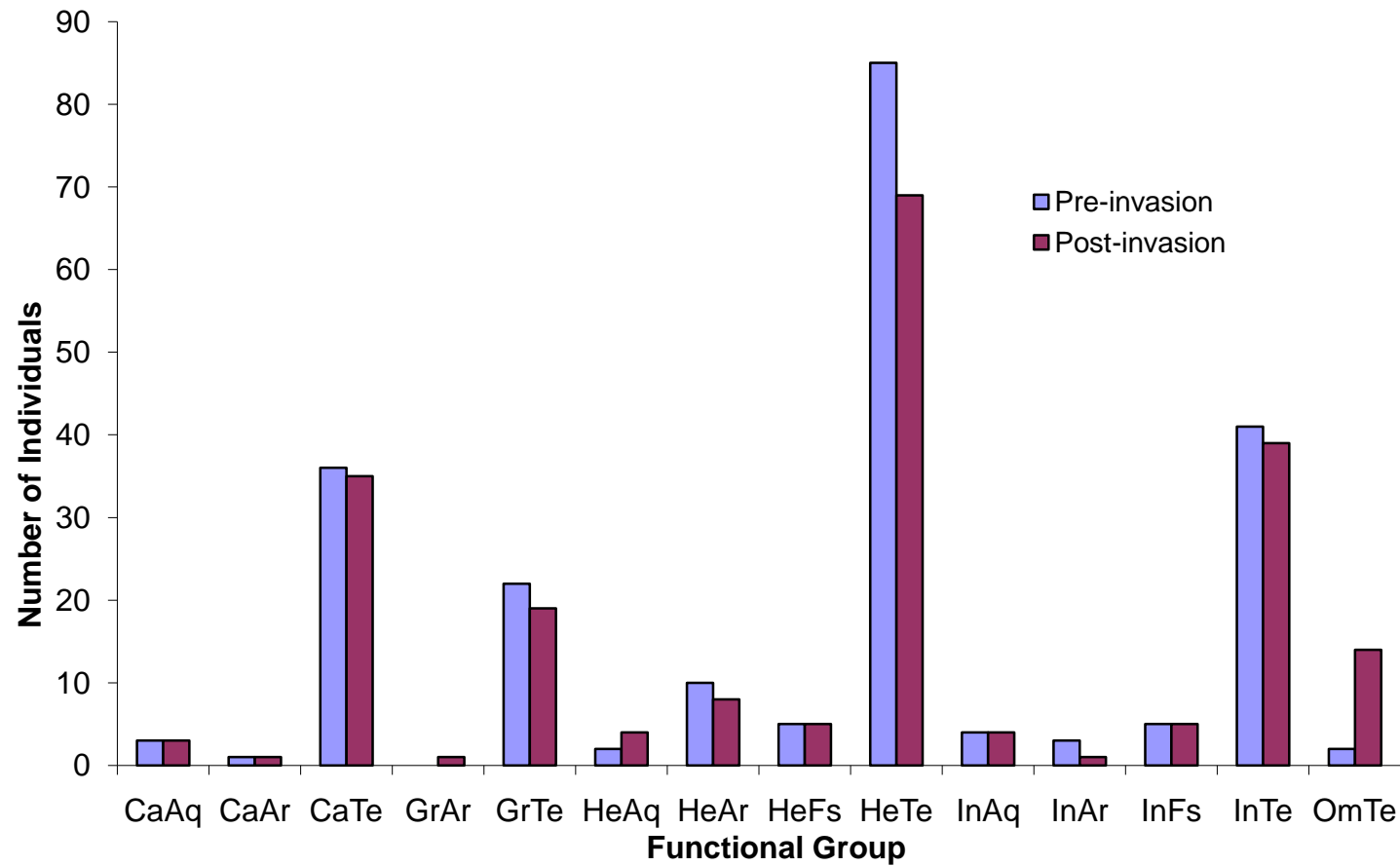


Figure 3.4. Comparison of the number of species belonging to each functional group across all body mass aggregations pre- and postinvasion for all the mammals of Mediterranean-climate ecosystems. Explanations for the functional group abbreviations are provided in Table 3.1.

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CHAPTER 4: CHANGES IN BODY MASS DISCONTINUITIES OVER PALEOECOLOGIC TIME

Recent population trends for 1,686 vertebrate species indicate a ~30% decline over the last 35 years (Loh et al. 2008). Human activities, both directly and indirectly, have contributed to the increasing rates of extinction of both described and undescribed species (Hilton-Taylor et al. 2008). In order to gain a better understanding of the dramatic ecological changes currently taking place, the complex relationship between animals and their environment must be understood from different perspectives. Biological systems are generally influenced by external, internal and historical processes (Marquet et al. 2008) and paleoecological data provides a temporal perspective on rates, patterns and causes of ecological change (Willard & Cronin 2007). Discerning patterns and processes of speciation events in the fossil record has been a topic of recent debate (reviewed in Benton & Pearson 2001), yet no study to date has utilized body mass distributions to elucidate speciation events.

The combination of physiological, ecological and historical factors make body mass one of the most integrative attributes of a species (Schmidt-Nielsen 1984, Allen et al. 2006, Marquet et al. 2008). The ecological significance of patterns found in body mass distributions has been well documented and reflect the spatiotemporal distribution of resources within ecosystems. Many studies have reported a discontinuous body mass distribution reflecting the ecological structure of the landscape (Restrepo et al. 1997, Bakker & Kelt 2000, Raffaelli et al. 2000, Havlicek & Carpenter 2001, Cumming & Havlicek 2002, Sendzimir et al. 2003). These studies support the textural discontinuity

hypothesis (TDH), which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape (Holling 1992).

The TDH implies that differences in body mass distributions between two faunal assemblages would indicate differences in landscape (Lambert & Holling 1998). The edges of body-mass aggregations, or scale breaks, in this discontinuous distribution may be regions of the greatest reward or highest cost (Holling et al. 2008). Poorly understood biological phenomena such as invasions and extinctions (Allen et al. 1999, Allen 2006), migrations and nomadism (Allen & Saunders 2002), population dynamics (Skillen & Maurer 2008) and possibly speciation may evolve at these scale breaks (Holling et al. 2008). However, evolution may lead to extinction of species without external forcing (Bak & Sneppen 1993).

Body size distributions have been analyzed at various scales; 1) local, biome and continental (Brown & Nicoletto 1991, Marquet & Cofre 1999), 2) hemispheric (Blackburn & Gaston 1994), and 3) Paleobiological scale (Jablonski 1997). However, there is very little information on the effects of paleoecological time on changes in body mass distributions. In Cenozoic mammals, climate has been suggested as a driver of body mass evolution and the analysis of body mass distributions is a good measure of habitat and climate (Gunnell et al. 1995). Legendre (1989) observed differences in body mass distributions between wet and dry habitats in the late Eocene and Oligocene of Occidental Europe. However, Alroy et al. (2000) did not find any link between climate and patterns in body mass distributions. Instead, they suggest that intrinsic, biotic factors seem to have played a more important role in broad patterns of North American mammalian evolution than external factors such as climate change.

Late Miocene global mammalian turnover has been attributed to changes in flora (i.e. a change in habitat) (Cerling et al. 1998). Holling et al. (2002, 2008) suggest it takes extreme disturbances over paleoecological time and space to change body mass distributions in a major way. Lambert and Holling (1998) reported that body mass distributions of Pleistocene mammals, after massive extinctions of large herbivores in North and South America approximately 11,000 years ago, remained the same in species under 41kg, but changed significantly in species over 41kg (species over 1,000kg were completely eliminated). Based on their results, they suggest that differences observed in body mass distributions were partly due to changes in key mesoscale (10m to 10km) aspects of the landscape. Smith et al. (2004) suggest that over evolutionary time, as lineages speciate and diversify, species do not occupy a greater range of body sizes. The Siberia megafauna extinction at the end of the Pleistocene may have triggered an irreversible regime shift from steppe grassland to tundra (Zimov et al. 1995, Lambert & Holling 1998, Folke et al. 2004).

Chronofaunas are discrete communities of species that are tracked through millions of years, resulting in the ability to identify the origination of new species and the extinction of others (Olson 1952, Olson 1966, MacFadden 2000). It has been suggested that the Eocene Epoch (55 to 34 million years ago (Ma)) was a period in the North American mammal faunal record that underwent rapid ecomorphological transitions (Janis & Wilhelm 1993, Hunter & Jernvall 1995, Jernvall et al. 1996). A likelihood ratio was calculated by Alroy et al. (2000) to determine how the dominance of different orders changed (from the late Paleocene through the Pleistocene) in relation to what would be expected at random given the observed origination and extinction rates. They reported

that four of the five highest likelihood ratios were observed in the Bridger and Uinta NALMA, due primarily to the radiation of artiodactyls and perissodactyls with a concurrent terminal decline of primates and “condylarths”, suggesting this span of time was one of major ecological transition and the most important period in the entire Tertiary. Mean body mass of mammals increased slowly throughout the Cenozoic and might have been due to high speciation or low extinction rates of larger species (Alroy et al. 2000).

The Bridger and Uinta NALMA may provide an ideal timeframe to examine speciation events and paleoecologic habitat reconstruction would be necessary to explore the TDH. Paleoecologic habitat reconstruction is generally done at either a global or regional scale and then extrapolated to a local scale. However, habitats for a sequential series of mammalian fossil assemblages have been reconstructed at a local scale for the Bridgerian North American Land Mammal Age (NALMA) and Uintan NALMA (Townsend 2004). A closer examination of body mass distributions from each assemblage may provide insightful information about their respective habitats and speciation events. Here I examined whether the vertebrate body mass structures are similar among the faunal zones. I also investigated whether speciation events were nonrandomly distributed in vertebrate body mass structures within each faunal zone.

METHODS

Study Area & Data Collection

The Cenozoic Era spans approximately 65 Ma to present day and consists of the Paleogene, Neogene and Quaternary Periods (MacFadden 2000). The Paleogene Period

consisted of the Paleocene, Eocene and Oligocene Epochs. The Eocene Epoch spanned 55 to 34 Ma and was characterized by a continual decrease in mean annual temperature (Bains et al. 2000) and a slow drying trend (Wing 1998). The Eocene Epoch was delineated into early (Wasatchian), middle (Bridgerian, Uintan, Duchesnean), and late (Chadronian) NALMA (Prothero 1998). I focused on the middle Bridgerian to the late Uintan mammalian fossil specimens, which spanned approximately 50 to 41 Ma.

The Bridgerian NALMA occurred at the beginning of the middle Eocene interval and spanned approximately five million years from 50.7 to 46 Ma (Woodburne and Swisher 1995). The Bridgerian formation was located in the Green River Basin of southwest Sweetwater County, Wyoming (Figure 4.1). In this analysis, the Bridgerian NALMA was organized into five sequential faunal assemblage zones (FAZ); Bridger FAZ 1) 35 mammalian genera, 3 unique to this FAZ, collected from 68 to 79m level, composed of taxa from nine localities and was predominantly made up of rodents, Bridger FAZ 2) 28 mammalian genera, 3 unique to this FAZ, collected from 122 to 161m level, composed of taxa from more than eighteen localities and was predominantly made up of rodents, Bridger FAZ 3) 28 mammalian genera, 4 unique to this FAZ, collected from the 163m level and 200 to 207m level, composed of taxa from twelve localities and was predominantly made up of rodents, primates and insectivores, Bridger FAZ 4) 18 mammalian genera collected from 210 to 257m level, composed of taxa from six localities (Townsend 2004). Bridger FAZ 4 represents the last faunal assemblage considered to be truly Bridgerian (McCarroll et al. 1996). Bridger FAZ 5 included 22 mammalian genera collected from 307 to 390m level, was composed of taxa from twelve

localities and is less diverse at the ordinal level than earlier FAZs (Townsend 2004). This zone is considered to be earliest Uintan in age (Evanoff et al. 1994).

The Uintan NALMA occurred at the middle of the middle Eocene and spanned approximately 6.5 million years from 40 to 46.5 Ma (Townsend 2004) and contains approximately 31% of modern mammalian families (Black & Dawson 1966). The Uintan formation was located in the Uinta Basin of east-central Uintah County, Utah and includes 483km²(Figure 4.1). In this analysis, the Uintan NALMA was organized into three sequential faunal assemblages; Uintan FAZ 1) 19 mammalian genera, 3 unique to this FAZ, collected from 0 to 87m level, composed of taxa from seventeen localities and was predominantly made up of rodents, primates and proteutherians, Uintan FAZ 2) 25 mammalian genera, 9 unique to this FAZ, collected from 96 to 128m level, composed of taxa from nineteen localities and was predominantly made up of rodents, Uintan FAZ 3) 20 mammalian genera, 4 unique to this FAZ, collected from 134 to 366m level, composed of taxa from 29 localities and was predominantly made up of rodents and artiodactyls (Townsend 2004).

All assemblages were time-averaged, due to the use of numerous localities in the construction of each assemblage. This resulted in the absence of a defined boundary or stratigraphic extent of each zone. The taphonomic contexts of both formations are similar, therefore it is assumed that any bias within or between assemblages will be statistically insignificant (Townsend 2004). Morphometric data on more than 2000 fossil specimens were taken on Bridgerian and Uintan mammalian fossils held at the American Museum of Natural History, the National Museum of Natural History (Smithsonian

Institution), Yale Peabody Museum, Carnegie Museum of Natural History, the University of Colorado Museum and Washington University (Townsend 2004).

Habitat Reconstructions & FAZ Composition

Townsend (2004) used univariate, bivariate and discriminant function analyses to reconstruct the habitats for each FAZ. There were many primates and an equal number of terrestrial taxa in Bridger FAZ 1, which would suggest that although there were trees present, it was not completely canopied. Thus, Bridger FAZ 1 was most likely open woodland habitat (Townsend 2004). Bridger FAZ 2 had a very similar species makeup to Bridger FAZ 1, therefore it was most likely open woodland habitat too (Townsend 2004).

The fauna in Bridger FAZ 3 contained the second highest percentage of arboreal mammals found in the entire sequence of FAZs, suggesting a large amount of canopy cover. Thus, Bridger FAZ 3 was most likely heavily wooded and densely forested (Townsend 2004). Bridger FAZ 4 most likely represented closed woodland or forested habitat, but this zone is the most ambiguous (due to missing taxa as a result of collecting bias or taphonomic considerations) (Townsend 2004). The fauna in Bridger FAZ 5 most likely represented closed woodland habitat (due to the high percentage of insectivores) or possibly forested (due to the high percentage of obligate arborealists and frugivores) (Townsend 2004).

Uinta FAZ 1 had fewer folivores and ground-dwelling taxa, but more insectivores (although very few compared to the Bridgerian FAZs) than the other Uinta FAZs, thus the fauna in this zone most likely represented woodland habitat. Uinta FAZ 2 consists of more terrestrial and folivorous taxa than in any prior FAZs, thus representing open habitat with very few trees (Townsend 2004). Uinta FAZ 3 was composed of more

folivores than frugivores, more terrestrial taxa than arboreal taxa, few small-bodied mammals and no insectivores. Uinta FAZ 3 had fauna that likely represented open habitat with the least amount of trees found in any FAZ (Townsend 2004).

Body Mass Estimates

Townsend (2004) used various phylogenetically consistent equations and the area of the crown of the lower first molar to determine body mass estimates of fossil mammals. The means of molar areas were used to calculate the final body mass values. Where possible, body mass estimates were obtained from published sources. The body sizes were recorded in grams and subsequently logarithmically transformed.

Discontinuity Analysis

Body mass pattern or structure refers to the distribution of body mass aggregations and gaps along the body mass axis. Body mass distributions were analyzed for discontinuities using simulations of actual data compared to a null distribution (a continuous unimodal kernel distribution of the log-transformed data (Restrepo et al. 1997). A body-mass aggregation consisted of three or more species with body masses that did not exceed the expectation of the null distribution and was defined by the upper and lower extremes of the aggregation (Allen et al. 1999). Gaps in body-mass aggregations were defined as significantly large areas between adjacent body masses that exceeded the expectation of the null distribution (Forys and Allen 2002). Significance of discontinuities in the data was determined by calculating the probability that the observed discontinuities were chance events (compared observed values with output of 1,000 simulations run against the null set (Restrepo et al. 1997). Two other methods were also used to confirm the location of discontinuities. I used Bayesian Classification and

Regression Trees (Chipman et al. 1998) and hierarchical cluster analysis (SAS Institute 1999) to further validate any patterns detected in body mass distributions. A multiple method approach in detecting significance in body mass patterns has been suggested as the best protocol (Allen et al. 2006, Stow et al. 2007).

Chi-square and Phi correlation analyses were used to determine differences in body mass patterns between geologic layers. The null hypothesis of the Chi-square analyses is that there are equal distributions among each of the four cells in a 2 x 2 table (one in the gap condition while the other is in the lump condition and vice versa, both in the lump condition and both in the gap condition), which would indicate that the two groups are unrelated. Gap space between body mass aggregations was calculated in two ways; total number of gaps (a more liberal approach) and the actual, calculated gap space (a more conservative approach). That is, a liberal approach would be limited by the smallest and largest species in each comparison and a conservative approach would be limited by the range of shared data between the two groups. A Bonferroni correction was applied to account for multiple comparisons in the same analysis (Miller 1981).

In order to provide additional information, a correlation of binary variables, using Phi correlation, was also utilized in order to elucidate more information and to determine the relationship between the different reconstructed paleo faunas. Species within each faunal zone were assigned a binary variable (where 1 represented a body mass aggregation and 0 represented a gap), based on the log₁₀ body mass axis, and divided into a 0.01 gram increment bin. The Phi coefficient ranges from -1 to 1. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a

negative correlation. Fleiss et al. (2003) suggested that values greater than ± 0.30 indicate a strong relationship.

Speciation Analysis

After completing the discontinuity analyses, new species placement (based on adjacent chronological faunas) within these distributions was determined. New species could be distributed in the body-mass patterns in various ways; randomly, within a limited range of body masses, at the edge of aggregations, only within body mass aggregations or only in the gaps (Figure 2.1). Both gaps and edges of body mass distributions represent changes in scale and areas of high variability (see Allen et al. 1999). Chi-square and t-test analyses were used to determine the significance of the locations of new species within each FAZ.

Chi-square goodness of fit testing was used to determine if new species occurred in gaps between body mass aggregations in the observed distribution more often than expected values if their distribution were random. As in the discontinuity analysis, gap space between body mass aggregations was calculated in two ways; total number of gaps (a more liberal approach) and the actual, calculated gap space (a more conservative approach). Wilcoxon two-sample tests were then used to determine the distance to body mass aggregation edge of new species compared to those species already present. T-tests were also used to determine the distance to body mass aggregation edge of new species compared to those species present in the prior geologic layer.

RESULTS

Lump/Gap Structure

Liberal Approach

Sixteen of 28 pairwise comparisons were significant (4 unique to this approach) in the liberal chi-square analysis and 8 of 28 pairwise comparisons were significant (3 unique to this approach) in the Phi correlation analysis (Tables 4.1 & 4.2). Bridger FAZ 1, 2, 3 and 4 were all similar to each other and Phi coefficients between Bridger FAZ 1 and 3, Bridger FAZ 1 and 4, Bridger FAZ 2 and 3 and Bridger FAZ 3 and 4 indicated strong positive relationships. Bridger FAZ 1 and 5 were similar to each other and the Phi coefficient indicated a strong positive relationship. Bridger FAZ 2 and 5 were similar to each other, and the Phi coefficient indicated a strong negative relationship. Bridger FAZ 3 and 4 were not similar in body mass pattern to Bridger FAZ 5.

Bridger FAZ 2, 3 and 5 were not similar in body mass pattern to Uinta FAZ 1. Bridger FAZ 1 and 4 were similar to Uinta FAZ 1, however low Phi coefficient values for each comparison indicated no strong relationships were evident. Bridger FAZ 5 was similar to Uinta FAZ 2, however a low Phi coefficient indicated no strong relationship. Bridger FAZ 1 and 5 were similar to Uinta FAZ 3 and Phi coefficients for each comparison indicated strong negative relationships. Bridger FAZ 4 was similar to Uinta FAZ 3, however a low Phi coefficient indicated no strong relationship. Bridger FAZ 3 and Uinta FAZ 1 were not similar in body mass pattern to Uinta FAZ 3. Uinta FAZ 1 was similar to Uinta FAZ 2, however a low Phi coefficient for the comparison indicated no strong relationship. Uinta FAZ 1 was not similar in body mass pattern to Uinta FAZ 3. Uinta FAZ 2 was similar to Uinta FAZ 3, however a low Phi coefficient indicated no strong relationship.

Conservative Approach

Fourteen of 28 pairwise comparisons were significant (2 unique to this approach) in the conservative chi-square analysis and 7 of 28 pairwise comparisons were significant (2 unique to this approach) in the Phi correlation analysis (Tables 4.3 & 4.4). Bridger FAZ 1, 2, 3 and 4 were all similar to each other and Phi coefficients for each comparison indicated strong positive relationships, except in the Bridger FAZ 2 versus Bridger FAZ 4 comparison. Bridger FAZ 1 was similar to Bridger FAZ 5, with a strong positive Phi coefficient. Bridger FAZ 2 was similar to Bridger FAZ 5, but the Phi coefficient indicated no strong relationship. Bridger FAZ 3 and 4 were not related to Bridger FAZ 5.

Bridger FAZ 1 was similar to Uinta FAZ 1 and had a Phi coefficient that indicated a strong positive relationship. Bridger FAZ 4 was also similar to Uinta FAZ 1, however a low Phi coefficient indicated no strong relationship. Bridger FAZ 2, 3 and 5 were not similar in body mass pattern to Uinta FAZ 1. Bridger FAZ 1, 4 and 5 were not similar in body mass pattern to Uinta FAZ 2. Bridger FAZ 2 and 3 were similar to Uinta FAZ 2, however Phi coefficients for each comparison indicated no strong relationships. Bridger FAZ 1 and 5 were not similar in body mass pattern to Uinta FAZ 3. Uinta FAZ 1 was not similar to Uinta FAZ 3. Uinta FAZ 2 and 3 were similar, however Phi coefficients for each comparison indicated no strong relationships.

Speciation

New species of mammals ($X^2=9.06$, 6 *df*; $P = 0.17$) did not occur at body mass aggregation edges in the observed distribution more often than expected. Based on calculated gap space, new species mammals ($X^2=1.85$, 6 *df*; $P = 0.93$) did not occur at body mass aggregation edges in the observed distribution more often than expected (Table 4.5). The distance to edge of new species was not less than those species already

present (Table 4.6). However, when new species were placed into the prior geologic layer, the distance to edge of new species was significantly less than those species already present in Bridger Zones 1 and 4 (Table 4.7).

DISCUSSION

Discontinuous body-mass distributions were found in all Bridger and Uinta FAZs examined. Discontinuities in body mass distributions that have been constructed from the fossil record have also been shown in Miocene mammals (Lambert 2006) and Pleistocene mammals (Lambert & Holling 1998). Discontinuities in body mass distributions have also been shown in North American birds (Skillen and Maurer 2008), south Florida herpetofauna, birds, and mammals (Allen 2006, Allen et al. 1999), tropical forest birds (Restrepo et al. 1997), boreal region birds and mammals (Holling 1992) and in various other taxa (reviewed in Sendzimir et al. 2003). This analysis extends these conclusions to include the Bridger and Uinta FAZs in southwestern Wyoming and northeastern Utah.

New species of mammals did not occur at body mass aggregation edges in the observed distribution more often than expected in either analysis conducted. The distance to edge of new species was not less than those species already present and was only significant in two FAZs when new species were placed into the prior geologic layer. There are numerous plausible explanations as to why speciation events in these FAZs were not near discontinuities. Sample sizes in 7 of the 10 FAZs were fewer than 30 species. A small sample size in statistical analyses can inflate the probability of committing Type I and Type II errors (Morrison 1988). In a chi-square goodness-of-fit test, the approximation of the sampling distribution of the test statistic improves as the

sample size increases (Ott & Longnecker 2001). This may also be due to the FAZ being in a state of transition or species could be exhibiting a delayed response to a past change in the environment (Alroy et al. 2000, Skillen & Maurer 2008). It has been suggested that North American mammal fossil record diversity has been inflated by 32 to 44% due to taxonomic biases which could affect speciation and extinction rates on both absolute and relative terms (Alroy 2002). There is also some debate as to the validity of “land-mammal ages” due to the diachrony inherent in the North American mammal fossil record as a result of undersampling (Alroy 1998). Another factor that may explain why speciation events were not located near discontinuities is taxonomic discrepancies. Alroy (2003) suggests that taxonomic discrepancies in the North America mammal fossil record are correlated with body mass, whereby small bodied species’ names are highly unreliable.

My results do not entirely correspond to the habitat reconstructions of Townsend (2004) and there was little agreement between the results of my liberal and conservative approaches to body mass distribution analysis. Bridger FAZ 1 and 2 were reconstructed as open woodland habitats and Bridger FAZ 3 and 4 were reconstructed as dense forest habitats, however, Chi-square analyses indicate that the body mass distributions of all four FAZs were similar and the corresponding Phi correlations were strong positive relationships. This suggests that these habitats were more alike than previously thought. My results parallel the taxonomic makeup of these FAZs. Bridger FAZ 1, 2, and 3 all exhibited similar patterns of ordinal diversity: primates, rodents, insectivores and perissodactyls were dominant. The Bridger FAZ 4 was somewhat different, with primates

and perissodactyls being the most dominant, but my results suggest not dominant enough to significantly change the faunal assemblage makeup.

Bridger FAZ 3 and 4 body mass distributions were not similar to the Bridger FAZ 5 body mass distribution and this evidence potentially supports the general view that Bridger FAZ 5 may be earliest Uintan in age (Evanoff et al. 1994). However, in the conservative approach, the Bridger FAZ 5 body mass distribution was not similar to Uinta FAZ 1 body mass distribution. Bridger FAZ 5 had a somewhat similar pattern of ordinal diversity as Bridger FAZ 3 and 4, but had fewer perissodactyls than any other Bridgerian FAZ. Thus, our results suggest that Bridger FAZ 5 may not be earliest Uintan in age and may represent a transition zone (Townsend 2004) between the Bridgerian and Uintan ages.

Bridger FAZ 1 (open woodland) and Bridger FAZ 5 (forest) had different habitat reconstructions, however, Chi-square analyses indicate that the body mass distributions were similar and the corresponding Phi correlation was a strong positive relationship. These two FAZs had similar patterns of ordinal diversity, both dominated by primates, rodents, insectivores and perissodactyls. The Bridger FAZ 5 had fewer perissodactyls than the Bridger FAZ 1, but my results suggest not deficient enough to significantly change the faunal assemblage makeup.

The Bridger FAZ 2 (open woodland) and 3 (dense forest) body mass distributions were not similar to the Uinta FAZ 1 (woodland) body mass distribution, which suggests that these two habitats were indeed different than the Uinta FAZ 1 habitat. Under the conservative approach, the Bridger FAZ 1 (open woodland) and Bridger FAZ 5 (forest) body mass distributions were not similar to the Uinta FAZ 3 (open) body mass

distribution, however under the liberal approach they were similar with a strong negative relationship.

Uinta FAZ 1 was reconstructed as woodland habitat and Uinta FAZ 2 and 3 were reconstructed as open habitats, however, conservative and liberal Chi-square analyses indicate that the body mass distributions of FAZ 1 and 2 and FAZ 2 and 3 were similar with no strong Phi correlations and Uinta FAZ 1 and 3 body mass distributions were not similar. Thus, my results concerning the similarity of body mass distributions between the Uinta FAZs are inconclusive, yet similar in both conservative and liberal approaches. My results may have been inconclusive due to the difficulty to distinguish qualitatively distinct mechanisms using time-slice paleontological data (Alroy 2000). Raup & Sepkoski (1984) suggest periodicity in extinctions of the geologic past, so the data I examined may not have been at a time with a sufficient number of extinctions.

Although some of my results describing the body mass distributions are in agreement with the habitat reconstructions of Townsend (2004), a majority of my results are not. These inconsistencies between the habitat reconstructions and the body mass distributions are most likely due to the fact that the species assemblages in each zone were time-averaged and that the fossil mammals came from multiple localities within a stratigraphic range, which likely resulted in a broad interpretation of habitat and a potentially imprecise reconstruction (Townsend 2004). Another possible explanation could be the methodology used by Townsend (2004) to reconstruct these faunal zones. Phylogeny reconstruction using morphological data has been a hotly debated subject in recent years (see Scotland et al. 2003 & Jenner 2004).

Biological systems are complex and their evolution can be influenced by internal (e.g; competition, predation), external (e.g; climate, human perturbation) and historical processes which most likely work together over the long-term to shape any given ecological system at any given moment in time (MacFadden 2000). This study identified and analyzed patterns in body mass distributions within paleoecological faunal assemblage zones, but the use of body mass may have been too coarse to detect speciation events. Sepkoski (1998) suggest periodicity in speciation events of the geologic past, so the data I examined may not have been at a time with a sufficient number of speciation events.

Table 4.1. Chi-Square probability levels for each pairwise comparison of Bridger and Uintan faunal assemblage zones using liberal data selection. Probability levels less than or equal to 0.001 were considered significant matches between the lump and gap architecture of the compared faunal assemblage zones.

Geologic Layer	Bridger Zone 2	Bridger Zone 3	Bridger Zone 4	Bridger Zone 5	Uinta Zone 1	Uinta Zone 2	Uinta Zone 3
Bridger FAZ 1	<.0001*	<.0001*	<.0001*	<.0001*	0.0065	0.0026	<.0001*
Bridger FAZ 2	---	<.0001*	0.0006*	<.0001*	0.5503	0.0473	0.0062
Bridger FAZ 3	---	---	<.0001*	0.2971	0.4595	0.0216	0.9860
Bridger FAZ 4	---	---	---	0.4131	0.0009*	0.7570	0.0001*
Bridger FAZ 5	---	---	---	---	0.0107	0.0007*	<.0001*
Uinta FAZ 1	---	---	---	---	---	0.0012*	0.6403
Uinta FAZ 2	---	---	---	---	---	---	<.0001*

Table 4.2. Phi coefficients for each pairwise comparison of Bridger and Uintan faunal assemblage zones using liberal data selection.

Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared faunal assemblage zones.

Geologic Layer	Bridger Zone 2	Bridger Zone 3	Bridger Zone 4	Bridger Zone 5	Uinta Zone 1	Uinta Zone 2	Uinta Zone 3
Bridger FAZ 1	0.2111	0.4170*	0.5083*	0.3111*	0.1295	-0.1434	-0.2860*
Bridger FAZ 2	---	0.3229*	0.1710	-0.2939*	-0.0297	0.0987	-0.1362
Bridger FAZ 3	---	---	0.6040*	-0.0509	-0.0375	0.1165	0.0009
Bridger FAZ 4	---	---	---	-0.0399	0.1727	0.0160	-0.2026
Bridger FAZ 5	---	---	---	---	-0.1236	-0.1638	-0.2889*
Uinta FAZ 1	---	---	---	---	---	0.1716	0.0247
Uinta FAZ 2	---	---	---	---	---	---	0.2142

Table 4.3. Chi-Square probability levels for each pairwise comparison of Bridger and Uintan faunal assemblage zones using conservative data selection. Probability levels less than or equal to 0.001 were considered significant matches between the lump and gap architecture of the compared faunal assemblage zones.

	Bridger Zone	Bridger Zone	Bridger Zone	Bridger Zone	Uinta Zone	Uinta Zone	Uinta Zone
Geologic Layer	2	3	4	5	1	2	3
Bridger FAZ 1	<.0001*	<.0001*	<.0001*	<.0001*	<.0001*	0.2957	0.1664
Bridger FAZ 2	---	<.0001*	<.0001*	0.0002*	0.2318	<.0001*	0.0104
Bridger FAZ 3	---	---	<.0001*	0.5428	0.2613	<.0001*	0.0024
Bridger FAZ 4	---	---	---	0.1765	<.0001*	0.0948	0.0138
Bridger FAZ 5	---	---	---	---	0.4552	0.7501	0.1755
Uinta FAZ 1	---	---	---	---	---	0.0012*	0.0377
Uinta FAZ 2	---	---	---	---	---	---	0.0004*

Table 4.4. Phi coefficients for each pairwise comparison of Bridger and Uintan faunal assemblage zones using conservative data selection. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Values greater than ± 0.30 indicate a strong relationship between the lump and gap architecture of the compared faunal assemblage zones.

	Bridger Zone	Bridger Zone	Bridger Zone	Bridger Zone	Uinta Zone	Uinta Zone	Uinta Zone
Geologic Layer	2	3	4	5	1	2	3
Bridger FAZ 1	0.3370*	0.6056*	0.7385*	0.3763*	0.3503*	0.0552	-0.0929
Bridger FAZ 2	---	0.3818*	0.2560	-0.1877	0.0631	0.2291	-0.1720
Bridger FAZ 3	---	---	0.6768*	0.0311	0.0598	0.2477	0.2033
Bridger FAZ 4	---	---	---	0.0709	0.2344	0.0893	0.1653
Bridger FAZ 5	---	---	---	---	0.0397	0.0170	0.0909
Uinta FAZ 1	---	---	---	---	---	0.1716	-0.1395
Uinta FAZ 2	---	---	---	---	---	---	0.2373

Table 4.5. Results of Chi-Square analyses and standard deviations (SD) of Bridger and Uintan faunal assemblage zones comparing the observed distribution of new species in gaps between body mass aggregations.

Faunal Assemblage Zone	Expected based on Gaps	Expected based on Vector Space	Observed
Bridger FAZ 2	1.45	1.91	2
Bridger FAZ 3	2.48	5.06	4
Bridger FAZ 4	0.53	1.02	1
Bridger FAZ 5	3.65	5.27	6
Uinta FAZ 1	4.80	6.79	8
Uinta FAZ 2	2.69	3.38	4
Uinta FAZ 3	2.29	3.08	5
χ^2	9.06(0.17)	1.85(0.93)	

Table 4.6. Results of t-tests comparing mean (x) and standard deviation (SD) of Bridger and Uintan faunal assemblage zones comparing mean (x) distance to edge of mammals that are new species and species that remained from the previous geologic layer.

Geologic Layer	Remained	New Species	z	P
	x(SD)	x(SD)		
Bridger FAZ 2	0.070(0.099)	0.082(0.082)	0.525	0.300
Bridger FAZ 3	0.075(0.090)	0.070(0.070)	0.000	0.500
Bridger FAZ 4	0.057(0.081)	0.109(0.154)	0.294	0.384
Bridger FAZ 5	0.065(0.092)	0.133(0.147)	-0.896	0.185
Uinta FAZ 1	0.060(0.102)	0.070(0.080)	-0.182	0.428
Uinta FAZ 2	0.061(0.077)	0.082(0.083)	0.500	0.309
Uinta FAZ 3	0.041(0.047)	0.040(0.062)	-0.536	0.296

Table 4.7. Results of t-tests comparing mean (x) and standard deviation (SD) of Bridger and Uintan faunal assemblage zones comparing mean (x) distance to edge of mammals that are new species and species that are from the previous geologic layer.

Geologic Layer	Prior Geologic Layer	New Species		
	x(SD)	x(SD)	z	P
Bridger FAZ 2 to Bridger FAZ 1	0.110(0.112)	0.04(0.065)	-1.695	0.045*
Bridger FAZ 3 to Bridger FAZ 2	0.070(0.094)	0.090(0.081)	0.986	0.162
Bridger FAZ 4 to Bridger FAZ 3	0.070(0.080)	0.130(0.134)	0.848	0.198
Bridger FAZ 5 to Bridger FAZ 4	0.060(0.087)	0.050(0.108)	-1.538	0.062*
Uinta FAZ 1 to Bridger FAZ 5	0.110(0.132)	0.070(0.122)	-0.928	0.177
Uinta FAZ 2 to Uinta FAZ 1	0.060(0.081)	0.080(0.080)	0.487	0.313
Uinta FAZ 3 to Uinta FAZ 2	0.070(0.079)	0.040(0.064)	-0.791	0.214

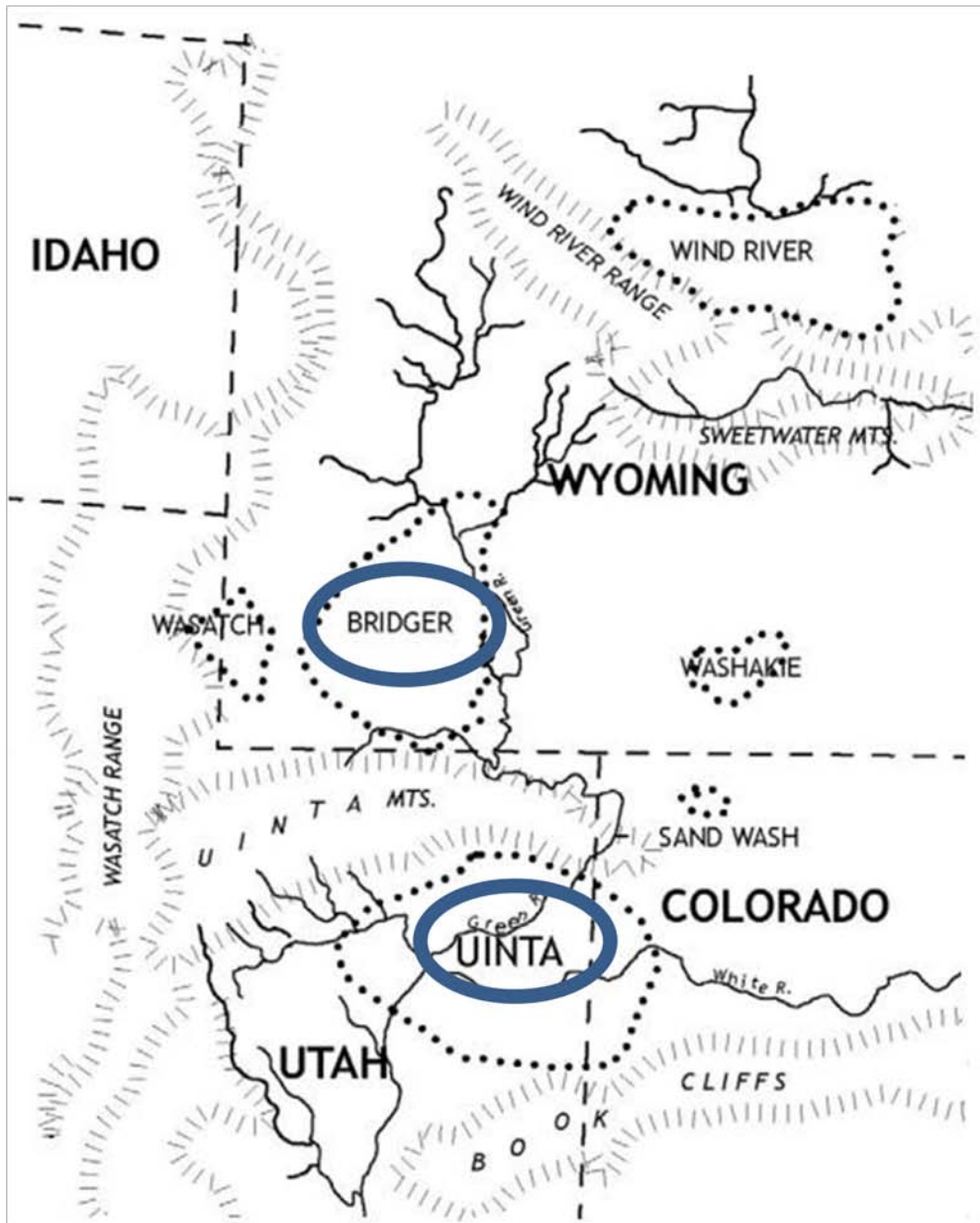


Figure 4.1. Map of Green River Basin (labeled as Bridger) and Uinta Basin. Basins are outlined in stipple and mountain ranges are outlined with hatch marks. Image adapted from Townsend (2004).

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CHAPTER 5: CHANGES IN BODY MASS DISCONTINUITIES WITH CHANGES IN SCALE

Landscapes form hierarchies that are structured by vegetative, geomorphologic and contagious disturbance processes (Holling 1992). These hierarchies are linked to the availability of resources throughout the landscape and structurally different landscapes will have differently scaled resources (Skillen & Maurer 2008). Ecological patterns observed at one scale cannot usually be extrapolated to other scales (Gaston & Blackburn 1999). The spatial and temporal patterns inherent in landscapes may reflect numerous processes, interacting on distinct scales, which potentially shape the assembly of animal communities and has been the subject of much debate.

In order to understand these patterns, one must be able to understand interactions taking place between organisms and each other and organisms with their environment. Studies have suggested predation (Winemiller 1989), age class (Yamahira et al. 1996) and diet (Siaw-Yang 1988) affect community structure at a local scale. Tropho-dynamics and productivity could reinforce clumping in body-mass distributions by increasing amplitude of the clumps (Holling 1992). Oksanen et al. (1979) suggest body mass discontinuities are caused by interspecific aggression modulated by habitat structure. There are conflicting results as to the relationship between habitat architecture and observed body mass patterns. Leaper et al. (2001) suggest that habitat architecture may not be as closely related to body-size patterns at a very fine, marine benthos scale, whereas Schwinghamer (1981) suggests a relationship.

Other studies have demonstrated the influence of community interactions on body mass distributions (Oksanen et al. 1979, Stubblefield et al. 1993, and Nummi et al. 2000). These studies support Hutchinson's (1959) community interaction hypothesis, which states that in the process of community formation species may be displaced, unfilled niches may be occupied, and niches may be portioned. Community interaction may become important only at a local scale, once community structure (under the premise of the textural discontinuity hypothesis) has formed (Allen et al. 2006). Historical processes that occur over evolutionary time have a major role in determining the species composition of a community (Mouquet et al. 2003). Rodriguez et al. (2006) also suggest environmental factors, such as climate, along with chance events play a role in the development of community structure.

Analysis of body mass patterns have been suggested as methods to provide insight about these underlying processes, as they are important in ecosystem level biotic structure (Forys & Allen 2002). Many studies have reported a discontinuous body mass distribution reflecting the architecture of the landscape (Restrepo et al. 1997, Raffaelli et al. 2000, Havlicek & Carpenter 2001). These studies support Holling's (1992) textural discontinuity hypothesis, which states that hierarchical landscape structures with scale-specific pattern entrain attributes of animals inhabiting the landscape.

Organisms of different body sizes have different requirements for resources and operate at different spatiotemporal scales (West et al. 1997). Therefore, an animal cannot simultaneously interact with multiple scales, but has to specialize at a single scale or shift between two scales (Allen & Saunders 2002). Allen et al. (1999) suggest that the gaps between body mass aggregations represent scale breaks and that these points in the body

mass distribution are highly susceptible to change in structure and ecological processes. Variations in body size may not only reflect cross-scale structure, but may also operate over micro-, meso-, and macroscales. In order to test Holling's (1992) textural discontinuity hypothesis, I determined whether discontinuous body mass patterns existed within each ecoregion, biome, continental and hemispheric scale. A lack of discontinuous body mass patterns would provide strong evidence against the textural discontinuity hypothesis.

Ecoregions are defined as large units of land which contain unique assemblages of species sharing similar environmental conditions (Olson et al. 2001). Ecoregions within biomes, or communities in the same biogeographical region, provide replicated examples of ecosystems that should be similar in history (Rodriguez et al. 2006). Thus, each biome enables the comparison of scale specific vertebrate community structure and the ecoregions within them should exhibit similar body mass patterns. In order to test Holling's (1992) textural discontinuity hypothesis, I examined whether the vertebrate body mass structures are similar among the ecoregions within each biome. A lack of similarity would provide strong evidence against the textural discontinuity hypothesis. If body mass structure reflects landscape structure at these different scales, this will support the textural discontinuity hypothesis and create a fundamental link between landscape and community ecology.

METHODS

I examined two terrestrial vertebrate groups: birds and mammals of the contiguous western hemisphere. Islands were not included in this study due in part to the

highly variable body size of animals on islands (Foster 1964). Humans were not included in mammal data because “the spatial grain and ambit of the human is restricted not by body-mass class, but by technological innovation” (Holling 1992). Digital distribution maps of birds (Ridgely et al. 2007) and mammals (Patterson et al. 2007) were obtained from NatureServe, Arlington, VA. Digital distribution maps of birds and mammals were also obtained online from the World Wildlife Fund (WWF) (Olson et al. 2004). Data sets were combined and duplicate entries were removed in order to obtain a more accurate distribution database. Ecoregions, biomes, and continents were delineated according to WWF protocol (Figure 5.1, Olson et al. 2001).

I examined mammal and bird body mass distributions at four distinct scales. I define scale as a range of spatial and temporal frequencies (see Peterson et al. 1998). The ecoregion scale included regions that ranged from $\sim 2,000 \text{ km}^2$ to $\sim 800,000 \text{ km}^2$. The biome scale included areas that ranged from $\sim 100,000 \text{ km}^2$ to $\sim 9,000,000 \text{ km}^2$. The continental scale included North America ($\sim 20,800,000 \text{ km}^2$) and South America ($\sim 17,800,000 \text{ km}^2$). The western hemisphere scale was the combination of North and South America’s land masses (km^2).

Body-mass estimates

In most cases, body mass estimates for birds were compiled from Dunning (1993) and estimates for mammals were obtained from Silva and Downing (1995). Where possible, body mass estimates from the region of interest were used, otherwise estimates were taken from the nearest geographic location. Male and female body mass estimates were averaged when weights for both sexes are provided. Body sizes were recorded in grams and subsequently logarithmically transformed.

Discontinuity Analysis

Body mass pattern or structure refers to the distribution of body mass aggregations and gaps along the body mass axis. Body mass distributions were analyzed for discontinuities using simulations of actual data compared to a null distribution (a continuous unimodal kernel distribution of the log-transformed data (Restrepo et al. 1997). A body-mass aggregation consisted of three or more species with body masses that did not exceed the expectation of the null distribution and was defined by the upper and lower extremes of the aggregation (Allen et al. 1999). Gaps in body-mass aggregations were defined as significantly large areas between adjacent body masses that exceeded the expectation of the null distribution (Forys and Allen 2002). Significance of discontinuities in the data was determined by calculating the probability that the observed discontinuities were chance events (compared observed values with output of 1,000 simulations run against the null set (Restrepo et al. 1997).

The number of species in my data sets vary from < 30 to > 150 , therefore I maintained a constant statistical power of approximately 0.50 when setting alpha for detecting discontinuities (Lipsey 1990). Two other methods were also used to confirm the location of discontinuities. I used Bayesian Classification and Regression Trees (Chipman et al. 1998) and hierarchical cluster analysis (SAS Institute 1999) to further validate any patterns detected in body mass distributions. A multiple method approach in detecting significance in body mass patterns has been suggested as the best protocol (Allen et al. 2006, Stow et al. 2007). Abundance data for each individual species was not integrated into my analyses. Changes in dominance of species (greater numbers of individuals) could change aggregation and gap locations, but body mass distribution

patterns will be conserved despite changes in species composition or number (Havlicek & Carpenter 2001).

Body Mass Distribution Pattern Analysis

Differences between taxa in the average gap size and in the average number of body mass aggregations, at each scale, were tested with a Mann-Whitney Rank Sum test because the data were not normally distributed. Correlations between the number of species in an ecosystem and the number of body mass aggregations were tested with a Pearson Product Moment Correlation (if the data was normally distributed) and a Spearman Rank Order Correlation (if the data was not normally distributed).

I tested for similarities in body mass distributions found within systems of similar landscape structure (biomes) by implementing a bootstrapping method (Chernick 2008) using R statistical software (R 2010). I wanted to determine whether the ecoregions within each biome were more similar to each other than by chance. All comparisons were made first by dividing the log₁₀ body mass axis into 0.001 increments. Then, each increment was assigned either a 0 or 1 condition, with 0 representing a gap and 1 representing a body mass aggregation. Upper and lower limits were determined by the largest and smallest body masses in the comparison. The sum of observations that were within each body mass aggregation across each of the ecoregions within a biome were calculated. Then, the variance of all the sums of observations across ecoregions, in each biome, was calculated to determine the observed variance.

Each biome contained a unique number of ecoregions and that number determined the size of each resample. Each biome was resampled with replacement, 1,000 times, to determine the approximate distribution or simulated variance. If the

simulated variance is the same as the observed (actual) variance, then the ecoregions in that particular biome are not similar to each other because simply selecting a random assortment of all possible ecoregions in the western hemisphere would result in a similar variance. The exact rank, or location of the observed variance, within the 1,000 simulated variances was used to determine if the simulated versus observed variances were significantly different. An exact rank of 950 or higher ($\alpha = 0.05$) was the level of significance. A binomial distribution was calculated in order to determine whether all the observed variances for ecoregions within each biome were higher than the simulated variances by chance alone.

A Phi correlation analysis was used to determine differences in ecoregion body mass distributions between biomes. Species in each ecoregion within a biome were assigned a binary variable (where 1 represented a body mass aggregation and 0 represented a gap), based on the log₁₀ body mass axis, and divided into a 0.001 gram increment bin. The Phi coefficient ranges from -1 to 1. Values approaching 1 indicate a positive correlation and values approaching -1 indicate a negative correlation. Fleiss et al. (2003) suggested that values greater than ± 0.30 indicate a strong relationship.

I analyzed the temperate broadleaf and mixed forest biome, boreal forests and taiga biome, temperate grasslands, savannas and shrublands biome, and the deserts and xeric shrublands biome. These four biomes were selected due to the similar number of ecoregions within each biome. I compared the mean phi correlation of ecoregions within one biome to the mean phi correlation of ecoregions in a different biome. A Kruskal-Wallis One Way Analysis of Variance on Ranks was used to determine significant differences in mean phi correlation values.

RESULTS

All bird and mammal body-mass distributions were discontinuous at the ecoregion, biome, continental, and hemisphere scales. I observed discontinuities in 253 mammal and 263 bird communities in 14 biomes located throughout the Western Hemisphere. There were between 6 - 20 body mass aggregations in each community. The numbers of body mass aggregations were positively correlated to the number of mammal species in each ecoregion within 12 out of 14 biomes (Table 5.1) and in each ecoregion within 11 out of 14 biomes in birds (Table 5.2). The numbers of body mass aggregations were positively correlated to the number of species in each biome, in mammals (Correlation Coefficient = 0.943, $p < 0.001$) and in birds (Correlation Coefficient = 0.968, $p < 0.001$). At the continental scale both taxa were combined and the numbers of body mass aggregations were positively correlated to the number of species (Correlation Coefficient = 0.997, $p = 0.003$).

In general, mammal communities at the ecoregion scale, had larger gaps on average than bird communities ($p = < 0.001$) (Table 5.3). Bird communities at the ecoregion scale, had more discontinuities than mammal communities as shown by a higher number of aggregations on average ($p = < 0.001$) (Table 5.4). When all mammal and bird body masses were combined into one distribution for each biome (i.e. biome scale), the average size of gaps in mammals was larger than birds ($p = < 0.001$). The average gap size in mammal body mass distributions at the biome scale were smaller than the gap size found at the ecoregion scale ($p = < 0.001$) (Table 5.5). At the biome scale, the average gap size in bird body mass distributions was smaller than the gap size at the

ecoregion scale ($p = 0.004$) (Table 5.6). The number of body mass aggregations were higher at the biome scale than in the ecoregion scale in both mammals ($p = < 0.001$) and birds ($p = 0.006$). As in the ecoregion and biome level scales, bird body mass aggregations had a smaller gap size ($p = 0.330$) and more aggregations ($p = 0.330$) than mammals at the continental scale, but were not significantly different (Table 5.7). At the hemisphere scale, both birds and mammals had equal size gaps, but birds had more aggregations (Table 5.8).

In mammals and birds, as scale increased from ecoregion to biome, the number of body mass aggregations increased (Figure 5.2). As scale increased from biome to continent in mammals, the number of body mass aggregations observed in both North America and South America equaled the highest number of body mass aggregations observed in the biome scale. In birds, as scale increased from biome to continent, the numbers of body mass aggregations in both North America and South America were as high as the upper half of body mass aggregations observed in the biome scale. In mammals, the highest numbers of body mass aggregations were observed at the hemisphere scale. In birds, at the hemisphere scale, the numbers of body mass aggregations observed were similar to the numbers of body mass aggregations observed at the South America continent scale, but higher than the numbers of body mass aggregations observed at the North America continent scale.

The observed variance in the sum of observations that were within each bird body mass aggregation across each of the ecoregions within a biome were higher than the observed variance in the sum of observations that were within each mammal body mass aggregation across each of the ecoregions within a biome (Table 5.9). The observed

variances in the sum of observations that were within each mammal body mass aggregation across each of the ecoregions within a biome were higher than the simulated variances in every biome and there was less than a 0.006% chance it is not a random effect (Table 5.10). In 9 out of 14 biomes, the simulated variance is not the same as the observed variance in the sum of observations that were within each mammal body mass aggregation across each of the ecoregions, which indicates that the ecoregions in those particular biomes are similar to each other. The observed variances in the sum of observations that were within each bird body mass aggregation across each of the ecoregions within a biome were higher than the simulated variances in all but three biomes and there was less than a 2% chance it is not a random effect (Table 5.11). In 7 out of 14 biomes, the simulated variance is not the same as the observed variance in the sum of observations that were within each bird body mass aggregation across each of the ecoregions, which indicates that the ecoregions in those particular biomes are similar to each other.

Body mass distributions of ecoregions within the temperate broadleaf and mixed forest biome were more similar to each other than to the body mass distributions of ecoregions within the boreal forests and taiga biome, temperate grasslands, savannas and shrublands biome, and the deserts and xeric shrublands biome. Body mass distributions of ecoregions within the boreal forests and taiga biome were more similar to each other than to the body mass distributions of ecoregions within the temperate grasslands, savannas and shrublands biome, and the deserts and xeric shrublands biome. There was no difference in body mass distributions of ecoregions within the temperate grasslands,

savannas and shrublands biome compared to the body mass distributions of ecoregions within the deserts and xeric shrublands biome (Table 5.12).

DISCUSSION

Discontinuous body-mass distributions were found in all bird and mammal communities that were examined. Discontinuities in body mass distributions have also been shown in North American birds (Skillen and Maurer 2008), south Florida herpetofauna, birds, and mammals (Allen 2006, Forsy & Allen 2002), Pleistocene and Miocene mammals (Lambert 2006, Lambert and Holling 1998), tropical forest birds (Restrepo et al. 1997), and boreal region birds and mammals (Holling 1992) and in various other taxa (reviewed in Sendzimir et al. 2003). This analysis extends these conclusions to include bird and mammal communities in almost every ecosystem of the western hemisphere.

As scale increased from ecoregion to hemisphere, each body mass distribution pattern was very distinct. There were no general patterns observed in any body mass distributions across scales or taxa. In some instances, some gaps and/or body mass aggregations remained at or near the exact location along the body mass axis as scale increased from ecoregion to hemisphere. In other cases, as scale increased, some gaps and/or body mass aggregations were observed at different locations along the body mass axis. Although there were no clear general patterns observed, gaps and/or body mass aggregations that remained at or near the same location along the body mass axis as scale increased may be the result of historical events. Marquet and Cofre (1999) observed a strong historic component, the Great American Biotic Interchange, which they suggest

explain the structure of multiple body mass modes in South American mammals at different spatial scales.

At the ecoregion, biome, continental, and hemisphere scales bird communities had more aggregations than mammal communities. Mammals had larger gap sizes than birds at every scale except hemispheric. These differences may be due to how each taxa perceive and interact with landscape structure. It has been suggested that birds live in a more three dimensional world and mammals live in a one dimensional world (Holling 1992). If this were the case, birds would be able to access a greater number of resources enabling the use of a more complex landscape structure and allow birds to occupy more niches. More niches could account for a greater number of aggregations, which would contradict the textural discontinuity hypothesis. TDH would posit that as a landscape becomes more structurally complex, the opportunity to fill new niches increases, not vice versa.

Birds and mammals also have different locomotory modes, affording birds the ability to utilize both terrestrial and aerial resources in any given landscape (Sendzimir 1998). Another explanation might be in our lack of understanding in the interaction between animals and the different structures in their environment. We cannot assume that different structural types in the landscape, in the eyes of an animal, are equally weighted in importance. Also, the body mass pattern analysis methodology used might not have been the appropriate manner to analyze the data at these various scales (Sendzimir 1998).

Only 64% of the biomes in mammals had ecoregions with similar structure and only 50% of the biomes in birds had ecoregions with similar structure. This lack of universal congruence between ecoregions within a biome, in both mammals and birds,

could be the result of the information put into the analysis. Current landscape classification schemes are presumably quite subjective in nature. They may also be too coarse and even exclude key components of a complex system (Strand 2011, Gallant 2009). If key components, or enough components in general, were excluded from one or more ecoregions within a biome then the similarity between ecoregions within a particular biome could be considerably diminished.

Phi correlation analyses comparing the body mass distributions of ecoregions between two different biomes indicated that the temperate broadleaf and mixed forest biome and boreal forests and taiga biome were unique among the biomes examined, which provides support that the ecoregions within these biomes were correctly classified into their respective biome categorization. Analyses comparing the body mass distributions of ecoregions between the temperate grasslands, savannas and shrublands biome with the body mass distributions of ecoregions within the deserts and xeric shrublands biome failed to detect any difference between them. As mentioned above, landscape classification schemes may be too coarse or exclude key components of a complex system which might have differentiated the body mass distributions of the ecoregions between these two biomes.

Ecosystems are a complex juxtaposition of numerous variables, many of which are poorly understood. Ecosystems are affected by different processes at different scales. Vegetative processes control the microscale, disturbance and environmental processes at the mesoscale and geomorphologic and evolutionary processes at the macroscale (Holling 1992). Analysis of animal-landscape interactions can now be used as an additional parameter when modeling ecosystems. Local community composition depends on local

and regional processes (Gaston & Blackburn 1999). Therefore, body mass distribution analysis at multiple scales will allow the identification of subsets of animal communities within subsets of the landscape structure.

This will provide a more refined and useful approach to the study of animal-landscape interactions. Processes unique at a specific landscape scale can be tied to animal communities that exist only at that scale, which makes the textural discontinuity hypothesis a key management tool. Correct use and knowledge of this paradigm shift in ecology will help simplify the complexity of nature so that effective conservation efforts can be realized.

Table 5.1. The correlation between body mass aggregations the number of mammal species in each ecoregion. The number of ecoregions within each biome, the correlation coefficient and p -values.

Biome	# of Ecoregions	Correlation Coefficient	P
1	71	0.853	0.001
2	26	0.911	0.001
3	9	0.953	0.001
4	19	0.573	0.011
5	29	0.859	0.001
6	15	0.709	0.003
7	8	0.856	0.002
8	19	0.568	0.011
9	6	0.683	0.136
10	9	0.939	0.002
11	10	0.449	0.172
12	4	0.937	0.063
13	22	0.906	0.001
14	6	0.939	0.017

Table 5.2. The correlation between body mass aggregations the number of bird species in each ecoregion. The number of ecoregions within each biome, the correlation coefficient and p -values.

Biome	Number of Ecoregions	Correlation Coefficient	P
1	72	0.921	0.001
2	27	0.943	0.001
3	9	0.962	0.001
4	19	0.781	0.001
5	30	0.73	0.001
6	17	0.917	0.001
7	8	0.913	0.001
8	19	0.775	0.001
9	6	0.765	0.103
10	9	0.973	0.001
11	15	0.939	0.001
12	4	0.889	0.111
13	22	0.962	0.001
14	6	0.96	0.002

Table 5.3. Mammal summary statistics, including standard deviations (SD), of all ecoregions within each biome.

The total number of ecoregions within each biome, the average distance of gaps between body mass aggregations within each biome, the total number of body mass aggregations within each biome and the average number of body mass aggregations within each biome.

Biome	Habitat	Number of Ecoregions	Average Distance of Gaps	Number of Aggregations	Average Number of Aggregations
1	Tropical and Subtropical Moist Broadleaf Forests	71	0.12(0.05)	618	8.70(1.21)
2	Tropical and Subtropical Dry Broadleaf Forests	26	0.15(0.06)	207	7.96(1.11)
3	Tropical and Subtropical Coniferous Forests	9	0.16(0.07)	71	7.89(1.45)
4	Temperate Broadleaf and Mixed Forests	19	0.23(0.04)	142	7.47(0.61)
5	Temperate Coniferous Forests	29	0.18(0.04)	214	7.64(0.87)
6	Boreal Forests/Taiga	15	0.28(0.05)	91	6.59(0.76)
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	8	0.10(0.04)	72	9(1.41)
8	Temperate Grasslands, Savannas and Shrublands	19	0.17(0.04)	150	7.89(0.81)
9	Flooded Grasslands and Savannas	6	0.15(0.05)	45	7.5(0.550)
10	Montane Grasslands and Shrublands	9	0.12(0.04)	76	8.44(1.33)
11	Tundra	10	0.32(0.08)	61	6.1(0.57)
12	Mediterranean Forests, Woodlands and Scrub	4	0.20(0.04)	29	7.25(0.96)
13	Deserts and Xeric Shrublands	22	0.18(0.09)	165	7.5(1.26)
14	Mangroves	6	0.10(0.03)	57	9.59(1.22)

Table 5.4. Bird summary statistics, including standard deviations (SD), of all ecoregions within each biome. The total number of ecoregions within each biome, the average distance of gaps between body mass aggregations within each biome, the total number of body mass aggregations within each biome and the average number of body mass aggregations within each biome.

Biome	Habitat	Number of Ecoregions	Average Distance of Gaps	Number of Aggregations	Average Number of Aggregations
1	Tropical and Subtropical Moist Broadleaf Forests	72	0.02(0.02)	1264	17.56(2.38)
2	Tropical and Subtropical Dry Broadleaf Forests	27	0.03(0.03)	437	16.19(3.01)
3	Tropical and Subtropical Coniferous Forests	9	0.03(0.01)	138	15.33(1.80)
4	Temperate Broadleaf and Mixed Forests	19	0.05(0.02)	217	12.06(0.54)
5	Temperate Coniferous Forests	30	0.05(0.01)	340	11.33(0.88)
6	Boreal Forests/Taiga	17	0.07(0.02)	176	10.35(1.17)
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	8	0.02(0.01)	145	18.13(2.23)
8	Temperate Grasslands, Savannas and Shrublands	19	0.04(0.01)	230	12.11(0.81)
9	Flooded Grasslands and Savannas	6	0.03(0.02)	88	14.67(1.97)
10	Montane Grasslands and Shrublands	9	0.05(0.03)	154	17.11(3.62)
11	Tundra	15	0.10(0.04)	122	8.13(1.64)
12	Mediterranean Forests, Woodlands and Scrub	4	0.06(0.02)	50	12.50(1.29)
13	Deserts and Xeric Shrublands	22	0.04(0.02)	307	13.95(2.21)
14	Mangroves	6	0.02(0.01)	102	17.00(2.00)

Table 5.5. All mammal species within each biome combined into one body mass distribution.

The average distance of gaps between body mass aggregations within each biome (including standard deviations (SD)), the total number of body mass aggregations within each biome and the total number of species within each biome.

Biome	Habitat	Average Distance of Gaps	Number of Aggregations	Number of Species
1	Tropical and Subtropical Moist Broadleaf Forests	0.02(0.03)	18	862
2	Tropical and Subtropical Dry Broadleaf Forests	0.02(0.02)	17	671
3	Tropical and Subtropical Coniferous Forests	0.06(0.06)	12	301
4	Temperate Broadleaf and Mixed Forests	0.07(0.04)	12	223
5	Temperate Coniferous Forests	0.08(0.06)	13	275
6	Boreal Forests/Taiga	0.15(0.06)	8	88
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	0.02(0.03)	15	457
8	Temperate Grasslands, Savannas and Shrublands	0.06(0.08)	14	346
9	Flooded Grasslands and Savannas	0.06(0.05)	11	230
10	Montane Grasslands and Shrublands	0.04(0.02)	14	421
11	Tundra	0.22(0.10)	7	64
12	Mediterranean Forests, Woodlands and Scrub	0.09(0.08)	9	161
13	Deserts and Xeric Shrublands	0.03(0.03)	17	584
14	Mangroves	0.05(0.04)	15	423

Table 5.6. All bird species within each biome combined into one body mass distribution. The average distance of gaps between body mass aggregations within each biome (including standard deviations (SD)), the total number of body mass aggregations within each biome and the total number of species within each biome.

Biome	Habitat	Average Distance of Gaps	Number of Aggregations	Number of Species
1	Tropical and Subtropical Moist Broadleaf Forests	0.01(0.01)	32	3304
2	Tropical and Subtropical Dry Broadleaf Forests	0.01(0.02)	28	2646
3	Tropical and Subtropical Coniferous Forests	0.02(0.02)	20	926
4	Temperate Broadleaf and Mixed Forests	0.03(0.03)	16	525
5	Temperate Coniferous Forests	0.02(0.03)	16	473
6	Boreal Forests/Taiga	0.05(0.04)	12	268
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	0.01(0.02)	25	1868
8	Temperate Grasslands, Savannas and Shrublands	0.02(0.02)	19	832
9	Flooded Grasslands and Savannas	0.02(0.03)	21	1231
10	Montane Grasslands and Shrublands	0.01(0.04)	27	1861
11	Tundra	0.06(0.04)	11	193
12	Mediterranean Forests, Woodlands and Scrub	0.03(0.03)	16	415
13	Deserts and Xeric Shrublands	0.01(0.02)	26	1868
14	Mangroves	0.01(0.03)	26	1845

Table 5.7. Species combined into one body mass distribution for each continent. The average distance of gaps between body mass aggregations within each continent (including standard deviations (SD)), the total number of body mass aggregations within each continent and the total number of species within each continent.

Continent	Average Distance of Gaps	Number of Aggregations	Number of Species
Mammals			
North			
America	0.04(0.04)	17	609
South			
America	0.02(0.02)	18	829
Birds			
North			
America	0.01(0.01)	23	1534
South			
America	0.01(0.02)	31	3136

Table 5.8. Mammal and bird body mass distribution summary statistics for the entire western hemisphere. The average distance of gaps between body mass aggregations, including standard deviations (SD), the total number of body mass aggregations and the total number of species.

	Average Distance of Gaps	Number of Aggregations	Number of Species
Mammals	0.01(0.02)	22	1261
Birds	0.01(0.01)	32	3560

Table 5.9. The observed variance is the sum of observations that were within each mammal and bird body mass aggregation across each of the ecoregions within a biome, including standard deviations (SD), at the ecoregion, biome and continental scales.

Biome	Habitat	Mammals	Birds
		Variance	Variance
1	Tropical and Subtropical Moist Broadleaf Forests	650.23(25.5)	722.54(26.88)
2	Tropical and Subtropical Dry Broadleaf Forests	71.53(8.46)	90.68(9.53)
3	Tropical and Subtropical Coniferous Forests	8.6(2.93)	13.49(3.67)
4	Temperate Broadleaf and Mixed Forests	42.67(6.53)	61.54(7.84)
5	Temperate Coniferous Forests	88.79(9.42)	148.81(12.2)
6	Boreal Forests/Taiga	34.13(5.84)	51.35(7.16)
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	8.73(2.96)	9.92(3.15)
8	Temperate Grasslands, Savannas and Shrublands	32.99(5.74)	56.5(7.52)
9	Flooded Grasslands and Savannas	4.38(2.09)	5.51(2.35)
10	Montane Grasslands and Shrublands	12.12(3.48)	10.12(3.18)
11	Tundra	16.1(4.01)	31.1(5.58)
12	Mediterranean Forests, Woodlands and Scrub	2.33(1.53)	2.56(1.6)
13	Deserts and Xeric Shrublands	43.52(6.6)	63.11(7.94)
14	Mangroves	4.79(2.19)	5.46(2.34)
Biome		18.12(4.26)	23.75(4.87)
Western Hemisphere		0.5(0.7)	0.58(0.76)

Table 5.10. The sum of observations that were within each mammal body mass aggregation across each of the ecoregions within a biome. The observed variance of all the sums of observations across ecoregions, in each biome. An exact rank of 950 or higher ($\alpha = 0.05$) was the level of significance. A binomial distribution was calculated in order to determine whether all the observed variances for ecoregions within each biome were higher than the simulated variances by chance alone.

Biome	Habitat	Simulated	Observed	Exact Rank
1	Tropical and Subtropical Moist Broadleaf Forests	405.7	650.23	1000
2	Tropical and Subtropical Dry Broadleaf Forests	56.73	71.53	998
3	Tropical and Subtropical Coniferous Forests	7.676	8.6	850
4	Temperate Broadleaf and Mixed Forests	30.89	42.67	1000
5	Temperate Coniferous Forests	70.2	88.79	1000
6	Boreal Forests/Taiga	19.92	34.13	1000
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	6.158	8.73	996
8	Temperate Grasslands, Savannas and Shrublands	31.04	32.99	768
9	Flooded Grasslands and Savannas	3.703	4.38	882
10	Montane Grasslands and Shrublands	7.618	12.12	1000
11	Tundra	9.246	16.1	1000
12	Mediterranean Forests, Woodlands and Scrub	1.859	2.33	920
13	Deserts and Xeric Shrublands	41.37	43.52	753
14	Mangroves	3.699	4.79	971
Binomial Distribution			6.10E-05	

Table 5.11. The sum of observations that were within each bird body mass aggregation across each of the ecoregions within a biome. The observed variance of all the sums of observations across ecoregions, in each biome. An exact rank of 950 or higher ($\alpha = 0.05$) was the level of significance. A binomial distribution was calculated in order to determine whether all the observed variances for ecoregions within each biome were higher than the simulated variances by chance alone.

Biome	Habitat	Simulated	Observed	Exact Rank
1	Tropical and Subtropical Moist Broadleaf Forests	663.7	722.54	1000
2	Tropical and Subtropical Dry Broadleaf Forests	94.86	90.68	169
3	Tropical and Subtropical Coniferous Forests	11.15	13.49	999
4	Temperate Broadleaf and Mixed Forests	47.59	61.54	1000
5	Temperate Coniferous Forests	116.6	148.81	1000
6	Boreal Forests/Taiga	38.15	51.35	1000
7	Tropical and Subtropical Grasslands, Savannas and Shrublands	8.844	9.92	949
8	Temperate Grasslands, Savannas and Shrublands	47.34	56.5	1000
9	Flooded Grasslands and Savannas	5.132	5.51	805
10	Montane Grasslands and Shrublands	11.05	10.12	114
11	Tundra	29.86	31.1	764
12	Mediterranean Forests, Woodlands and Scrub	2.376	2.56	760
13	Deserts and Xeric Shrublands	63.26	63.11	483
14	Mangroves	5.12	5.46	772
Binomial Distribution			0.02	

Table 5.12. Mean (x) and standard deviation (SD) of Phi coefficients and difference in ranks for each comparison between biomes.

Biome Comparisons	x	Difference in Ranks	p
Temperate Broadleaf and Mixed Forests	0.568(0.1560)		
with Boreal Forests/Taiga	0.48(0.1210)	233.577	<0.05
with Temperate Grasslands, Savannas and Shrublands	0.469(0.1210)	281.236	<0.05
with Deserts and Xeric Shrublands	0.417(0.0968)	478.105	<0.05
Boreal Forests/Taiga	0.637(0.1410)		
with Temperate Grasslands, Savannas and Shrublands	0.414(0.0794)	662.276	<0.05
with Deserts and Xeric Shrublands	0.398(0.0671)	727.445	<0.05
Temperate Grasslands, Savannas and Shrublands	0.506(0.1490)		
Deserts and Xeric Shrublands	0.454(0.1180)	149.355	>0.05
Deserts and Xeric Shrublands	0.472(0.1480)		



Figure 5.1. Biomes of the Western Hemisphere. Image adapted from Olson et al. (2001).

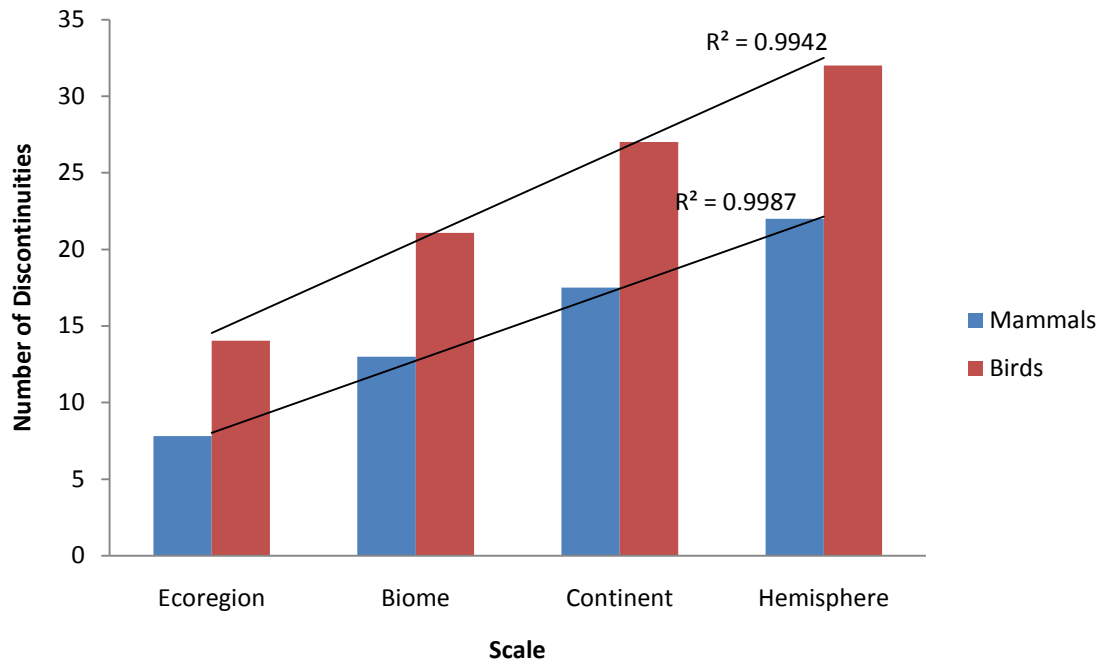


Figure 5.2. The number of discontinuities detected by Bayesian CART and SAS cluster analysis at four different scales. The average number of discontinuities was used at the ecoregion, biome and continental scale.

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CHAPTER 6: A SOCIO-ECOLOGICAL ANALYSIS OF GLOBAL INVASIONS AND EXTINCTIONS

Multiple forces, exacerbated by human activity, work together to cause the decline of many species (Wilson 2002). Human population size, CO₂ production, biomass consumption, energy use and geographical range size are orders of magnitude greater than any other species (Fowler & Hobbs 2003). The debate, as to what aspect(s) of humanity is responsible for environmental degradation, has been going on since the 1970s (Ehrlich & Holdren 1971, Commoner et al. 1971) and current governance regimes are unable to mitigate the adverse ecological impacts of socio-ecological systems (UNEP 2007).

One of the major unresolved problems, at the forefront of worldwide environmental concerns, is the increase in invasive and endangered species. Unfortunately, even after 40 years of study, ecologists are still not able to determine the processes which govern invasions and extinctions with any predictive power (Bright 1998). There are thousands of invasive plants and animal species that have established themselves throughout the world and the number is rising. Invasive species can alter the evolutionary pathway of native species via predation, hybridization, niche displacement, competitive exclusion and possibly extinction (Mooney & Cleland 2001). As native species decline, introduced non-indigenous species may become established and can affect ecosystem processes at varying scales and can potentially lead to the further extinction of native species (Williamson 1996, Vitousek et al. 1997a, Forsy & Allen 2002).

Biodiversity has been increasing for the past 600 million years (Signor 1990); yet recent studies indicate a global decline in biodiversity, no reduction in rate of decline, and an increase in pressures on biodiversity (Butchart et al. 2010). Changes in biodiversity due to human actions have been more profound in the last 50 years than in entire recorded human history, with an astonishing 52% of cycads, 32% of amphibians, 25% of conifers, 23% of mammals, and 12% of bird species threatened with extinction (Millennium Ecosystem Assessment 2005). According to The World Conservation Union's (IUCN) Invasive Species Specialist Group (ISSG), 1159 species have possibly gone extinct and 22% of vertebrates, 41% of invertebrates and 70% of plants are endangered (Vie et al. 2009). The integrity of the ecosystem declines with the loss of native species (Noss 1995, Sanders et al. 2003) and may affect the delivery of ecosystem services (Ehrlich & Ehrlich 1992). Extinction rates are 100 to 1,000 times their pre-human levels (Pimm et al. 1995, UNEP 2007). This potential loss of native species diversity may disrupt the numerous ecological processes that inherently shape landscape structure, such as predator-prey dynamics, dispersal, foraging behavior and functional group composition.

The term "landscape structure" has many different definitions in science, business, government, and in different cultures. There is a critical need for integrated concepts and research capable of uniting the natural and social sciences (Pickett et al. 1997). Studies have recognized the need to couple human systems with environment systems (Turner et al. 2003), the convergence of environmental and financial markets (Sandor et al. 2002), the importance of socio-cultural dynamics in natural resource management (Stratford & Davidson 2002) and the tremendous impact of humans on the

environment in comparison to other species (Fowler & Hobbs 2003). Numerous studies have focused on only one aspect of the socio-ecological relationship such as carbon emissions (Kratena 2004), water (Postel 2003) or human population growth (Struglia & Winter 2002).

At present, only three projects have attempted to focus on and integrate multiple socio-ecological factors at a national scale, with an emphasis on their roles in an ecologically sustainable society, into an index of values that can be ranked and compared. In 2003, the Global Footprint Network was established in an effort to establish and maintain a sustainable future. As part of that effort, the Ecological Footprint was created. This metric, comprised of 5 levels and 6 sub-categories, calculates how much natural resources we have, how much we use and who uses it in order to track human demands on the biosphere (Ewing et al. 2008). The U.S. National Aeronautics and Space Administration's (NASA) Socioeconomic Data and Applications Center (SEDAC) published three indexes, the 2005 Environmental Sustainability Index (ESI) (Esty et al. 2005), the 2006 Environmental Performance Index (EPI) (Esty et al. 2006) and the 2008 EPI (Esty et al. 2008). Each index was developed in order to explore the relationships, at a national scale, between multiple socio-ecological factors and their effect on a country's environmental performance and sustainability.

The 2005 ESI consisted of 76 socio-ecological variables that were grouped into 21 subcategories, under 5 main categories. The 2006 EPI consisted of 16 socio-ecological variables that were grouped into 6 main categories. The 2008 EPI consisted of 26 socio-ecological variables that were grouped into 10 subcategories, under 6 main categories. In 2004, the Environmental Vulnerability Index (EVI) was created by the South Pacific

Applied Geoscience Commission and the United Nations Environment Program in order to provide a rapid and standardized method of assessing a country's vulnerability to negative impacts on sustainable development (Kaly et al. 2004). This index incorporated 50 indicators organized under 7 main categories. Final countrywide index results, the indicator selection process and indicator definitions can be found in their respective final reports.

These indices utilized indicators that represented another layer of the ecosystem, previously unaccounted for in ecological research. These indices demonstrated that the economic, demographic, environmental and societal variables are not mutually exclusive, but highly integrated and have profound impacts on a country's sustainability. Indices create a single condensed quantity based on multiple, multi-dimensional variables (Ebert & Welsch 2004). In order to ascertain which socio-ecological variables are important, I must utilize multi-modal inference and model selection. There is a clear need for better models that can help elucidate the complex interactions between humans and their environment (Balmford et al. 2005).

The three aforementioned indices incorporated a diverse range and number of indicators, unique categorical organization of indicators, and were created in order to measure a country's environmental performance and sustainability. A more relevant measure of a country's sustainability or performance lies in its ecosystem resilience (Carpenter et al. 2001). Ecosystem resilience is defined as the magnitude of disturbance that can be absorbed by a system before it changes its structure and control (Holling & Gunderson 2002). I suggest that ecosystem resilience can be measured by the proportion of invasive and endangered bird and mammal species within a country. Those countries

with a higher proportion of invasive and endangered bird and mammal species would presumably have lower ecosystem resilience.

I plan to utilize a unique set of socio-ecological factors to explore their relationships with the proportion of endangered and invasive birds and mammals within each country and resilience. I divided fifteen socio-ecological factors into three broad groups which included; (i) Economic, (ii) Ecological, and (iii) Social/Governance. The economic group included; 1) GDP per capita, 2) Export/Import ratio, 3) Tourism, 4) Under Nourishment, and 5) Energy Efficiency. The ecological group included; 1) Agriculture intensity, 2) Rainfall, 3) Water stress, 4) Wilderness protection, and 5) Total biodiversity. The Social/Governance group included; 1) Life expectancy, 2) Adult literacy, 3) Pesticide regulations, 4) Political stability, and 5) Women in government. I also included three other factors and two reference indexes.

Socio-ecological Factors

Economic

Gross national product (GNP) has been shown to have an inverse relation to species richness; countries with high GNP are located in the upper latitudes with low biodiversity and countries with low GNP are located in the tropics with high biodiversity (Huston 1994). Gross domestic product per capita (GDP), a standard measure of affluence, has been shown to have a curvilinear relationship with environmental impact and this relationship has been termed the environmental Kuznets curve (EKC) (Cavlovic et al. 2000, Stern 2004, Dietz et al. 2007). Species richness of invasive plants (Liu et al. 2005) and all invasive taxonomic groups combined (Lin et al. 2007) were both positively

correlated with increased GDP per capita. International trade positively affects a country's income (Frankel and Romer 1999). Therefore, limiting trade would limit a country's income and subsequently diminish opportunities for biological invasions. In fact, a closed international trade policy helped Eastern European bloc countries limit invasive bird species introductions during the Cold War (Chiron et al. 2010).

Commercial energy consumption was reported to be positively correlated with exports (developing countries were observed having a stronger correlation than developed countries) and imports by industrialized countries were associated with less energy consumption (Suri & Chapman 1998). International trade typically results in the exporting country bearing more ecological costs than the country importing those goods (Machado et al. 2001). International trade has grown at twice the rate of economic growth since 1950 (Kates & Parris 2003) and this globalization has resulted in a sharp increase in non-indigenous introductions (GISP 2001, Perrings et al. 2002). A 3-24% increase in invasive insects and plants by 2020 were predicted by using establishment rates of non-indigenous species and international trade data for the past 100 years in the United States with projected trade forecasts for the United States (Levine & D'Antonio 2003). These non-indigenous introductions are hard to reverse, amount to severe economic losses, alter the structure of and function of ecosystems and decrease the diversity of native species (Vitousek et al. 1997a).

International tourist arrivals are estimated to reach approximately 1.6 billion people per year by 2020 (UNWTO 2009a) combined with a similar number of domestic tourists (Holden 2009). Nature has been hypothesized and empirically shown to be an influential factor in tourism demand. Based on tourism arrivals per capital (mass-

tourism) and tourism expenditures per GDP (individual tourism), Freytag & Vietze (2010) suggest that nature is an influential factor in individual tourism demand. Tourism may provide new employment, income, local participation in ownership or management and eliminate dependence on exploitative natural resource activities (reviewed in Mbaiwa & Stronza 2009). However, eco-tourism has a paradoxical nature; the more attractive a site is, the more popular it becomes, which in turn brings more tourists and this heavy visitation will degrade the site and other numerous environmental impacts (Hillery et al. 2001, reviewed in Holden 2009). In other words, there is a “resource paradox” in which tourism needs environmental resources, but it depends on the protection of those resources for sustained competitiveness (Williams & Ponsford 2009). Tourism in a country is positively correlated with its degree of biodiversity and a high degree of endangered biodiversity is negatively correlated with tourism (Freytag & Vietze 2009).

Tourism infrastructure (i.e. roads, trails, fences, parking lots, tourist shops, etc.) limits available land and resources of the local people, promotes human sprawl, enables encroachment on wildlife and degradation of protected areas (Vanderpost 2006, Geneletti & Dawa 2009), causes noise pollution and results in poor waste management (Mbaiwa 2003) and can even result in harmful provisioning of food to wildlife (Orams 2002). This loss of habitat and degradation of protected areas could negatively impact biodiversity and international tourism development, specifically in developing countries, has given little regard to the environmental impact caused by tourists (Honey 1999). Non-indigenous species can enter a country unintentionally as a byproduct of tourism, enabling species to overcome geographic barriers (Chown et al. 1998, Wilcove et al. 1998, GISP 2001, Perrings et al. 2002, Messing & Wright 2006). Climate change could

alter tourism and commerce, thereby potentially linking new geographic regions which may be subsequently invaded by non-indigenous species (Armstrong & Ball 2005, Hellmann et al. 2008).

The Food and Agriculture Organization of the United Nations estimates that more than 1.02 billion people in the world are undernourished and most live in developing countries (FAO 2009). Malnutrition can reduce the economic performance of people and promote unsustainable farming practices that can lead to more poverty, political instability, violence, and environmental degradation (Gonzalez 2004, Chapman et al. 2006, Gonzalez 2006). Smith et al. (2010) suggest that on a global scale, regions with high levels of undernourishment have weaker governance, which results in a failure of governments to regulate overfishing, bycatch and the environmental impacts of aquaculture.

Energy efficiency is a measure of technology. As technology improves, humans become more energy efficient. Environmental impacts can therefore be potentially reduced via “refinement of production” or super industrialization (Mol 1995). Improved technology cannot substitute for an ecosystem service, but it allows humans the ability to determine the trade-offs among ecosystem services (Rodriguez et al. 2006). An increase in agricultural energy efficiency would reduce the withdrawal of freshwater, which would in turn lessen the impact on the environment (Kates & Parris 2003) and lead to improvements in the supply of ecosystem services (Carpenter et al. 2006).

Ecological

Land use is determined by what ecosystem services can be provided and are needed by humans (Nelson et al. 2006). Humans have appropriated, managed and

modified approximately 50% of the terrestrial ecosystem (Vitousek et al. 1997b). Agricultural land is now one of the largest terrestrial biomes in the world, occupying about 40% of the earth's surface, and continual expansion has resulted in increased energy demand, water consumption, pesticide application and fertilizer use (Kates & Parris 2003, reviewed in Foley et al. 2005). Drainage, crop rotation, tillage, intercropping, grazing and the use of fertilizers and pesticides are the tools used in most agricultural activities and they have negative impacts on biodiversity (reviewed in McLaughlin & Mineau 1995). Using the past 35 years of global trends in agriculture, Tilman et al. (2001) have forecasted, by the year 2050, that the eutrophication and habitat destruction inherent in agriculture will lead to unprecedented species extinctions and loss of critical ecosystem services. Dobson et al. (1997) observed a positive correlation between agricultural activity and the density of endangered plants, mammals and birds at the state level in the United States of America. More invasive plant species were observed next to intensely farmed fields than next to moderate or low intensity farmed fields (Boutin & Jobin 1998). Human-modified ecosystems can provide the optimal environment for successful invasions (Vitousek et al. 1997a) and the disturbance or complete destruction of natural habitat via human agricultural activities is responsible for the global distribution of invasive weeds and pests (Huston 1994).

Small mammal (Williams 1995, Badgley & Fox 2000, Heaney 2001), bird (Rensburg et al. 2002, Hawkins et al. 2003), amphibian and reptile species richness increased with an increase in annual rainfall (Woinarski et al. 1999). However, in the case of desert rodents, it has been shown that the highest species richness occurred at intermediate levels of rainfall, with biodiversity actually decreasing in areas of extremely

high rainfall (Abramsky & Rosenzweig 1984). Mean annual precipitation was positively correlated with the number of threatened bird and mammal species in a global country by country analysis (McKee et al. 2003) and positively correlated with the overall density of endangered species within the United States of America at the state level (Dobson et al. 1997). There is no evidence of an increase in invasive birds or invasive mammals in relation to higher rainfall in the literature. However, a positive correlation between high rainfall and an increase in the biomass of non-indigenous plants has been reported in the deserts (Brooks & Pyke 2001) and grasslands (Hobbs & Mooney 1991) of North America. Annual variation in the abundance of invasive Argentine ants *Linepithema humile* was positively correlated to annual rainfall (Bolger 2007).

Approximately one quarter of the fresh water supply on the earth has been modified, managed or appropriated by humans (Postel et al. 1996) and the costs (e.g., considerable biodiversity losses) outweigh the benefits (Postel 2003, reviewed in Foley et al. 2005). Agricultural use accounts for approximately 90% of total human consumption of freshwater resources (CSD 1997). Running water ecosystems, which may be the most impacted ecosystem on the planet, may be rapidly degrading due to damming, diversion and extraction (Malmqvist & Rundle 2002). On a state by state analysis in the United States of America, human water use was negatively correlated to the density of endangered reptiles (Dobson et al. 1997). Physiological differences in invasive plant species, such as higher reproductive effort (White et al. 1997) and more efficient water use (Lambrinos 2002), can negatively impact native plant species when water availability is limited.

Approximately 12% of the Earth's land is protected and less than half of it is for the sake of biodiversity conservation (Hoekstra et al. 2005). Wilderness protection, if done correctly and based on science, is an essential factor in the preservation and conservation of the remaining biodiversity worldwide and the protection has been shown to be effective (DeFries et al. 2005, reviewed in Fischer et al. 2006). In order to protect and preserve global biodiversity, it is estimated that each country would have to set aside at least 50% of its land area (Soulé & Sanjayan 1998). However, more reserves in a country does not necessarily equate to the protection of more endangered species. In a study of 30 important reserves in Mexico, 10 reserves represented all the endangered mammals in their analysis (Ceballos 2007). Well managed, protected wilderness areas, via strict control and restoration measures, can help reduce, slow or even halt potential spread of invasive species (Randall 2000). Oceanic island reserves have a higher number of invasive species than mainland reserves, reserve size is inversely related to the proportion of invasive plants in mediterranean-type ecosystems, and reserves located in extreme climates have fewer invasive species than reserves located in moderate climates (reviewed in Cole & Landres 1996).

The use of total biodiversity, or species richness, as an indicator has been used in other studies, but with mixed results. In areas of high species richness, there were relatively low proportions of threatened species. However, in hotspots that contained a high proportion of threatened species, there was also higher overall species richness (Orme et al. 2005). McKee et al. (2003) observed a positive correlation between the number of threatened bird and mammal species with species richness. There are also conflicting results in invasive species analyses. As species richness increased, invasive

species success declined in some studies (Chown et al. 1998, Levine 2000). However, invasive species richness was positively correlated with native species richness in another study (Lonsdale 1999). There is no single theory or process that can explain biological diversity (Huston 1994).

Social/Governance

Life expectancy has been extended by more than 20 years since 1950 and that improvement reflects a reduction in infant and child mortality due to improvements in water quality, sanitation, nutrition and immunizations (Kates & Parris 2003). Life expectancies are higher in high-income countries than in developed countries, but that gap has closed in recent years (Nelson et al. 2006). Life expectancy is a complex metric that has many direct and indirect components and has been used in other studies to determine the extent to which human well-being could increase without an accompanying increase in environmental deterioration (Dietz et al. 2007).

Miller (2002) reports that fewer than 20% of Americans are literate enough to comprehend a science book, read a science article in a newspaper or understand a science-based television program. Literacy in many areas of science is lacking and may be insufficient to maintain an informed citizenry (Jordan et al. 2009). A high degree of adult literacy brings with it greater access to information, which in turn would suggest that better decisions would be made concerning the environment. In a study that compared approximately 140 countries, a higher adult literacy rate correlated with less pollution in the cases of sulfur dioxide, heavy particles, dissolved oxygen, fecal coliform pollution and sanitation (Torrás & Boyce 1998).

The notion that pesticides are detrimental to the environment was first brought to the forefront of public concern by Rachel Carson, in her book entitled 'Silent Spring' (Carson 1962). Pesticide use has led to declines in amphibians (Sparling et al. 2001), birds (Anthony et al. 1993) and the decimation of pollination systems (Kearns et al. 1998) and numerous other deleterious effects (reviewed in Pimentel et al. 1992). The enactment and implementation of pesticide regulations can control direct, human-caused mortality of endangered species (Miller et al. 2002). Pesticides have been used to successfully control invasive species, but these attempts are very expensive and the targeted species may develop a resistance (Pimentel et al. 1992, Schmitz & Simberloff 1997, McKee et al. 2009).

The pygmy hippo *Hexaprotodon liberiensis* is now classified as endangered due to political instability and unrest in West African nations (IUCN 2006). Environmental degradation can lead to social collapse, famine, disputes within and between nations, and war and vice versa (McNeely 2000, Nelson et al. 2006). Political stability is essential to the success of ecological restoration projects, which are typically undergone to increase species richness in degraded ecosystems. It has been asserted that the persistence of these conservation projects is directly tied to the degree and frequency of political unrest (Soulé 1991). Di Castri (2000) presents new approaches and tools that could be used to promote environmental health, such as early monitoring of invasive species, but the success of these recommendations relies on many factors, including political stability. Risk assessment and risk management are political processes (Peterson et al. 2000).

Studies have shown that women have a very different perspective than men on the subject of the environment. Since the turn of the twentieth century, women have been

political champions of the environmental protection and conservation movement (reviewed in Kleehammer 2011). Women are more concerned about the pain and suffering of animals (e.g., more opposed to hunting, predator control, and trapping), more involved in protest efforts and constitute the majority membership of humane societies and animal-welfare organizations (Kellert & Berry 1987). In a political survey conducted in the United States of America, women respondents answered in overwhelming majority that a female candidate would be more competent at protecting the environment than a male candidate (Sapiro 1981). In the United Kingdom, the Labour and Green Parties have the highest proportion of women candidates (Norris & Lovenduski 1995). In 1992, at the Democratic National Convention, women candidates had a common theme of protecting the environment (Kahn & Gordon 1997). A country with a high proportion of seats held by women in national parliament or government could result in more environmental protections, which could result in fewer endangered or invasive species.

Other Factors

Total population has been suggested as a key driver of environmental impact (Ehrlich and Holdren 1971) and numerous studies have reported negative effects of population on environmental impact (Kates & Parris 2003, McKee et al. 2003, York et al. 2003, Dietz et al. 2007). In one study, a positive relationship was found between the number of United States federally listed mammals and total human population (Kirkland & Ostfeld 1999). Human total population growth is an ultimate driver in the increase in biological invasions by non-indigenous species (Vitousek et al. 1997b, Mooney & Cleland 2001).

Latitude and total land area were used in the analyses to account for climate and biogeography (York et al. 2003, Dietz et al. 2007). Diversity is lowest near the poles and increases towards the tropics for most plant and animal species (Huston 1994). Latitude has been shown to influence local environmental conditions which affect plant growth (Li et al. 1998). Environmental impacts have been reported to be greater the further a country is from the tropics (York et al. 2003). Non-indigenous species richness has been shown to peak in the subtropics and decline with latitude (Sax 2001) and this was also observed at a country level scale (Liu et al. 2005). Ecological niche models based on four future climate scenarios predicted that the invasive Argentine ant would retract its range in the tropics and expand into higher latitudes (Roura-Pascual et al. 2004). The species-area relationship states that species richness tends to increase with increasing sampling area (Gleason 1922, MacArthur & Wilson 1967, Rosenzweig 1995). Species richness of land mammals in 155 countries was positively correlated with total land area (Ceballos & Brown 1995). Studies have reported that as total land area increased, a country's environmental impact increased, due to presumed effects of energy demand, efficiency and usage (York et al. 2003, Dietz et al. 2007).

METHODS

Data Collection

There were sufficient data to analyze 100 countries, which contain approximately 87% of the world's population, 43% of the world GDP per capita and 74% of the earth's total land area (Appendix S-U). The region of Africa was represented by 26 countries. The region of Asia was represented by 29 countries. The region of Europe was

represented by 22 countries. The regions of North and Central America were represented by 12 countries. South America and Oceania were represented by 11 countries. Energy efficiency or total primary energy consumption was calculated by tabulating the consumption of petroleum, dry natural gas, coal, and net hydroelectric, nuclear, and geothermal, solar, wind, wood and waste electric power, and net electricity imports (electricity imports minus electricity exports) (Esty et al. 2006). Total biodiversity, included known mammals, birds, reptiles, plants, amphibians and fishes in each country as of 2004 (WRI 2005). Each country's latitude was obtained from the Central Intelligence Agency's World Factbook (CIA 2008a).

Total population and GDP per capita for each country were reported in the 2008 EPI and values represented the year 2005 (Esty et al. 2008). Total land area was reported in the 2008 EPI (Esty et al. 2008). Water stress was defined as the percentage of national territory with water withdrawals exceeding 40% of available water (Esty et al. 2008). Agricultural intensity was measured as the percentage of cropland area that is in agriculture-dominated landscapes. High agricultural intensity was defined as having more than 60% of a country's lands cultivated, low intensity having at least 40% of the land uncultivated (Esty et al. 2008). Pesticide regulation was the legislative status of countries under the Rotterdam and Stockholm conventions and to what degree they have followed through on the convention bylaws (Esty et al. 2008).

The numbers of international tourist arrivals per country were obtained by accessing the United Nations World Tourism Organization database (UNWTO 2009b). Undernourishment, or the percentage of the population between 2001 and 2003 that were malnourished, were obtained from the FAO (FAOc 2006). Annual rainfall data were

reported by the FAO and represent annual rainfall for the year 2002 (FAOb 2006). Average rainfall between 1971 and 2000 was not different than rainfall in 2002 (Mann-Whitney Rank Sum Test, $p=0.839$). Adult literacy or the percentage aged 15 and above, between 1995 – 2005, that are literate were obtained from the United Nations Development Programme's Human Development Report (UNDP 2007). Political stability within a country measured the likelihood that the government would be destabilized or overthrown by unconstitutional or violent means (Kaufmann et al. 2008).

Women in government represent the proportion of seats held by women in national parliament as of 2007 (MDGD 2009). Exports were divided by imports to create an export/import ratio. Export and import data were obtained from the FAO and represented the year 2004 (FAOa 2006). Wilderness protection was defined as the amount of land classified by the United Nations Statistics Division as protected ("an area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means"), divided by the total land area of a country (UNSD 2008). Life expectancy data were obtained from the CIA World Factbook and were calculated as the overall life expectancy at birth regardless of gender (CIA 2008b).

For the purpose of this study, endangered birds and mammals included those species that were classified by the IUCN Red List of Threatened Species as vulnerable, endangered, critically endangered, extinct in the wild and extinct (IUCN 2008). Invasive birds were determined using Birdlife International's world bird database, avibase (BLI 2008). Invasive mammals were determined using J.L. Long's definitive book on

introduced mammals of the world (Long 2004). The total number of birds and mammals in each country were determined using the IUCN database (IUCN 2008).

Data Analysis

Models are devices that make abstractions clear and understandable, by providing the ability to compare and contrast those abstractions with other models (Carpenter et al. 2005). The Information Theoretic approach (Burnham & Anderson 2002; Johnson & Ohmland 2004) was used to model these data based on Akaike Information Criteria (AIC) (Akaike 1973). For the complete set of models, AIC, the difference in AIC for that model relative to the best-fitting model with the minimum AIC (termed ΔAIC) and the Akaike weight (termed w_i) were all calculated. The best-fitting model was defined as that with the lowest AIC. Models that differed by less than 2 AIC units have substantial support in terms of explaining the data (Burnham & Anderson 2002). Evidence ratios were also calculated for each model (Burnham & Anderson 2002). Colinearity between explanatory variables was investigated using correlation matrices. Although associations were apparent, they were not sufficient to preclude their inclusion into the modeling process. All data were log transformed, when appropriate, so that every variable would be on the same scale. All analyses were performed in SAS version 9.1 (SAS 1999).

An a-priori set of models was selected and included a combination of all socio-ecological landscape factors, each socio-ecological landscape factor by itself and various combinations of socio-ecological landscape factors grouped into three broad classes. These three broad classes included; (i) Economic, (ii) Ecological, and (iii) Social/Governance. The economic class included; 1) GDP per capita, 2) Export/Import ratio, 3) Tourism, 4) Under Nourishment, and 5) Energy Efficiency. The ecological class

included; 1) Agriculture intensity, 2) Rainfall, 3) Water stress, 4) Wilderness protection, and 5) Total biodiversity. The Social/Governance class included; 1) Life expectancy, 2) Adult literacy, 3) Pesticide regulations, 4) Political stability, and 5) Women in government.

Analyses were separated by taxonomic group. The dependent variable in each of the four groups was; 1) the number of endangered mammals divided by the total number of mammals within a country, 2) the number of endangered birds divided by the total number of birds within a country, 3) the number of invasive mammals divided by the total number of mammals within a country, and 4) the number of invasive birds divided by the total number of birds within a country. In the resilience analysis, the dependent variable was calculated as the number of both endangered and invasive birds and mammals divided by the total number of birds and mammals in each country.

The ESI and EVI were used as reference indexes in these analyses. The EPI was not used as one of the reference indexes because it was correlated with both the ESI and its predecessor the EVI (Pearson CC= 0. 434, Pearson CC= 0. 544). The ecological footprint was not used as reference index because it does not account for local impacts (Dietz et al. 2007). Each country's percent of endangered and invasive birds and mammals, their combined percentage (i.e. resilience), ESI, EVI and total population can be found in Appendix A. Each country's GDP per capita, total land area, latitude, life expectancy, water stress, agricultural intensity, pesticide regulations, adult literacy and tourism values can be found in Appendix B. Each country's under nourishment, annual rainfall, energy efficiency, wilderness protection, total biodiversity, exports/imports, political stability and women in government values can be found in Appendix C.

RESULTS

The percentage of endangered birds in a country was best predicted by a model that included total biodiversity ($w_i = 0.442$) and total land area ($w_i = 0.275$) (Table 6.1). As total biodiversity and total land area increased, the percentage of endangered birds in a country increased (Table 6.2). The island nation of New Zealand was the exception to this trend (Figures 6.1 & 6.2). The percentage of endangered mammals in a country was best predicted by the ecological class model ($w_i = 0.938$) which included; 1) Agriculture intensity, 2) Rainfall, 3) Water stress, 4) Wilderness protection, and 5) Total biodiversity (Table 6.3). All variables had a positive correlation, except wilderness protection which was inversely correlated with the percentage of endangered mammals (i.e., as the amount of wilderness area protected decreased, the proportion of endangered mammals in a country increased). Only rainfall and water stress were significant in the model (Table 6.4).

The percentage of invasive birds in a country was best predicted by a model that included GDP per capita ($w_i = 0.938$) (Table 6.5). As GDP per capita increases, the percentage of invasive birds increases (Table 6.6). The island nation of New Zealand was the exception to this trend (Figure 6.3). The percentage of invasive mammals in a country was best predicted by a model that included GDP per capita ($w_i = 0.837$) (Table 6.7). As GDP per capita increases, the percentage of invasive mammals increases (Table 6.8). The island nations of New Zealand and the United Kingdom were the exceptions to this trend (Figure 6.4).

The resilience of a country was best predicted by a model that included life expectancy ($w_i = 0.526$) (Table 6.9). As life expectancy increases, the resilience of a country decreases (i.e., as human life expectancy increases, the proportion of endangered and invasive birds and mammals in a country increases, which indicates less resilience) (Table 6.10). The island nation of New Zealand was the exception to this trend. Twenty-three of the 26 countries in the Africa region were included in the top 25 most resilient countries (Figure 6.5).

DISCUSSION

Although the correlational nature of this study limits our ability to determine causal factors, the patterns observed in this study have provided insight into the dynamics of a complex, global, socio-ecological system. The percentage of endangered birds in a country was positively correlated with total biodiversity and total land area. These results were similar to other analyses on the relationship between the percentage of endangered birds and total biodiversity (McKee et al. 2003, Orme et al. 2005). Although there was no evidence in the literature describing a relationship between the percentage of endangered birds and total land area, I could presume that an increase in sampling area would result in an increase in species richness (reviewed in Huston 1994) and therefore a possibility of there being more endangered species present.

The percentage of endangered mammals in a country was correlated with a combination of factors (i.e., the ecological class of variables). The results were similar to other analyses on the relationship between endangered mammals and agriculture intensity (Dobson et al. 1997), total annual rainfall and total biodiversity (McKee et al. 2003).

Water stress has been reported as a threat to endangered species populations throughout the United States (Flather et al. 1998). There is evidence that the preservation of more habitat will allow for the survival of more species (Bruner et al. 2001), therefore I might assume fewer endangered species to be present. The inverse correlation observed in this study between wilderness protection and the percentage of endangered mammals suggests that this may be true.

The relationship between GDP per capita and invasive species has been observed in other studies. GDP per capita and invasives have been found to correlate with the richness of alien spiders (Kobelt & Nentwig 2008), plants (Liu et al. 2005), fishes (Leprieur et al. 2008), birds and mammals in Europe (Hulme 2007) and all taxonomic groups combined (Lin et al. 2007). The United Kingdom had a far greater percentage of invasive mammals than predicted by the model. This may be due to the unique history of this island nation. Around 1775 AD, London was at the crossroads of the ‘globalization’ of European trade routes, establishing trade between the Dutch, Spanish, Portuguese and French (Di Castri 1989). This vast trade economy, at an early period in this island nation’s history, presumably has also meant a long history of biological invasions.

The most resilient countries were those located in Africa and this may be due to the lack of invasive species in most of these countries. There were only 29 invasive bird species and 39 invasive mammal species reported in the 26 African countries included in this study. The lack of invasive species in these African countries may best be explained by international trade. Increased international trade has been positively correlated with an increase in invasive species (GISP 2001, Perrings et al. 2002). I obtained international trade data of the past 60+ years from the World Trade Organization (WTO 2008) and

found that African countries comprised approximately 50% or more of the countries in a list of the 25 countries with the least average amount (in U.S. dollars) of exports, imports and both figures combined. These countries have had very little international trade relative to a majority of the countries in this analysis due to a closed trade policy (Sachs & Warner 1997).

In every analysis, New Zealand was an extreme outlier. New Zealand had the highest proportion of endangered birds, invasive birds, and invasive mammals, and had the highest proportion of endangered and invasive species combined. New Zealand's complete lack of native terrestrial mammals (Diamond 1990) was a key factor in its outlier position relative to the rest of the countries analyzed. New Zealand has had a massive invasion by non-indigenous species since its human colonization, in the past 700-800 years, and this has resulted in catastrophic biodiversity loss (Clout 2001). New Zealand's invasive species crisis may be due in large part to its isolation, high endemism and recent human colonization (Norton 2009). Island ecosystems are often the most invaded and consequently threatened worldwide (Towns et al. 2006).

No other analysis to date has calculated a measure of resilience for a country, thus there is no evidence in the literature regarding its relationship with human life expectancy. Dietz et al. (2007) found no relationship between life expectancy and environmental impact. The overall trend in high income countries, with improvements to the HDI (which includes human life expectancy as one of its variables), is one towards a disproportionately larger negative impact on their ecological footprint. However, some lower income countries have high level of development without high impact on ecosystem services (Moran et al. 2008).

Human population control is a potential option that may increase the resilience of a country. Total population was not a factor in the selected models, but population control may be one of many crucial steps that must be taken in order to conserve global biodiversity (McKee et al. 2003). Furthermore, more humans may equate to higher human population densities which has been shown to have a negative correlation with the size of protected areas (Luck 2007). Even though the population growth rate is decreasing in developed countries, developed countries still have a high level of consumption which may be equivalent to rapid population growth in developing countries that have low consumption (York et al. 2003). The total fertility rate of the world has been declining since 1965, but this has been counteracted by a decreasing death rate (UNPD 2008). The idea of having only one offspring per human, or replacement level fertility, would be one method of human population control and stabilization. Educated women tend to have fewer children than uneducated women, therefore an education program for young women would be effective in developing regions of the world (Osili & Long 2008).

Humans are an integral part of the ecosystems in which they inhabit, accounting for the consumption of nearly 40% of potential terrestrial net primary productivity (Vitousek et al. 1986). As global climate change occurs, warming in some areas and cooling in others, species will be forced to shift their ranges (Thomas & Lennon 1999). These range shifts will be difficult for some species, impossible for others and beneficial to those that can quickly adapt (Parmesan et al. 1999). Conservation will only be successful if local communities are given the incentives, tools and capacity to manage ecosystems sustainably (Leader-Williams 2002) and understand that they are living on environmental capital rather than on interest (Jones 2003). The negative impacts of

humans will continue to increase as the global population grows, therefore humans must realize that the economic value of conserving what is left in nature is a magnitude of order greater than that of developing it (Costanza et al. 1997). Until that paradigm shift is achieved, we will continue to “have the appearance of a great deal of intellectual activity, but an outcome of social inertia” (Albrecht 2001).

Table 6.1. Model selection results for endangered birds. Bold values indicate variables in the best model. Evidence Ratio (ER).

Model	AIC _c	ΔAIC _c	w _i	ER
Total Biodiversity	214.13	0.00	0.44	1.00
Total Land Area	215.08	0.94	0.23	1.60
Life Expectancy	217.43	3.30	0.09	5.20
Total Population	218.58	4.44	0.05	9.21
Agricultural Intensity + Annual Rainfall + Water Stress + Wilderness Protection + Total Biodiversity	219.54	5.40	0.03	14.91
Adult Literacy	220.46	6.33	0.02	23.68
Under Nourishment	220.70	6.56	0.02	26.63
Pesticide Regulations	221.10	6.97	0.01	32.62
Water Stress	221.43	7.30	0.01	38.46
Null	221.84	7.71	0.01	47.22
Export Import Ratio	222.52	8.39	0.01	66.29
Energy Efficiency	222.75	8.62	0.01	74.37
Annual Rainfall	223.05	8.91	0.01	86.19
Tourism	223.08	8.94	0.01	87.53
Environmental Vulnerability Index	223.13	9.00	0.01	89.79
GDP per capita	223.70	9.57	0.00	119.46
Wilderness Protection	223.79	9.66	0.00	125.02
Agricultural Intensity	223.85	9.71	0.00	128.57
Environmental Sustainability Index	223.88	9.75	0.00	130.84
Women in Government	223.96	9.83	0.00	135.98
Political Stability	223.96	9.83	0.00	136.25
Latitude	223.97	9.84	0.00	136.66
Life Expectancy + Adult Literacy + Pesticide Regulations + Political Stability + Women in Government	224.42	10.29	0.00	171.26
GDP per capita + Export Import Ratio + Tourism + Under Nourishment + Energy Efficiency	227.43	13.30	0.00	771.01
All variables	236.31	22.18	0.00	1000.4

Table 6.2. Parameter estimates for variables selected in the best models in the endangered birds analysis.

Variable	Estimate	Standard Error	t Value	Pr > t
Intercept	2.41576	0.38245	6.32	<.0001
Total Biodiversity	0.09117	0.02864	3.18	0.002
Intercept	2.77852	0.3239	8.58	<.0001
Total Land Area	0.04528	0.015	3.02	0.0032

Table 6.3. Model selection results for endangered mammals.

Model	AIC _c	ΔAIC _c	w _i	Evidence Ratio
Ag. Intensity + Ann. Rainfall + Water Stress + Wilderness Protection + Total Biodiversity	353.704	0.000	0.938	1.00
Life Expectancy + Adult Literacy + Pesticide Regulations + Political Stability + Women in Government	361.023	7.319	0.024	38.84
Total Biodiversity	363.402	8.731	0.012	78.67
Water Stress	364.166	9.495	0.008	115.27
Political Stability	365.664	10.993	0.004	243.79
Total Population	365.962	11.291	0.003	282.96
Environmental Sustainability Index	366.407	11.736	0.003	353.47
Latitude	367.070	12.399	0.002	492.41
Life Expectancy	368.364	13.694	0.001	940.40
Environmental Vulnerability Index	368.877	14.206	0.001	1215.37
Null	369.554	14.756	0.001	1600.66
GDP per capita	369.473	14.802	0.001	1637.30
Annual Rainfall	369.714	15.043	0.001	1846.97
Total Land Area	370.190	15.519	0.000	2343.27
Agricultural Intensity	370.221	15.550	0.000	2379.88
Under Nourishment	370.291	15.620	0.000	2464.65
All variables	363.166	15.708	0.000	2576.05
Pesticide Regulations	370.653	15.982	0.000	2953.67
Women in Government	370.816	16.145	0.000	3204.48
Wilderness Protection	371.210	16.539	0.000	3902.23
Tourism	371.389	16.718	0.000	4267.59
Export Import Ratio	371.472	16.801	0.000	4448.42
Energy Efficiency	371.505	16.834	0.000	4522.43
Adult Literacy	371.546	16.875	0.000	4616.09
GDP per capita + Export Import Ratio + Tourism + Under Nourishment + Energy Efficiency	370.805	17.101	0.000	5169.33

Table 6.4. Parameter estimates for variables selected in the best model(s) in the endangered mammals analysis.

Variable	Estimate	Standard Error	t Value	Pr > t
Intercept	4.21526	1.82371	2.31	0.023
Agricultural Intensity	0.05029	0.04093	1.23	0.2222
Annual Rainfall	0.33938	0.12343	2.75	0.0072
Water Stress	0.14313	0.04013	3.57	0.0006
Wilderness Protection	-0.07443	0.05017	-1.48	0.1413
Total Biodiversity	0.10826	0.06668	1.62	0.1078

Table 6.5. Summary of model selection results for invasive birds. Evidence Ratio (ER).

Model	AIC _c	ΔAIC _c	w _i	ER
GDP per capita	57.040	0.000	0.879	1.00
GDP per capita + Export Import Ratio + Tourism + Under Nourishment + Energy Efficiency	61.762	4.722	0.083	10.60
Life Expectancy	64.650	7.610	0.020	44.93
Life Expectancy + Adult Literacy + Pesticide Regulations + Political Stability + Women in Government	66.333	9.293	0.008	104.20
Political Stability	66.350	9.310	0.008	105.11
Under Nourishment	71.228	14.188	0.001	1204.90
Adult Literacy	72.777	15.737	0.000	2613.90
Pesticide Regulations	74.712	17.672	0.000	6877.77
Latitude	74.965	17.925	0.000	7803.68
Environmental Vulnerability Index	76.748	19.708	0.000	19036.24
Women in Government	78.143	21.103	0.000	38230.92
Wilderness Protection	78.557	21.516	0.000	47013.97
Tourism	78.733	21.693	0.000	51351.53
Null	79.771	22.731	0.000	86280.22
Environmental Sustainability Index	79.786	22.746	0.000	86946.65
Annual Rainfall	80.828	23.788	0.000	146400.31
Energy Efficiency	80.830	23.790	0.000	146554.11
Total Biodiversity	80.998	23.957	0.000	159324.77
Total Land Area	81.010	23.970	0.000	160323.67
Water Stress	81.037	23.997	0.000	162518.97
Agricultural Intensity	81.340	24.300	0.000	189084.64
Total Population	81.704	24.664	0.000	226828.61
Export Import Ratio	81.886	24.846	0.000	248425.93
All variables	82.153	25.113	0.000	283908.51
Agricultural Intensity + Annual Rainfall + Water Stress + Wilderness Protection + Total Biodiversity	83.303	26.263	0.000	504536.59

Table 6.6. Parameter estimates for variables selected in the best model in the invasive birds analysis.

Variable	Estimate	Standard Error	t Value	Pr > t
Intercept	0.15755	0.1856	0.85	0.398
GDP per capita	0.06531	0.01242	5.26	<.0001

Table 6.7. Summary of model selection results for invasive mammals. Evidence Ratio (ER).

Model	AIC _c	ΔAIC _c	w _i	ER
GDP per capita	418.705	0.000	0.837	1.00
Political Stability	422.967	4.262	0.099	8.42
Latitude	425.283	6.578	0.031	26.82
Life Expectancy + Adult Literacy + Pesticide Regulations + Political Stability + Women in Government	426.261	7.556	0.019	43.74
GDP per capita + Export Import Ratio + Tourism + Under Nourishment + Energy Efficiency	427.430	8.725	0.011	78.47
Life Expectancy	431.141	12.436	0.002	501.70
Adult Literacy	433.987	15.282	0.000	2081.83
Under Nourishment	434.133	15.428	0.000	2239.48
Pesticide Regulations	434.323	15.618	0.000	2462.67
Women in Government	435.917	17.212	0.000	5464.35
Environmental Sustainability Index	436.273	17.568	0.000	6528.94
Environmental Vulnerability Index	440.229	21.524	0.000	47192.96
Tourism	440.453	21.748	0.000	52785.93
Null	440.779	22.074	0.000	62122.01
Total Biodiversity	440.981	22.276	0.000	68734.05
Wilderness Protection	441.104	22.399	0.000	73093.89
Export Import Ratio	441.250	22.545	0.000	78629.33
Agricultural Intensity	442.300	23.595	0.000	132919.64
Annual Rainfall	442.414	23.709	0.000	140716.15
Energy Efficiency	442.683	23.978	0.000	160974.30
Water Stress	442.685	23.980	0.000	161135.35
Total Population	442.723	24.018	0.000	164226.20
Total Land Area	442.890	24.185	0.000	178527.87
Ag. Intensity + Ann. Rainfall + Water Stress + Wilderness Protection + Total Biodiversity	445.434	26.729	0.000	637108.76
All variables	450.474	31.769	0.000	7918460.13

Table 6.8. Parameter estimates for variables selected in the best model in the invasive mammals analysis.

Variable	Estimate	Standard Error	t Value	Pr > t
Intercept	0.06318	1.13218	0.06	0.9556
GDP per capita	0.39242	0.07576	5.18	<.0001

Table 6.9. Summary of model selection results for resilience (endangered + invasive birds and mammals). Evidence Ratio (ER).

Model	AIC _c	ΔAIC _c	w _i	ER
Life Expectancy	312.859	0	0.526	1
Total Biodiversity	316.466	3.607	0.087	6.07
Pesticide Regulations	316.796	3.937	0.073	7.16
Total Land Area	316.819	3.96	0.073	7.24
Adult Literacy	317.581	4.722	0.05	10.6
Under Nourishment	317.822	4.963	0.044	11.96
GDP per capita	319.103	6.244	0.023	22.69
Total Population	319.224	6.365	0.022	24.11
Life Expectancy + Adult Literacy + Pesticide Regulations + Political Stability + Women in Government	319.745	6.886	0.017	31.28
Tourism	320.37	7.511	0.012	42.76
Environmental Vulnerability Index	320.687	7.828	0.01	50.1
Political Stability	320.789	7.93	0.01	52.72
Null	321.067	8.208	0.009	60.57
Annual Rainfall	321.195	8.336	0.008	64.59
Women in Government	321.673	8.814	0.006	82.02
Wilderness Protection	321.909	9.05	0.006	92.3
Export Import Ratio	321.947	9.088	0.006	94.07
Water Stress	322.381	9.522	0.004	116.86
Environmental Sustainability Index	322.421	9.562	0.004	119.22
Latitude	322.601	9.742	0.004	130.45
Agricultural Intensity + Annual Rainfall + Water Stress + Wilderness Protection + Total Biodiversity	322.956	10.097	0.003	155.82
Agricultural Intensity	323.19	10.331	0.003	175.13
Energy Efficiency	323.193	10.334	0.003	175.39
GDP per capita + Export Import Ratio + Tourism + Under Nourishment + Energy Efficiency	324.895	12.036	0.001	410.84
All variables	338.079	25.22	0	1000.13

Table 6.10. Parameter estimates for variables selected in the best model in the resilience analysis.

Variable	Estimate	Standard Error	t Value	Pr > t
Intercept	-4.1801	3.09793	-1.35	0.1803
Life Expectancy	0.14422	0.04415	3.27	0.0015

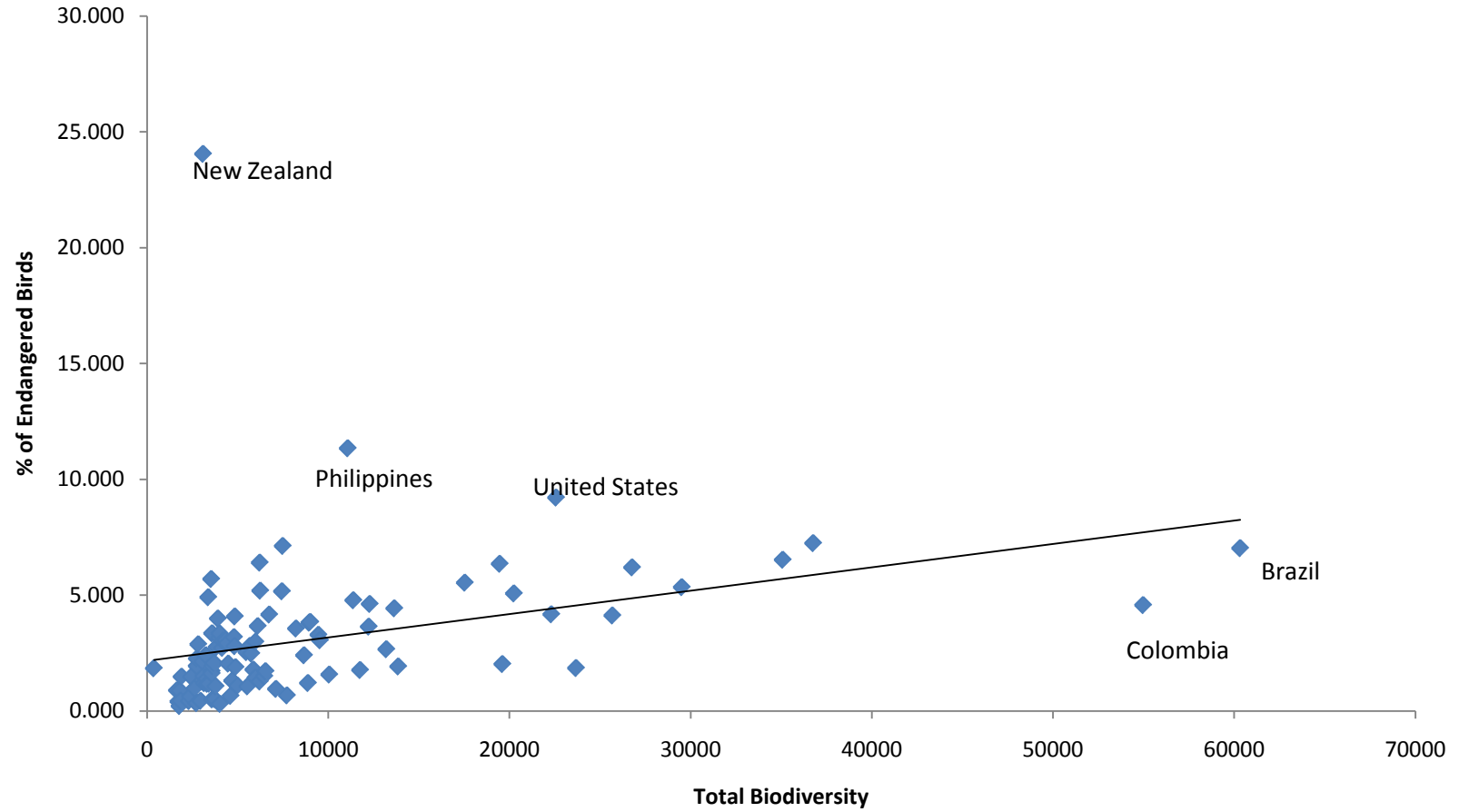


Figure 6.1. Endangered birds in relation to total biodiversity.

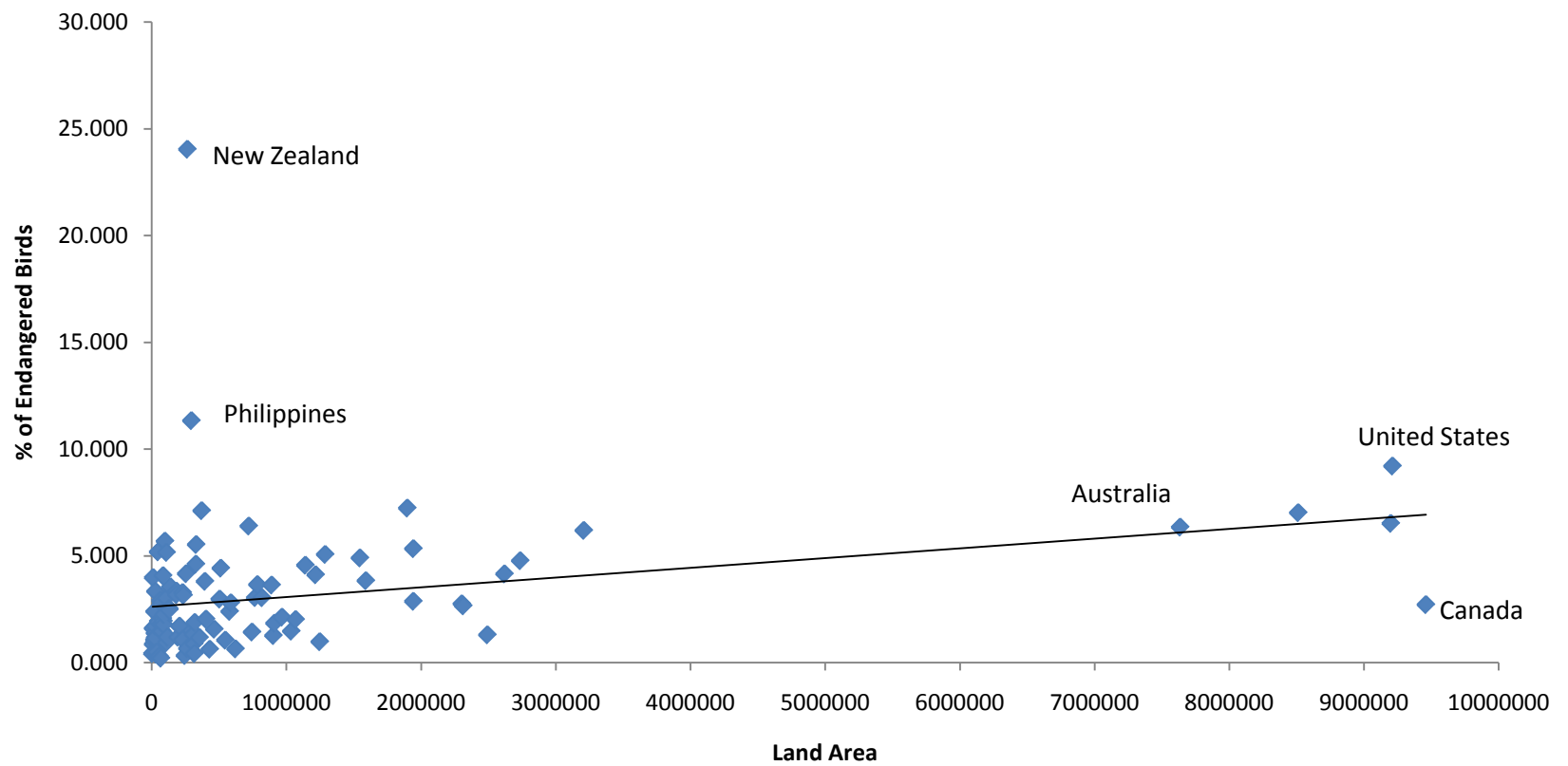


Figure 6.2. Endangered birds in relation to total land area.

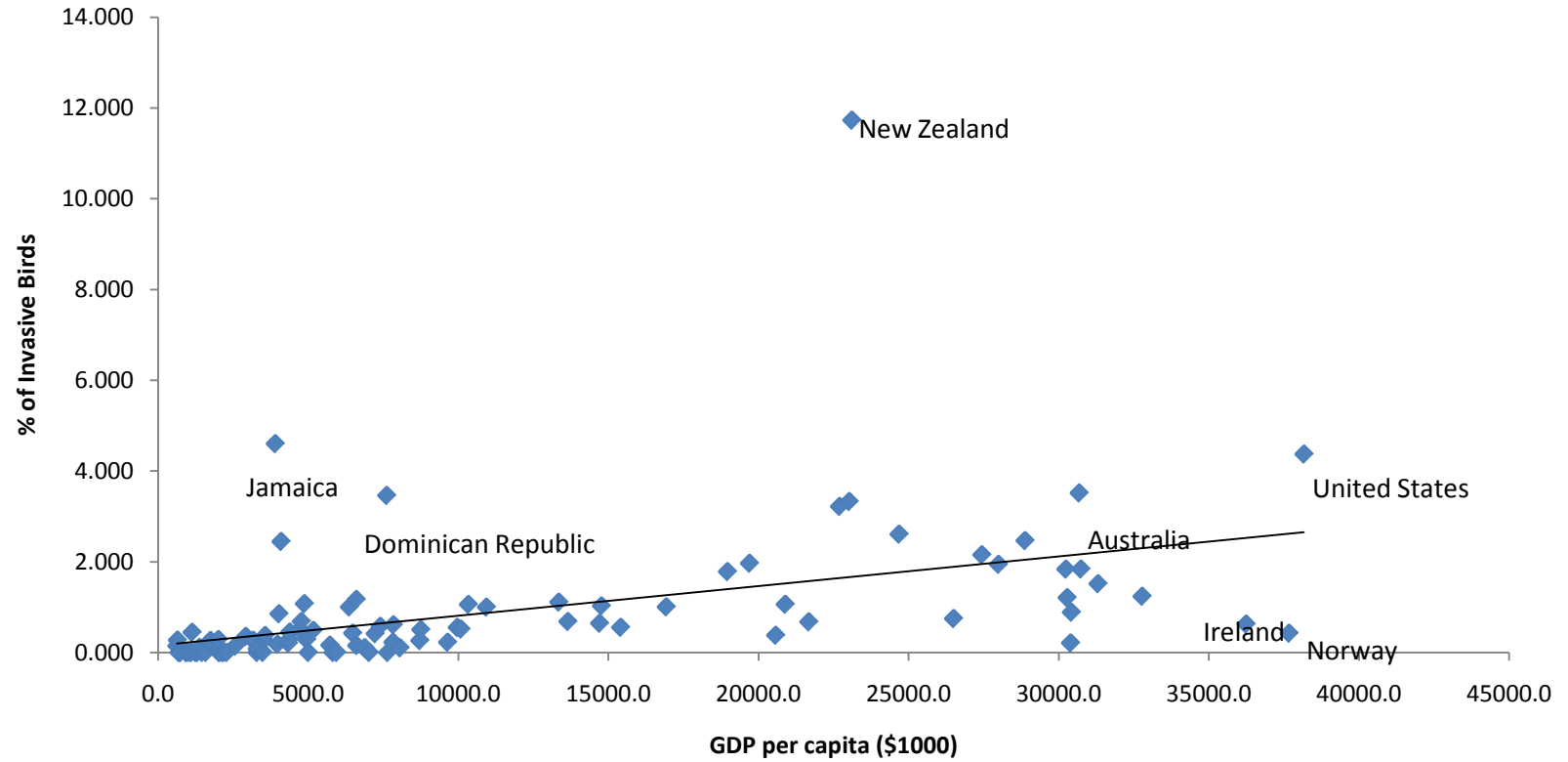


Figure 6.3. Invasive birds in relation to GDP per capita.

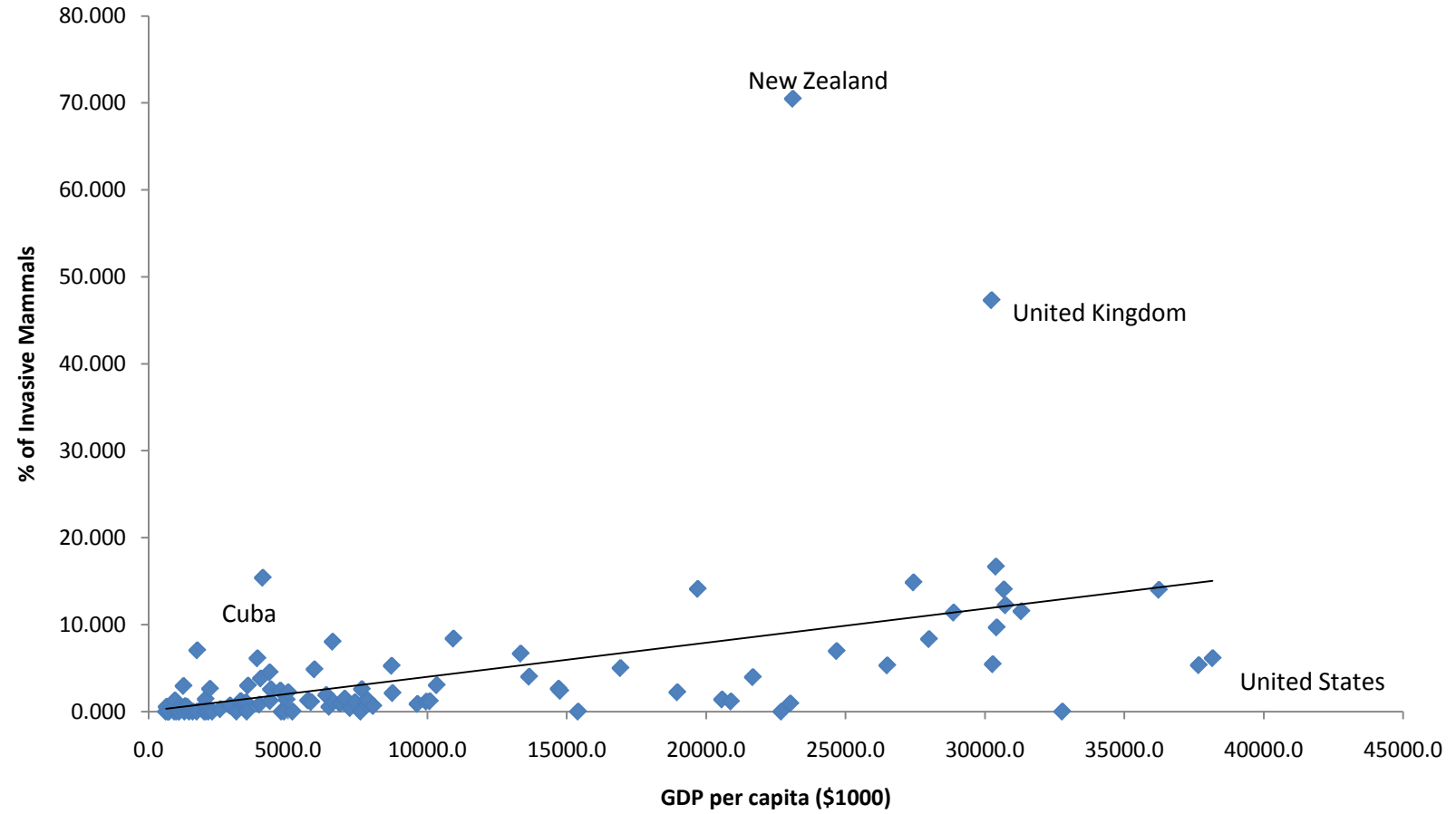


Figure 6.4. Invasive mammals in relation to GDP per capita.

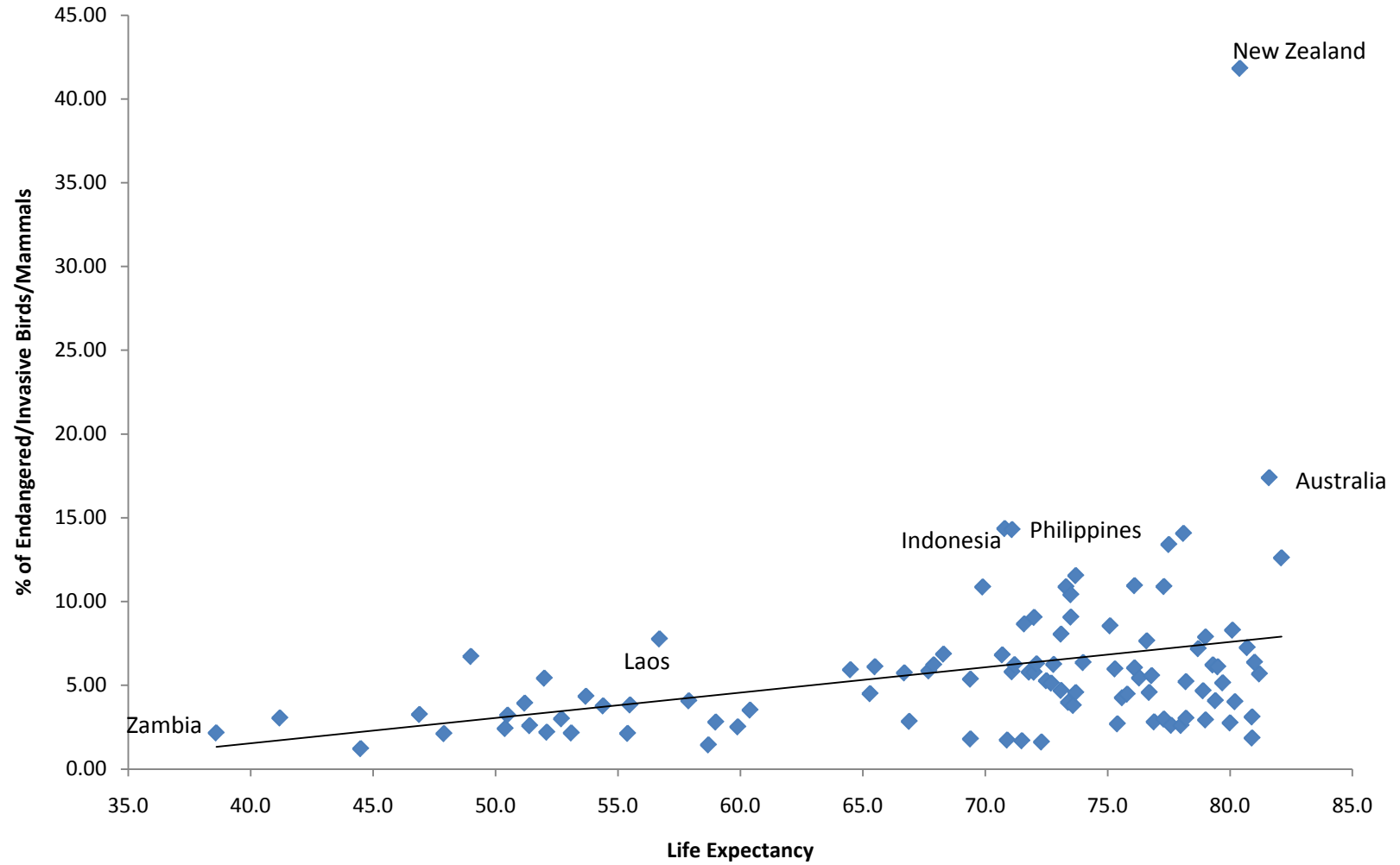


Figure 6.5. The percentage of endangered and invasive birds and mammals combined in relation to life expectancy. A higher percentage equates to a lower resilience.

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CHAPTER 7: CONCLUSION

In chapter 2 and chapter 3, I examined body mass distributions in five Mediterranean-climate ecosystems. Specifically, I examined the relationship between endangered and invasive bird and mammal species and their location along a body mass axis. I also explored changes in alpha, beta, and gamma diversity of function after the introduction of invasive bird and mammal species and the loss of endangered species. Discontinuous body-mass distributions were found in all Mediterranean-climate ecosystems and taxa examined. Discontinuities in body mass distributions have also been shown in North American birds (Skillen and Maurer 2008), south Florida herpetofauna, birds, and mammals (Allen 2006, Allen et al. 1999), Pleistocene and Miocene mammals (Lambert 2006, Lambert and Holling 1998), tropical forest birds (Restrepo et al. 1997), and boreal region birds and mammals (Holling 1992) and in various other taxa (reviewed in Sendzimir et al. 2003). This analysis extends these conclusions to include animal communities across Mediterranean-climate ecosystems of the world.

More non-indigenous species (NIS) and endangered species were found to occur at the edges of body mass aggregations than could be expected by chance alone in 40% of datasets and in all datasets when analyzed by taxonomic group pooling ecosystems, thus supporting similar analyses which examined the distribution of NIS and endangered species in relation to body mass aggregations (Allen et al. 1999). However, this was not consistent across Mediterranean-climate ecosystems or taxa. Skillen and Maurer (2008) reported an average of 52% of declining species were closer to body mass aggregation edges than the median distance to the nearest body mass aggregation edge, but showed an

average of 72% of NIS were further away from body mass aggregation edges than the median distance (i.e., located in gaps).

Parker et al. (1999) and Vitousek et al. (1996) describe many kinds of environmental effects associated with non-indigenous species: including genetic (loss of genetic diversity and evolutionary pressure due to hybridization), ecosystem (alteration of nutrient cycling and productivity), and population or community (species richness may increase, but abundance of native species declines). Although there were no significant differences between pre- and post- invasion Shannon Index values, results of this study demonstrated a trend of non-indigenous species positively affecting alpha diversity and affecting beta diversity of function in birds and mammals differently.

Alpha diversity of function increased in 9 out of the 10 Mediterranean-climate ecosystems analyzed when NIS were introduced into the community. Removal of endangered species from the community after introducing NIS yielded mixed results. Alpha diversity of function increased in eight ecosystems and decreased in two. Beta diversity of function in birds decreased in 80% of the Mediterranean-climate pair-wise comparisons analyzed when NIS were introduced and remained the same or decreased in 60% of the Mediterranean-climate pair-wise comparisons with the removal of endangered species from the communities. Beta diversity of function in mammals increased in 70% of the Mediterranean-climate pair-wise comparisons analyzed when NIS were introduced and remained the same or increased in 80% of comparisons when endangered species were subsequently removed from the communities. Gamma diversity of function in birds and mammals increased with the inclusion of NIS and after endangered species were removed.

Most functional group membership declines were in insectivorous birds and mammals. Within each individual body mass aggregation, there were changes in species composition and functional groups present, however, there were no differences in functional group richness preinvasion and post invasion for each terrestrial vertebrate group or for both taxonomic groups combined. There was a decrease in cross-scale redundancy of functional groups in mammals and when both taxonomic groups were combined.

Despite differences in pre- and post- invasion species communities, the number of functional groups in a given body mass aggregation (functional richness w/n scales) remained similar pre- and postinvasion. However, changes in the distribution of species within functional groups across different body mass aggregations in mammals and when both taxonomic groups were combined, further validate an apparent decrease in functional redundancy and cross-scale resilience. The loss of cross-scale resilience and simplification of these communities due to human influences is well documented and may also have unforeseen consequences (i.e., functional groups involved have changed) (Regier and Baskerville 1986, Peterson et al. 1998, Forys and Allen 2002). With regard to cross-scale redundancy in birds remaining the same, it may be that the overall net effect of the establishment of NIS birds in these ecosystems has offset losses of species due to other anthropogenic impacts (e.g., habitat fragmentation, degradation or destruction) (Forys and Allen 2002). This may also be a result of no niche replacement, but conservation of broad functional patterns via functional complementarity (Rosenfeld 2002).

In my fourth chapter, I examined changes in body mass distributions and speciation events over paleoecologic time. Discontinuous body-mass distributions were found in all Bridger and Uinta faunal assemblage zones (FAZ) examined. Discontinuities in body mass distributions that have been constructed from the fossil record have also been shown in Miocene mammals (Lambert 2006) and Pleistocene mammals (Lambert & Holling 1998). New species of mammals did not occur at body mass aggregation edges in the observed distribution more often than expected in either analysis conducted. The distance to edge of new species was not less than those species already present and was only significant in two FAZs when new species were placed into the prior geologic layer. This study identified and analyzed patterns in body mass distributions within paleoecological faunal assemblage zones, but the use of body mass may have been too coarse to detect speciation events.

In my fifth chapter, I examined changes in body mass distributions with changes in scale. Specifically, I examined whether the vertebrate body mass structures are similar among the ecoregions within each biome. A lack of similarity would provide strong evidence against the textural discontinuity hypothesis. If body mass structure reflects landscape structure at these different scales, this will support the textural discontinuity hypothesis and create a fundamental link between landscape and community ecology. All bird and mammal body-mass distributions were discontinuous at the ecoregion, biome, continental, and hemisphere scales. At the ecoregion, biome, continental, and hemisphere scales bird communities had more aggregations than mammal communities. Mammals had larger gap sizes than birds at every scale except hemispheric.

In my last chapter, I examined what socio-ecological factors within a country might help determine the number of endangered and invasive birds and mammals. The percentage of endangered birds in a country was positively correlated with total biodiversity and total land area. These results were similar to other analyses on the relationship between the percentage of endangered birds and total biodiversity (McKee et al. 2003, Orme et al. 2005). Although there was no evidence in the literature describing a relationship between the percentage of endangered birds and total land area, I could presume that an increase in sampling area would result in an increase in species richness (reviewed in Huston 1994) and therefore a possibility of there being more endangered species present.

The percentage of endangered mammals in a country was correlated with a combination of factors (i.e., the ecological class of variables). The results were similar to other analyses on the relationship between endangered mammals and agriculture intensity (Dobson et al. 1997), total annual rainfall and total biodiversity (McKee et al. 2003). Water stress has been reported as a threat to endangered species populations throughout the United States (Flather et al. 1998). There is evidence that the preservation of more habitat will allow for the survival of more species (Bruner et al. 2001), therefore I might assume fewer endangered species to be present. The inverse correlation observed in this study between wilderness protection and the percentage of endangered mammals suggests that this may be true. The relationship between GDP per capita and invasive species has been observed in other studies. GDP per capita and invasives have been found to correlate with the richness of alien spiders (Kobelt & Nentwig 2008), plants (Liu

et al. 2005), fishes (Leprieur et al. 2008), birds and mammals in Europe (Hulme 2007) and all taxonomic groups combined (Lin et al. 2007).

The idea of actively trying to reduce human life expectancy in order to increase the resilience of a country is one that will never be adopted by humanity. However, human population control is a potential option. Total population was not a factor in the selected models, but population control may be one of many crucial steps that must be taken in order to conserve global biodiversity (McKee et al. 2003). No other analysis to date has calculated a resilience factor for a country, thus there is no evidence in the literature regarding its relationship with human life expectancy. Dietz et al. (2007) found no relationship between life expectancy and environmental impact. The overall trend in high income countries, with improvements to the Human Development Index (which includes human life expectancy as one of its variables), is one towards a disproportionately larger negative impact on their ecological footprint. However, some lower income countries have high level of development without high impact on ecosystem services (Moran et al. 2008).

My work consisted of loosely connected empirical analyses that built upon each other to improve our knowledge of cross-scale ecosystem structure and contributed to the basic foundations of the field of landscape ecology, thus providing a broader understanding of how animals interact with their environment. This work provided further support for Holling's textural discontinuity hypothesis. The results of this research has provided future researchers with the ability to predict the impact and success of invasive species and the ability to determine which native species are at greatest risk of extinction. This research elucidated the distribution of biological diversity in space and

time and the socio-ecological factors that are contributing to the worldwide increase in invasive and endangered species. Humans are an integral part of the ecosystems in which they inhabit, thus understanding their role is crucial. The negative socio-ecological impacts of humans will continue to increase, thereby altering the architecture of the landscape and changing the composition of animal communities.

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APPENDIX A. Mediterranean-climate ecosystem mammals in California. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Sorex ornatus</i>	Ornate Shrew	0.698	1	InTe	
<i>Sorex vagrans</i>	Vagrant Shrew	0.707	1	InTe	
<i>Reithrodontomys megalotis</i>	Harvest Mouse	1.049	1	GrTe	
<i>Peromyscus maniculatus</i>	Deer Mouse	1.299	1	GrTe	
<i>Peromyscus boylii</i>	Brush Mouse	1.329	1	HeAr	
<i>Perognathus californicus</i>	California Pocket Mouse	1.367	1	GrTe	
<i>Peromyscus truei</i>	Pinyon Mouse	1.427	1	GrTe	
<i>Peromyscus californicus</i>	California Mouse	1.656	2	GrTe	
<i>Tamias obscurus</i>	Obscure Chipmunk	1.748	2	HeAr	
<i>Dipodomys stephensi</i>	Stephen's Kangaroo Kat	1.807	2	GrTe	x
<i>Dipodomys heermanni</i>	Heermann Kangaroo Kat	1.857	2	GrTe	
<i>Tamias merriami</i>	Merriam's Chipmunk	1.875	2	GrTe	
<i>Dipodomys venustus</i>	Santa Cruz Kangaroo Rat	1.929	2	GrTe	
<i>Dipodomys elephantinus</i>	Big-eared Kangaroo Rat	1.93	2	GrTe	
<i>Thomomys bottae</i>	Botta Pocket Gopher	2.049	2	HeFs	
<i>Neotoma lepida</i>	Desert Wood Rat	2.164	2	HeTe	
<i>Mustela frenata</i>	Long-tailed Weasel	2.167	2	CaTe	
<i>Peromyscus eremicus</i>	Cactus Mouse	2.276	2	GrTe	
<i>Neotoma fuscipes</i>	Dusky-footed Woodrat	2.281	2	HeAr	
<i>Spermophilus beecheyi</i>	California Ground Squirrel	2.781	3	HeTe	
<i>Sylvilagus bachmani</i>	Brush Rabbit	2.785	3	HeTe	
<i>Sylvilagus auduboni</i>	Audubon Cottontail	2.879	3	HeTe	
<i>Spilogale gracilis</i>	Spotted Skunk	2.888	3	CaTe	
<i>Bassaricus astutus</i>	Ringtail	3.053	3	CaTe	x

APPENDIX A. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Mephitis mephitis</i>	Striped Skunk	3.253	3	InTe	
<i>Urocyon cinereoargenteus</i>	Gray Fox	3.548	4	CaTe	
<i>Procyon lotor psora</i>	Raccoon	3.557	4	OmTe	
<i>Taxidea taxus</i>	Badger	3.857	4	CaTe	
<i>Lynx rufus</i>	Bobcat	3.889	4	CaTe	
<i>Canis latrans</i>	Coyote	4.102	4	CaTe	
<i>Odocoileus hemionus</i>	Mule Deer	4.635	5	HeTe	
<i>Felis concolor</i>	Cougar	4.754	5	CaTe	
<i>Felis onca</i>	Jaguar	5.061	5	CaTe	x
<i>Ursos arctos</i>	Grizzly Bear	5.19	5	OmTe	x
Non-Indigenous Species					
<i>Mus musculus</i>	House Mouse	1.205		HeTe	
<i>Rattus rattus</i>	Black Rat	2.328		OmTe	
<i>Rattus norvegicus</i>	Norway Rat	2.384		OmTe	
<i>Sciurus niger</i>	Fox Squirrel	2.875		OmTe	
<i>Didelphis virginianus</i>	Common Opossum	3.301		OmTe	
<i>Castor canadensis</i>	Beaver	4.297		HeAq	

APPENDIX B. Mediterranean-climate ecosystem mammals in Chile. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin name	Common name	Mass(g)	BCART	Functional Group	Endangered
<i>Marmosa elegans</i>	Mouse Opossum	1.481	1	InTe	
<i>Oryzomys longicaudatus</i>	Rice Rat	1.560	1	GrTe	
<i>Akodon olivaceus</i>	Olivaceous Akodon	1.639	1	GrTe	
<i>Phyllotis darwini</i>	Darwin's Leaf-eared Mouse	1.789	1	HeTe	
<i>Akodon longipilis</i>	Long-haired Akodon	1.796	1	InTe	
<i>Notiomys megalonyx</i>	Mole Mouse	1.830	1	GrTe	
<i>Chelemys macronyx</i>	Field Mole Mouse	1.865	1	GrTe	
<i>Euneonys mordax</i>	Biting Chinchilla Mouse	1.914	1	HeTe	
<i>Octodon bridgesi</i>	Bridges' Degu	1.966	1	HeTe	
<i>Spalacopus cyanus</i>	Coruro	2.011	1	HeFs	
<i>Aconaemys fuscus</i>	Chilean Rock Rat	2.090	1	HeTe	
<i>Ctenomys maulinus</i>	Maule Tuco-Tuco	2.215	1	HeTe	
<i>Octodon degus</i>	Degu	2.264	1	HeAr	
<i>Abrocoma bennetti</i>	Chinchilla Rat	2.363	1	HeAr	
<i>Octodon lunatus</i>	Moon-toothed Degu	2.367	1	HeTe	
<i>Lagidium viscacia</i>	Mountain Vizcacha	3.188	2	HeTe	
<i>Galictis guia</i>	Grison	3.199	2	CaTe	
<i>Conepatus chinga</i>	Molina's Hog-nosed Skunk	3.275	2	InTe	
<i>Felis Guigna</i>	Kokod	3.348	2	CaTe	
<i>Felis Colocolo</i>	Pampas Cat	3.470	2	CaAr	
<i>Myocaster coypus</i>	Nutria	3.579	2	HeAq	
<i>Dusicyon griseus</i>	Gray Fox	3.601	2	CaTe	
<i>Lutra felina</i>	Marine Otter	3.653	2	InAq	
<i>Dusicyon culpaeus</i>	Culpeo Fox	3.867	3	CaTe	

APPENDIX B. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Lutra provocax</i>	Southern River Otter	3.880	3	CaAq	
<i>Pudu puda</i>	Pudu	3.989	3	HeTe	X
<i>Felis concolor</i>	Mountain Lion	4.549	4	CaTe	
<i>Hippocamelus bisulcus</i>	Patagonian Huemul	4.845	4	HeTe	X
<i>Lama guanicoe</i>	Guanaco	5.079	4	HeTe	X
Non-Indigenous Species					
<i>Mus musculus</i>	House Mouse	1.230		CaAq	
<i>Rattus rattus</i>	Black Rat	2.328		CaTe	
<i>Rattus norvegicus</i>	Norway Rat	2.384		HeAq	
<i>Ondatra zibethica</i>	Muskrat	3.009		HeAq	
<i>Mustela vison</i>	Mink	3.167		HeTe	
<i>Oryctolagus cuniculus</i>	European Rabbit	3.196		HeTe	
<i>Lepus capensis</i>	Brown Hare	3.301		HeTe	
<i>Felis catus</i>	Feral Cat	3.591		CaTe	
<i>Castor canadensis</i>	North American Beaver	4.297		HeAq	
<i>Dama dama</i>	Fallow Deer	4.686		HeTe	
<i>Sus scrofa</i>	Wild Hog	4.740		OmTe	
<i>Cervus elaphus</i>	Red Deer	5.188		HeTe	

APPENDIX C. Mediterranean-climate ecosystem mammals in Spain. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Suncus etruscus</i>	Pygmy White-toothed Shrew	0.352	1	InTe	
<i>Sorex minutus</i>	Pygmy Shrew	0.477	1	InTe	
<i>Micromys minutus</i>	Harvest Mouse	0.756	1	GrTe	
<i>Sorex granarius</i>	Spanish Shrew	0.796	1	GrTe	X
<i>Crocidura russula</i>	Greater White-toothed Shrew	0.806	1	InTe	
<i>Crocidura suaveolens</i>	Lesser White-toothed Shrew	0.825	1	InTe	
<i>Mus spretus</i>	Algerian Mouse	1.090	2	GrTe	
<i>Neomys fodiens</i>	Water Shrew	1.114	2	InAq	
<i>Neomys anomalus</i>	Miller's Water Shrew	1.134	2	InTe	
<i>Pitymys lusitanicus</i>	Lusitanian Pine Vole	1.212	2	HeTe	
<i>Microtus arvalis</i>	Common Vole	1.262	2	HeFs	
<i>Clethrionomys glareolus</i>	Bank Vole	1.288	2	HeTe	
<i>Apodemus sylvaticus</i>	Wood Mouse	1.344	2	InTe	
<i>Microtis agrestis</i>	Field Vole	1.344	2	HeTe	
<i>Pitymys duodecimcostatus</i>	Mediterranean Pine Vole	1.345	2	HeTe	
<i>Talpa caeca</i>	Blind Mole	1.505	3	InFs	
<i>Microtus nivalis</i>	Snow Vole	1.591	3	HeTe	
<i>Microtus cabrerae</i>	Cabrera's vole	1.645	3	HeTe	X
<i>Galemys pyrenaicus</i>	Pyrenean Desman	1.760	3	InAq	
<i>Talpa europaea</i>	Common Mole	1.881	3	InFs	
<i>Talpa romana</i>	Roman Mole	1.966	3	InFs	
<i>Eliomys quercinus</i>	Garden Dormouse	1.980	3	HeTe	X
<i>Mustela nivalis</i>	Weasel	2.150	4	CaTe	
<i>Arvicola sapidus</i>	Southern Water Vole	2.230	4	HeAq	X

APPENDIX C. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Myoxis glis</i>	Edible Dormouse	2.284	4	HeAr	
<i>Sciurus vulgaris</i>	Red Squirrel	2.398	4	HeAr	X
<i>Erinaceus europaeus</i>	Western Hedgehog	2.805	5	InTe	
<i>Erinaceus algirus</i>	Algerian Hedgehog	2.845	5	InTe	
<i>Martes foina</i>	Beech Marten	3.000	5	CaTe	
<i>Mustela putorius</i>	Western Polecat	3.000	5	CaTe	
<i>Oryctolagus cuniculus</i>	Rabbit	3.196	6	HeTe	
<i>Lepus capensis</i>	Brown Hare	3.310	6	HeTe	
<i>Lepus granatensis</i>	Iberian Hare	3.334	6	HeTe	
<i>Felis silvestris</i>	Wildcat	3.385	6	CaTe	
<i>Vulpes vulpes</i>	Red Fox	3.678	7	CaTe	
<i>Lutra lutra</i>	Otter	3.796	7	CaAq	X
<i>Meles meles</i>	Eurasian Badger	3.964	7	InTe	
<i>Lynx pardinus</i>	Pardel Lynx	4.043	7	CaTe	X
<i>Macaca sylvanus</i>	Barbary Ape	4.049	7	HeTe	
<i>Capreolus capreolus</i>	Roe Deer	4.079	7	HeTe	
<i>Castor fiber</i>	European Beaver	4.145	7	HeTe	
<i>Canis lupus</i>	Wolf	4.632	8	CaTe	
<i>Sus scrofa</i>	Wild Boar	4.740	8	HeTe	
<i>Capra pyrenaica</i>	Spanish Ibex	4.760	8	HeTe	X
<i>Cervus elaphus</i>	Red Deer	5.176	8	HeTe	
<i>Ursus arctos</i>	Brown Bear	5.247	8	HeTe	
Non-Indigenous Species					
<i>Mus musculus</i>	House Mouse	1.283		GrTe	
<i>Rattus rattus</i>	Black Rat	2.092		HeTe	

APPENDIX C. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Rattus norvegicus</i>	Norway Rat	2.455		HeTe	
<i>Atelerix algirus</i>	Algerian Hedgehog	3.097		InTe	
<i>Genetta genetta</i>	Common Genet	3.236		CaTe	
<i>Herpestes ichneumon</i>	Egyptian Mongoose	3.474		CaTe	
<i>Ammotragus lervia</i>	Barbary Sheep	4.926		HeTe	

APPENDIX D. Mediterranean-climate ecosystem mammals in South Africa. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Mus minutoides</i>	Pygmy Mouse	0.767	1	HeTe	
<i>Suncus varilla</i>	Lesser Dwarf Shrew	0.813	1	InTe	
<i>Dendromus melanotis</i>	Grey Climbing Mouse	0.826	1	InTe	
<i>Crocidura cyanea</i>	Reddish-grey Musk Shrew	0.934	1	InTe	
<i>Dendromus mesomelas</i>	Brant's Climbing Mouse	1.053	1	InTe	
<i>Malacothrix typica</i>	Large-eared Mouse	1.127	1	HeTe	
<i>Myosorex varius</i>	Forest Shrew	1.130	1	InTe	
<i>Acomys subspinosus</i>	Cape Spiny Mouse	1.325	2	HeTe	
<i>Steatomys krebsi</i>	Kreb's Fat Mouse	1.380	2	HeTe	
<i>Graphiurus murinus</i>	Woodland Dormouse	1.450	2	InTe	
<i>Gerbillurus paeba</i>	Hairy-footed Gerbil	1.511	2	HeTe	
<i>Rhabdomys pumilo</i>	Striped Mouse	1.559	2	HeTe	
<i>Macroscelides proboscideus</i>	Round-eared Elephant Shrew	1.582	2	InTe	
<i>Myomyscus verroxii</i>	Verreaux's Mouse	1.613	2	InTe	
<i>Desmmodillus auricularis</i>	Short-tailed Gerbil	1.664	2	HeTe	
<i>Aethomys namequensis</i>	Namaqua Rock Mouse	1.688	2	HeTe	
<i>Chrysochloris asiatica</i>	Cape Golden Mole	1.690	2	InFs	
<i>Amblysomus hottentotus</i>	Hottentot Golden Mole	1.832	3	InFs	
<i>Graphiurus ocularis</i>	Spectacled Dormouse	1.838	3	InTe	
<i>Cryptomys hottentotus</i>	Common Molerat	1.897	3	HeFs	
<i>Mystromys albicaudatus</i>	White-tailed Mouse	1.939	3	HeTe	X
<i>Dasymys incomtus</i>	Water Rat	1.972	3	HeTe	
<i>Tatera afra</i>	Cape Gerbil	1.987	3	HeTe	
<i>Otomys saundersiae</i>	Saunders's Vlei Rat	2.013	3	HeTe	

APPENDIX D. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Otomys irroratus</i>	Vlei Rat	2.072	3	HeTe	
<i>Crocidura flavescens</i>	Greater Musk Shrew	2.088	3	InTe	
<i>Otomys unisulcatus</i>	Bush Karoo Rat	2.095	3	HeTe	
<i>Otomys laminatus</i>	Laminate Vlei Rat	2.176	3	HeTe	
<i>Georychus capensis</i>	Cape Molerat	2.338	3	HeTe	
<i>Poecilogale albinucha</i>	Striped Weasel	2.338	3	CaTe	
<i>Bathyergus suillus</i>	Cape Dune Molerat	2.796	4	HeFs	
<i>Ictonyx striatus</i>	Striped Polecat	2.866	4	InTe	
<i>Herpestes pulverulenta</i>	Small Grey Mongoose	2.901	4	InTe	
<i>Cynictis penicillata</i>	Yellow Mongoose	2.919	4	InTe	
<i>Pronolagus rupestris</i>	Smith's Red Rock Rabbit	3.210	5	HeTe	
<i>Genetta tigrina</i>	Large-spotted Genet	3.270	5	CaTe	
<i>Genetta genetta</i>	Small-spotted Genet	3.279	5	CaTe	
<i>Lepus capensis</i>	Cape Hare	3.310	5	HeTe	
<i>Vulpes chama</i>	Cape Fox	3.423	5	CaTe	
<i>Procavia capensis</i>	Rock Dassie	3.480	5	HeTe	
<i>Atilax paludinosus</i>	Water Mongoose	3.531	5	CaTe	
<i>Lepus saxatilis</i>	Scrub Hare	3.556	5	HeTe	
<i>Felis libyca</i>	African Wildcat	3.633	5	CaTe	
<i>Proteles cristatus</i>	Aardwolf	3.840	6	InTe	
<i>Canis mesomelas</i>	Black-backed Jackal	3.898	6	CaTe	
<i>Mellivora capensis</i>	Honey Badger	3.899	6	CaTe	
<i>Oreotragus oreotragus</i>	Klipspringer	4.009	6	HeTe	X
<i>Raphicerus melanotis</i>	Grysbok	4.011	6	HeTe	X
<i>Felis caracal</i>	Caracal	4.029	6	CaTe	

APPENDIX D. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Felis serval</i>	Serval	4.047	6	CaTe	
<i>Raphicerus campestris</i>	Steenbok	4.053	6	HeTe	
<i>Aonyx capensis</i>	Clawless Otter	4.061	6	CaAq	
<i>Hystrix africaeaustralis</i>	Porcupine	4.097	6	HeTe	
<i>Sylvicapra grimmia</i>	Grey Duiker	4.207	6	HeTe	
<i>Pelea capreolus</i>	Grey Ribbok	4.352	6	HeTe	
<i>Papio ursinus</i>	Chacma Baboon	4.365	6	HeTe	
<i>Panthera pardus</i>	Leopard	4.416	6	CaTe	
<i>Hyaena brunnea</i>	Brown Hyaena	4.583	6	CaTe	X
<i>Orycteropus afer</i>	Aardvark	4.719	6	InTe	
<i>Damaliscus dorcas dorcas</i>	Bontebuck	4.826	6	HeTe	X
<i>Alcelaphus buselaphus</i>	Hartebeest	5.134	7	HeTe	X
<i>Panthera leo</i>	Lion	5.193	7	CaTe	X
<i>Hippotragus leucophaeus</i>	Bluebuck	5.204	7	HeTe	X
<i>Equus zebra</i>	Cape Mountain Zebra	5.388	7	HeTe	X
<i>Tragelaphus oryx</i>	Eland	5.587	7	HeTe	X
<i>Diceros bicornis</i>	Black Rhinoceros	5.939	7	HeTe	X
Non-Indigenous Species					
<i>Mus musculus</i>	House Mouse	1.205		HeTe	
<i>Rattus rattus</i>	Black Rat	2.045		OmTe	
<i>Rattus norvegicus</i>	Norway Rat	2.455		OmTe	
<i>Sciurus carolinensis</i>	Grey Squirrel	2.763		GrAr	
<i>Felis catus</i>	Domestic Cat	3.533		CaTe	
<i>Hemitragus jemlahicus</i>	Himalayan Tahr	4.167		HeTe	
<i>Cervus dama</i>	Fallow Deer	4.686		HeTe	

APPENDIX D. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Sus scrofa</i>	Feral Pig	4.740		OmTe	

APPENDIX E. Mediterranean-climate ecosystem mammals in southwestern Australia. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Tarsipes rostratus</i>	Honey Possum	0.954	1	HeAr	
<i>Cercartetus concinnus</i>	Western Pygmy-possum	1.114	1	InAr	
<i>Sminthopsis dolichura</i>	Little Long-tailed Dunnart	1.134	1	InTe	
<i>Sminthopsis crassicaudata</i>	Fat-tailed Dunnart	1.176	1	InTe	
<i>Sminthopsis griseoventer</i>	Grey-bellied Dunnart	1.243	1	InTe	X
<i>Sminthopsis gilberti</i>	Gilbert's Dunnart	1.290	1	InTe	
<i>Sminthopsis granulipes</i>	White-tailed Dunnart	1.398	1	InTe	
<i>Pseudomys albocinereus</i>	Ash-grey Mouse	1.484	1	HeTe	
<i>Pseudomys occidentalis</i>	Western Mouse	1.531	1	HeTe	X
<i>Pseudomys nanus</i>	Western Chestnut Mouse	1.531	1	HeTe	X
<i>Notomys alexis</i>	Spinifex Hopping Mouse	1.544	1	GrTe	
<i>Pseudomys fieldi</i>	Shark Bay Mouse	1.653	1	HeTe	X
<i>Antichinus flavipes</i>	Yellow-footed Antechinus	1.653	1	InTe	
<i>Phascogale calura</i>	Red-tailed Phascogale	1.712	1	InAr	X
<i>Notomys mitchelli</i>	Mitchell's Hopping Mouse	1.716	1	GrTe	
<i>Notomys macrotis</i>	Big-eared Hopping Mouse	1.720	1	GrTe	X
<i>Parantechinus apicalis</i>	Southern Dibbler	1.837	1	InTe	X
<i>Pseudomys shortridgei</i>	Heath Rat	1.845	1	HeTe	
<i>Notomys longicaudatus</i>	Long-tailed Hopping Mouse	2.000	2	GrTe	X
<i>Rattus tunneyi</i>	Pale Field-Rat	2.093	2	HeTe	X
<i>Rattus fuscipes</i>	Bush Rat	2.122	2	InTe	
<i>Phascogale tapoatafa</i>	Brush-tailed Phascogale	2.287	2	InAr	X
<i>Perameles bougainville</i>	Western Barred Bandicoot	2.354	2	HeTe	X
<i>Myrmecobius fasciatus</i>	Numbat	2.673	3	InTe	X

APPENDIX E. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Hydromys chrysogaster</i>	Water Rat	2.833	3	InAq	
<i>Potorous platyops</i>	Broad-faced Potoroo	2.845	3	HeTe	X
<i>Isodon obesulus</i>	Southern Brown Bandicoot	2.889	3	InTe	
<i>Pseudocheirus occidentalis</i>	Western ringtail Possum	3.000	3	HeAr	X
<i>Dasyurus geoffroii</i>	Western Quoll	3.041	3	CaTe	X
<i>Potorus tridactylus</i>	Long-nosed Potoroo	3.041	3	HeTe	X
<i>Lagorchestes hirsutus</i>	Rufous Hare-wallaby	3.102	3	HeTe	X
<i>Bettongia penicillata</i>	Brush-tailed Bettong	3.114	3	HeTe	X
<i>Bettongia leseur</i>	Burrowing Betong	3.176	3	HeTe	X
<i>Lagostrophus fasciatus</i>	Banded Hare-wallaby	3.230	3	HeTe	X
<i>Trichosurus vulpecula</i>	Common Brushtail Possum	3.419	4	HeAr	
<i>Setonix brachyurus</i>	Quokka	3.512	4	HeTe	X
<i>Onychogalea lunata</i>	Crescent Nailtail Wallaby	3.544	4	HeTe	X
<i>Petrogale lateralis</i>	Black-footed Rock-wallaby	3.602	4	HeTe	X
<i>Tachyglossus aculeatus</i>	Short-beaked Echidna	3.653	4	InTe	
<i>Macropus eugenii</i>	Tammar Wallaby	3.813	4	HeTe	X
<i>Macropus irma</i>	Western Brush Wallaby	3.903	4	HeTe	X
<i>Canis lupus</i>	Dingo	4.225	5	CaTe	X
<i>Macropus robustus</i>	Common Wallaroo	4.327	5	HeTe	
<i>Macropus fuliginosus</i>	Western Grey Kangaroo	4.345	5	HeTe	
Non-Indigenous Species					
<i>Mus musculus</i>	House Mouse	1.217		HeTe	
<i>Rattus rattus</i>	Black Rat	2.447		OmTe	
<i>Rattus norvegicus</i>	Brown Rat	2.505		OmTe	
<i>Oryctolagus cuniculus</i>	European Rabbit	3.199		HeTe	

APPENDIX E. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Felis catus</i>	Cat	3.626		CaTe	
<i>Vulpes vulpes</i>	Red Fox	3.775		CaTe	
<i>Capra hircus</i>	Goat	4.525		HeTe	
<i>Sus scrofa</i>	Feral Hog	4.936		OmTe	

APPENDIX F. Mediterranean-climate ecosystem birds in California. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass (g)	BCART	Functional Group	Endangered
<i>Calypte costae</i>	Costa's Hummingbird	0.491	1	HeAe	
<i>Cynanthus latirostris</i>	Broad-billed Hummingbird	0.491	1	HeAe	
<i>Archilochus alexandri</i>	Black-chinned Hummingbird	0.531	1	HeAe	
<i>Calypte anna</i>	Anna's Hummingbird	0.623	1	HeAe	
<i>Poliophtila melanura</i>	Black-tailed Gnatcatcher	0.708	1	InFo	
<i>Psaltiriparus minimus</i>	Bushtit	0.724	1	InFo	
<i>Poliophtila caerulea</i>	Blue-gray Gnatcatcher	0.778	1	InFo	
<i>Wilsonia pusilla</i>	Wilson's Warbler	0.839	1	InFo	
<i>Dendroica nigrescens</i>	Black-throated Gray Warbler	0.922	2	InFo	
<i>Vireo bellii pusillus</i>	Least Bell's Vireo	0.929	2	InFo	x
<i>Vermivora celata</i>	Orange-crowned Warbler	0.954	2	InFo	
<i>Dendroica petechia</i>	Yellow Warbler	0.978	2	GrFo	x
<i>Carduelis psaltria</i>	Lesser Goldfinch	0.978	2	InFo	
<i>Thryomanes bewickii</i>	Bewick's Wren	0.996	2	InTe	
<i>Empidonax difficilis</i>	Pacific-slope Flycatcher	1.000	2	InAe	
<i>Geothlypis trichas</i>	Common Yellowthroat	1.004	2	InFo	
<i>Poecile gambeli</i>	Mountain Chickadee	1.033	2	InFo	
<i>Troglodytes aedon</i>	House Wren	1.037	2	InTe	
<i>Carduelis lawrencei</i>	Lawrence's Goldfinch	1.039	2	GrFo	
<i>Cistothorus palustris</i>	Marsh Wren	1.051	2	InTe	
<i>Vireo huttoni huttoni</i>	Hutton's Vireo	1.064	2	InFo	
<i>Spizella atrogularis cana</i>	Black-chinned Sparrow	1.076	2	InTe	
<i>Spizella passerina</i>	Chipping Sparrow	1.090	2	InTe	

APPENDIX F. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Catherpes mexicanus</i>	Canyon Wren	1.100	2	InTe	
<i>Contopus sordidulus</i>	Western Wood-Pewee	1.107	2	InAe	
<i>Vireo vicinior</i>	Gray Vireo	1.107	2	InFo	x
<i>Carduelis tristis</i>	American Goldfinch	1.111	2	GrFo	
<i>Empidonax traillii</i>	Willow Flycatcher	1.127	2	InAe	x
<i>Tachycineta thalassina</i>	Violet-green Swallow	1.151	2	InAe	
<i>Chamaea fasciata</i>	Wrentit	1.166	2	InFo	
<i>Vireo gilvus</i>	Warbling Vireo	1.170	2	InFo	
<i>Stelgidopteryx serripennis</i>	Northern Rough-winged Swallow	1.182	2	InAe	
<i>Passerina amoena</i>	Lazuli Bunting	1.190	2	InTe	
<i>Ammodramus savannarum</i>	Grasshopper Sparrow	1.230	3	InTe	x
<i>Baeolophus inornatus</i>	Oak Titmouse	1.243	3	InAe	
<i>Sayornis nigricans</i>	Black Phoebe	1.271	3	InAe	
<i>Aimophila ruficeps</i>	Rufous-crowned Sparrow	1.272	3	InTe	
<i>Amphispiza belli</i>	Sage Sparrow	1.286	3	InTe	
<i>Melospiza melodia</i>	Song Sparrow	1.291	3	InTe	
<i>Sitta carolinensis</i>	White-breasted Nuthatch	1.324	3	InBa	
<i>Sayornis saya</i>	Say's Phoebe	1.326	3	InAe	
<i>Carpodacus mexicanus</i>	House Finch	1.330	3	GrTe	
<i>Petrochelidon pyrrhonota</i>	Cliff Swallow	1.334	3	InAe	
<i>Phainopepla nitens</i>	Phainopepla	1.380	4	HeFo	
<i>Icterus cucullatus</i>	Hooded Oriole	1.386	4	InFo	
<i>Carpodacus purpureus</i>	Purple Finch	1.396	4	GrTe	
<i>Icteria virens auricollis</i>	Yellow-breasted Chat	1.403	4	InFo	
<i>Picoides pubescens</i>	Downy Woodpecker	1.431	4	InBa	

APPENDIX F. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Myiarchus cinerascens</i>	Ash-throated Flycatcher	1.435	4	InAe	
<i>Sialia mexicana</i>	Western Bluebird	1.448	4	InAe	
<i>Passerina caerulea</i>	Blue Grosbeak	1.453	4	InTe	
<i>Chondestes grammacus</i>	Lark Sparrow	1.462	4	GrTe	
<i>Pipilo chlorurus</i>	Green-tailed Towhee	1.468	4	InTe	
<i>Catharus ustulatus</i>	Swainson's Thrush	1.489	4	InFo	
<i>Eremophila alpestris</i>	Horned Lark	1.496	4	GrTe	x
<i>Aeronautes saxatalis</i>	White-throated Swift	1.507	4	InAe	
<i>Passerella iliaca</i>	Fox Sparrow	1.509	4	InTe	
<i>Icterus bullockii</i>	Bullock's Oriole	1.526	4	InFo	
<i>Picoides nuttallii</i>	Nuttall's Woodpecker	1.583	5	InBa	
<i>Campylorhynchus brunneicapillus</i>	Cactus Wren	1.590	5	InTe	
<i>Tyrannus verticalis</i>	Western Kingbird	1.598	5	InAe	
<i>Pheucticus melanocephalus</i>	Black-headed Grosbeak	1.623	5	InFo	
<i>Molothrus ater</i>	Brown-headed Cowbird	1.642	5	InTe	
<i>Tyrannus vociferans</i>	Cassin's Kingbird	1.659	5	InAe	
<i>Lanius ludovicianus</i>	Loggerhead Shrike	1.676	5	InAe	
<i>Mimus polyglottos</i>	Northern Mockingbird	1.686	5	InTe	
<i>Progne subis subis</i>	Purple Martin	1.694	5	InAe	
<i>Chordeiles acutipennis</i>	Lesser Nighthawk	1.698	5	InAe	
<i>Phalaenoptilus nuttallii</i>	Common Poorwill	1.713	5	InAe	
<i>Agelaius phoeniceus</i>	Red-winged Blackbird	1.721	5	InTe	
<i>Agelaius tricolor</i>	Tricolored Blackbird	1.769	6	InTe	x
<i>Euphagus cyanocephalus</i>	Brewer's Blackbird	1.797	6	InTe	
<i>Coccyzus americanus</i>	Yellow-billed Cuckoo	1.806	6	InFo	x

APPENDIX F. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Xanthocephalus xanthocephalus</i>	Yellow-headed Blackbird	1.810	6	InTe	x
<i>Porzana carolina</i>	Sora	1.873	7	GrTe	x
<i>Aphelocoma californica</i>	Western Scrub-Jay	1.904	7	OmTe	
<i>Melanerpes formicivorus</i>	Acorn Woodpecker	1.906	7	OmBa	
<i>Rallus limicola limicola</i>	Virginia Rail	1.914	7	InAq	
<i>Toxostoma redivivum</i>	California Thrasher	1.926	7	InTe	
<i>Ixobrychus exilis hesperis</i>	Least Bittern	1.936	7	CaAq	
<i>Charadrius vociferus</i>	Killdeer	1.985	7	InTe	
<i>Sturnella neglecta</i>	Western Meadowlark	2.003	7	InTe	
<i>Falco sparverius</i>	American Kestrel	2.063	7	InAe	
<i>Zenaida macroura</i>	Mourning Dove	2.076	7	GrTe	
<i>Colaptes auratus</i>	Northern Flicker	2.102	7	InTe	
<i>Cyanocitta stelleri</i>	Steller's Jay	2.107	7	OmTe	
<i>Megascops kennicottii</i>	Western Screech Owl	2.155	7	CaAe	
<i>Athene cunicularia</i>	Burrowing Owl	2.190	7	InAe	x
<i>Callipepla gambelii</i>	Gambel's Quail	2.220	7	GrTe	
<i>Callipepla californica</i>	California Quail	2.238	7	GrTe	
<i>Butorides virescens anthonyi</i>	Green Heron	2.326	7	CaAq	
<i>Oreortyx pictus</i>	Mountain Quail	2.367	7	GrTe	
<i>Asio otus wilsonianus</i>	Long-eared Owl	2.418	7	CaAe	x
<i>Elanus leucurus</i>	White-tailed Kite	2.522	8	CaAe	x
<i>Geococcyx californianus</i>	Greater Roadrunner	2.575	8	InTe	
<i>Patagioenas fasciata</i>	Band-tailed Pigeon	2.593	8	HeFo	
<i>Circus cyaneus hudsonius</i>	Northern Harrier	2.639	8	CaAe	

APPENDIX F. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Accipiter cooperii</i>	Cooper's Hawk	2.642	8	CaAe	
<i>Corvus brachyrhynchos</i>	American Crow	2.651	8	OmTe	
<i>Tyto alba pratincola</i>	Barn Owl	2.719	8	CaAe	
<i>Buteo lineatus</i>	Red-shouldered Hawk	2.747	8	CaAe	
<i>Strix occidentalis</i>	Spotted Owl	2.785	8	CaAe	x
<i>Falco mexicanus</i>	Prairie Falcon	2.850	8	CaAe	
<i>Dendrocygna bicolor</i>	Fulvous Whistling-duck	2.851	8	HeAq	x
<i>Falco peregrinus</i>	Peregrine Falcon	2.893	8	CaAe	x
<i>Nycticorax nycticorax</i>	Black-crowned Night Heron	2.946	8	CaAq	
<i>Buteo swainsoni</i>	Swainson's Hawk	2.995	8	CaAe	x
<i>Buteo jamaicensis</i>	Red-tailed Hawk	3.052	8	CaAe	
<i>Corvus corax clarionensis</i>	Common Raven	3.079	8	OmTe	
<i>Bubo virginianus</i>	Great Horned Owl	3.117	8	CaAe	
<i>Cathartes aura</i>	Turkey Vulture	3.166	8	CaAe	
<i>Aquila chrysaetos</i>	Golden Eagle	3.623	9	CaAe	
<i>Gymnogyps californicus</i>	California Condor	4.004	9	CaAe	x
Non-Indigenous Species					
<i>Lonchura punctulata</i>	Nutmeg Mannikin	1.146		GrTe	
<i>Passer domesticus</i>	House Sparrow	1.442		GrTe	
<i>Cardinalis cardinalis</i>	Northern Cardinal	1.650		GrTe	
<i>Sturnus vulgaris</i>	European Starling	1.915		InTe	
<i>Streptopelia decaocto</i>	Eurasian Collared-Dove	2.173		GrTe	
<i>Aratinga erythrogenys</i>	Red-masked Conure	2.243		HeFo	
<i>Aratinga acuticaudata</i>	Blue-crowned Parakeet	2.279		HeFo	
<i>Calocitta coliei</i>	Black-throated Magpie-Jay	2.377		OmFo	

APPENDIX F. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Amazona viridigenalis</i>	Red-crowned Parrot	2.468		GrFo	
<i>Bubulcus ibis</i>	Cattle Egret	2.529		InTe	
<i>Columba livia</i>	Rock Dove	2.550		GrTe	
<i>Phasianus colchicus</i>	Ring-necked Pheasant	3.055		HeTe	
<i>Aix galericulata</i>	Mandarin Duck	3.233		OmAq	
<i>Meleagris gallopavo</i>	Wild Turkey	3.764		OmTe	

APPENDIX G. Mediterranean-climate ecosystem birds in Chile. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass (g)	BCART	Functional Group	Endangered
<i>Sephanoides sephanoides</i>	Green-backed Firecrown	0.716	1	NeAe	
<i>Anairetes parulus</i>	Tufted Tit-Tyrant	0.771	1	InFo	
<i>Tachuris rubrigastra</i>	Many-colored Rush-Tyrant	0.857	1	InFo	
<i>Pseudocolopteryx flaviventris</i>	Warbling Doradito	0.875	1	InFo	
<i>Cistothorus platensis</i>	Sedge Wren	0.954	1	InFo	
<i>Leptasthenura aegithaloides</i>	Plain-mantled Tit-Spinetail	0.954	1	InFo	
<i>Pygochelidon cyanoleuca</i>	Blue-and-white Swallow	1.000	1	InAe	
<i>Troglodytes aedon</i>	House Wren	1.017	1	InFo	
<i>Sylviorthorhynchus desmursii</i>	Des Murs's Wiretail	1.037	1	InTe	
<i>Scytalopus magellanicus</i>	Magellanic Tapaculo	1.085	1	InTe	
<i>Aphrastura spinicauda</i>	Thorn-tailed Rayadito	1.086	1	InBa	
<i>Lessonia rufa</i>	Austral Negrito	1.130	2	InTe	
<i>Phleocryptes melanops</i>	Wren-like Rushbird	1.152	2	InFo	
<i>Elaenia albiceps</i>	White-Crested Elaenia	1.201	2	InAe	
<i>Sicalis luteola</i>	Puna Yellow-Finch	1.204	2	GrTe	
<i>Carduelis barbata</i>	Black-chinned Siskin	1.220	2	GrTe	
<i>Phrygilus alaudinus</i>	Band-tailed Sierra-Finch	1.255	2	GrTe	
<i>Tachycineta meyeni</i>	Chilean Swallow	1.265	2	InAe	
<i>Anthus correndera</i>	Correndera Pipit	1.299	2	InTe	
<i>Asthenes pyrrholeuca</i>	Lesser Canastero	1.299	2	InFo	
<i>Patagona gigas</i>	Giant Hummingbird	1.305	2	NeAe	
<i>Phrygilus gayi</i>	Gray-hooded Sierra-Finch	1.312	2	GrTe	
<i>Zonotrichia capensis</i>	Rufous-collared Sparrow	1.312	2	GrTe	

APPENDIX G. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Asthenes humicola</i>	Dusky-tailed Canastero	1.313	2	InFo	
<i>Pygarrhichas albogularis</i>	White-throated Treerunner	1.325	2	InBa	
<i>Hymenops perspicillata</i>	Spectacled Tyrant	1.380	2	InAe	
<i>Agelasticus thilius</i>	Yellow-winged Blackbird	1.477	3	InFo	
<i>Geositta cunicularia</i>	Common Miner	1.477	3	InTe	
<i>Diuca diuca</i>	Common Diuca-Finch	1.491	3	GrTe	
<i>Cinclodes fuscus</i>	Bar-winged Cinclodes	1.502	3	InTe	
<i>Laterallus jamaicensis</i>	Black Rail	1.530	3	InAq	X
<i>Thraupis bonariensis</i>	Blue-and-yellow Tanager	1.556	3	HeAe	
<i>Phrygilus fruticeti</i>	Mourning Sierra-Finch	1.589	3	GrTe	
<i>Picoides lignarius</i>	Striped Woodpecker	1.592	3	InBa	
<i>Chilia melanura</i>	Crag Chilia	1.602	3	InTe	
<i>Phytotoma rara</i>	Rufous-tailed Plantcutter	1.602	3	HeFo	
<i>Charadrius alexandrinus</i>	Snowy Plover	1.617	3	InTe	
<i>Caprimulgus longirostris</i>	Band-winged Nightjar	1.635	3	InAe	
<i>Eugralla paradoxa</i>	Ochre-flanked Tapaculo	1.643	3	InTe	
<i>Thinocorus rumicivorus</i>	Least Seedsnipe	1.643	3	GrTe	
<i>Pyrope pyrope</i>	Fire-eyed Diucon	1.685	3	InAe	
<i>Upucerthia dumetaria</i>	Scale-throated Earthcreeper	1.693	3	InTe	
<i>Columbina picui</i>	Picui Ground-Dove	1.699	3	GrTe	
<i>Scelorchilus albicollis</i>	White-throated Tapaculo	1.705	3	InTe	
<i>Cinclodes patagonicus</i>	Dark-bellied Cinclodes	1.713	3	InTe	
<i>Charadrius falklandicus</i>	Two-banded Plover	1.813	4	InTe	
<i>Mimus thenca</i>	Chilean Mockingbird	1.820	4	InAe	
<i>Cinclodes nigrofumosus</i>	Chilean Seaside Cinclodes	1.824	4	InTe	

APPENDIX G. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Nycticryphes semicollaris</i>	South American Painted Snipe	1.839	4	InTe	
<i>Glaucidium nanum</i>	Austral Pygmy-Owl	1.858	4	CaAe	
<i>Scelorchilus rubecula</i>	Chucazo Tapaculo	1.881	4	InTe	
<i>Ixobrychus involucris</i>	Stripe-backed Bittern	1.948	4	PiAq	
<i>Curaeus curaeus</i>	Austral Blackbird	1.954	4	InTe	
<i>Turdus falcklandii</i>	Austral Thrush	1.975	4	InTe	
<i>Agriornis livida</i>	Great Shrike-Tyrant	1.997	4	InTe	
<i>Colaptes pitius</i>	Chilean Flicker	2.000	4	InTe	
<i>Sturnella loyca</i>	Long-tailed Meadowlark	2.053	5	InTe	
<i>Zenaida auriculata</i>	Eared Dove	2.057	5	GrTe	
<i>Falco sparverius</i>	American Kestrel	2.063	5	CaAe	
<i>Pteroptochos megapodius</i>	Moustached Turca	2.076	5	InTe	
<i>Gallinago gallinago</i>	Common Snipe	2.086	5	InAq	
<i>Metriopelia melanoptera</i>	Black-winged Ground Dove	2.097	5	GrTe	
<i>Oreopholus ruficollis</i>	Tawny-throated Dotterel	2.124	5	InAq	
<i>Porphyriops melanops</i>	Spot-flanked Gallinule	2.130	5	HeAq	
<i>Pteroptochos castaneus</i>	Chestnut-throated Huet-Huet	2.149	5	InTe	
<i>Sterna trudeaui</i>	Snowy-crowned Tern	2.185	5	PiAe	
<i>Athene cunicularia</i>	Burrowing Owl	2.189	5	CaAe	
<i>Himantopus mexicanus</i>	Black-necked Stilt	2.220	5	InAq	
<i>Sterna hirundinacea</i>	South American Tern	2.265	5	PiAe	
<i>Pardirallus sanguinolentus</i>	Plumbeous Rail	2.294	5	InAq	
<i>Columba araucana</i>	Chilean Pigeon	2.301	5	HeTe	
<i>Enicognathus ferrugineus</i>	Austral Parakeet	2.301	5	HeFo	
<i>Rollandia rolland</i>	White-tufted Grebe	2.395	6	InAq	

APPENDIX G. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Cyanoliseus patagonus</i>	Burrowing Parrot	2.436	6	HeFo	
<i>Enicognathus leptorhynchus</i>	Slender-billed Parakeet	2.458	6	GrFo	
<i>Milvago chimango</i>	Chimango Caracara	2.471	6	CaAe	
<i>Larus maculipennis</i>	Brown-hooded Gull	2.505	6	InTe	
<i>Vanellus chilensis</i>	Southern Lapwing	2.515	6	InTe	
<i>Merganetta armata</i>	Torrent Duck	2.519	6	InAq	
<i>Elanus leucurus</i>	White-tailed Kite	2.522	6	CaAe	
<i>Falco femoralis</i>	Aplomado Falcon	2.523	6	InAe	
<i>Podiceps occipitalis</i>	Silvery Grebe	2.524	6	HeAq	
<i>Accipiter bicolor</i>	Bicolored Hawk	2.532	6	CaAe	
<i>Asio flammeus</i>	Short-eared Owl	2.540	6	CaAe	
<i>Egretta thula</i>	Snowy Egret	2.569	6	PiTe	
<i>Strix rufipes</i>	Rufous-legged Owl	2.582	6	CaAe	
<i>Anas cyanoptera</i>	Cinnamon Teal	2.586	6	HeAq	
<i>Anas flavirostris</i>	Speckled Teal	2.597	6	HeAq	
<i>Circus cinereus</i>	Cinereus Harrier	2.623	6	CaAe	
<i>Podilymbus podiceps</i>	Pied-billed Grebe	2.645	7	PiAq	
<i>Fulica leucoptera</i>	White-winged Coot	2.653	7	OmAq	
<i>Nothoprocta perdicaria</i>	Chilean Tinamou	2.661	7	GrTe	
<i>Anas platalea</i>	Red Shoveler	2.719	7	HeAq	
<i>Tyto alba</i>	Barn Owl	2.719	7	CaAe	
<i>Heteronetta atricapilla</i>	Black-headed Duck	2.726	7	GrAq	
<i>Oxyura jamaicensis</i>	Ruddy Duck	2.736	7	GrAq	
<i>Oxyura vittata</i>	Lake Duck	2.748	7	GrAq	
<i>Anas georgica</i>	Yellow-billed Pintail	2.766	7	InAq	

APPENDIX G. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Fulica rufifrons</i>	Red-fronted Coot	2.791	7	OmAq	
<i>Plegadis chihi</i>	White-faced Ibis	2.793	7	CaAq	
<i>Haematopus palliatus</i>	American Oystercatcher	2.801	7	InAq	
<i>Fulica armillata</i>	Red-gartered Coot	2.824	7	HeAq	
<i>Falco peregrinus</i>	Peregrine Falcon	2.893	8	CaAe	
<i>Phalcoboenus megalopterus</i>	Mountain Caracara	2.900	8	CaAe	
<i>Anas sibilatrix</i>	Chiloe Wigeon	2.918	8	HeAq	
<i>Parabuteo unicinctus</i>	Harris's Hawk	2.926	8	CaAe	
<i>Buteo polyosoma</i>	Red-backed Hawk	2.936	8	CaAe	
<i>Casmerodius albus</i>	Common Egret	2.941	8	PiTe	
<i>Nycticorax nycticorax</i>	Black-crowned Night-Heron	2.946	8	PiAq	
<i>Polyborus plancus</i>	Crested Caracara	2.951	8	CaAe	
<i>Larus dominicanus</i>	Kelp Gull	2.954	8	CaAq	
<i>Anas specularis</i>	Spectacled Duck	2.989	8	OmAq	X
<i>Netta peposaca</i>	Rosy-billed Pochard	3.000	8	HeAq	
<i>Phalacrocorax brasilianus</i>	Neotropic Cormorant	3.041	8	PiAq	
<i>Podiceps major</i>	Great Grebe	3.067	8	PiAq	
<i>Bubo virginianus</i>	Great Horned Owl	3.117	8	CaAe	
<i>Cathartes aura</i>	Turkey Vulture	3.166	9	CaAe	
<i>Theristicus caudatus</i>	Buff-necked Ibis	3.190	9	InTe	
<i>Coragyps atratus</i>	Black Vulture	3.318	9	CaAe	
<i>Geranoaetus melanoleucus</i>	Black-chested Buzzard-Eagle	3.353	9	CaAe	
<i>Cygnus melancoryphus</i>	Black-necked Swan	3.672	9	HeAq	
Non-Indigenous Species					
<i>Passer domesticus</i>	House Sparrow	1.442		GrTe	

APPENDIX G. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Molothrus bonariensis</i>	Shiny Cowbird	1.548		InTe	
<i>Myopsitta monachus</i>	Monk Parakeet	2.079		HeFo	
<i>Callipepla californica</i>	California Quail	2.238		GrTe	
<i>Bubulcus ibis</i>	Cattle Egret	2.529		InTe	
<i>Columba livia</i>	Rock Dove	2.550		GrTe	
<i>Phasianus colchicus</i>	Ring-necked Pheasant	3.055		HeTe	
<i>Cairina moschata</i>	Muscovy Duck	3.392		HeAq	

APPENDIX H. Mediterranean-climate ecosystem birds in Spain. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass (g)	BCART	Functional Group	Endangered
<i>Aegithalos caudatus</i>	Long-tailed Tit	0.895	1	InFo	
<i>Phylloscopus bonelli</i>	Western Bonelli's Warbler	0.913	1	InFo	
<i>Certhia brachydactyla</i>	Short-toed Treecreeper	0.914	1	InBa	
<i>Cisticola juncidis</i>	Zitting Cisticola	0.940	1	InTe	
<i>Sylvia cantillans</i>	Subalpine Warbler	0.964	1	InFo	
<i>Troglodytes troglodytes</i>	Winter Wren	0.973	1	InFo	
<i>Sylvia undata</i>	Dartford Warbler	0.973	1	InTe	
<i>Parus ater</i>	Coal Tit	0.987	1	InFo	
<i>Sylvia conspicillata</i>	Spectacled Warbler	1.004	1	InFo	
<i>Parus caeruleus</i>	Blue Tit	1.029	1	InFo	
<i>Hippolais pallida</i>	Eastern Olivaceous Warbler	1.039	1	InFo	X
<i>Hippolais polyglotta</i>	Melodious Warbler	1.041	1	InFo	
<i>Serinus serinus</i>	European Serin	1.077	1	GrTe	
<i>Parus cristatus</i>	Crested Tit	1.099	1	InFo	
<i>Riparia riparia</i>	Bank Swallow	1.119	1	InAe	
<i>Cettia cetti</i>	Cetti's Warbler	1.125	1	InTe	
<i>Sylvia melanocephala</i>	Sardinian Warbler	1.129	1	InTe	
<i>Delichon urbica</i>	Common House-Martin	1.161	1	InAe	
<i>Saxicola torquata</i>	Stonechat	1.185	1	InAe	
<i>Muscicapa striata</i>	Spotted Flycatcher	1.197	1	InAe	
<i>Hirundo rustica</i>	Barn Swallow	1.204	1	InAe	
<i>Carduelis carduelis</i>	European Goldfinch	1.210	1	GrTe	
<i>Phoenicurus ochruros</i>	Black Redstart	1.211	1	InTe	

APPENDIX H. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Oenanthe hispanica</i>	Black-eared Wheatear	1.217	1	InTe	X
<i>Erithacus rubecula</i>	European Robin	1.223	1	InTe	
<i>Motacilla cinerea</i>	Grey Wagtail	1.261	1	InTe	
<i>Motacilla flava</i>	Yellow Wagtail	1.268	1	InTe	
<i>Parus major</i>	Great Tit	1.272	1	InFo	
<i>Carduelis cannabina</i>	Eurasian Linnet	1.290	1	GrTe	
<i>Sylvia atricapilla</i>	Blackcap	1.291	1	InFo	
<i>Luscinia megarhynchos</i>	Common Nightingale	1.312	1	InTe	
<i>Motacilla alba</i>	White Wagtail	1.322	1	InTe	
<i>Sylvia hortensis</i>	Western Orphean Warbler	1.324	1	InFo	
<i>Fringilla coelebs</i>	Chaffinch	1.331	1	GrTe	
<i>Calandrella brachydactyla</i>	Greater Short-toed Lark	1.347	1	InTe	X
<i>Hirundo daurica</i>	Red-rumped Swallow	1.347	1	InAe	
<i>Ptyonprogne rupestris</i>	Eurasian Crag-Martin	1.364	1	InAe	
<i>Emberiza cia</i>	Rock Bunting	1.366	1	GrTe	
<i>Calandrella rufescens</i>	Lesser Short-toed Lark	1.377	1	InTe	X
<i>Cercotrichas galactotes</i>	Rufous-tailed Scrub-Robin	1.387	1	InTe	X
<i>Oenanthe oenanthe</i>	Northern Wheatear	1.389	1	InTe	
<i>Emberiza cirrus</i>	Cirl Bunting	1.408	1	GrTe	
<i>Lullula arborea</i>	Wood Lark	1.417	1	InTe	
<i>Carduelis chloris</i>	European Greenfinch	1.418	1	GrTe	
<i>Passer domesticus</i>	House Sparrow	1.442	1	GrTe	
<i>Anthus campestris</i>	Tawny Pipit	1.459	1	InTe	
<i>Acrocephalus arundinaceus</i>	Great Reed-Warbler	1.479	1	InFo	
<i>Lanius senator</i>	Woodchat Shrike	1.512	1	InAe	X

APPENDIX H. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Alcedo atthis</i>	Common Kingfisher	1.550	1	CaAe	X
<i>Galerida theklae</i>	Thekla Lark	1.566	1	InTe	
<i>Alauda arvensis</i>	Eurasian Skylark	1.585	1	InTe	
<i>Oenanthe leucura</i>	Black Wheatear	1.600	1	InTe	X
<i>Apus apus</i>	Common Swift	1.630	1	InAe	
<i>Galerida cristata</i>	Crested Lark	1.650	1	GrTe	
<i>Miliaria calandra</i>	Corn Bunting	1.694	1	GrTe	
<i>Merops apiaster</i>	European Bee-eater	1.741	2	InAe	
<i>Monticola solitarius</i>	Blue Rock Thrush	1.756	2	InAe	
<i>Coccothraustes coccothraustes</i>	Hawfinch	1.763	2	GrFo	
<i>Cinclus cinclus</i>	White-throated Dipper	1.778	2	InAq	
<i>Melanocorypha calandra</i>	Calandra Lark	1.783	2	InTe	
<i>Upupa epops</i>	Hoopoe	1.788	2	InTe	
<i>Larius excubitor</i>	Northern Shrike	1.802	2	InAe	
<i>Turnix sylvatica</i>	Small Buttonquail	1.813	2	GrTe	X
<i>Caprimulgus ruficollis</i>	Red-necked Nightjar	1.836	2	InAe	
<i>Oriolus oriolus</i>	Eurasian Golden Oriole	1.847	2	InFo	
<i>Glareola pratincola</i>	Collared Pratincole	1.904	3	InAe	X
<i>Dendrocopos major</i>	Great Spotted Woodpecker	1.906	3	InBa	
<i>Caprimulgus europaeus</i>	Eurasian Nightjar	1.929	3	InAe	
<i>Otus scops</i>	European Scops-Owl	1.930	3	InAe	
<i>Sturnus unicolor</i>	Spotless Starling	1.938	3	InTe	
<i>Turdus merula</i>	Eurasian Blackbird	1.967	3	InTe	
<i>Coturnix coturnix</i>	Common Quail	2.007	3	GrTe	
<i>Tachymarptis melba</i>	Alpine Swift	2.017	3	InAe	

APPENDIX H. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Rallus aquaticus</i>	Water Rail	2.063	3	InAq	
<i>Cuculus canorus</i>	Common Cuckoo	2.065	3	InFo	
<i>Turdus viscivorus</i>	Mistle Thrush	2.071	3	InTe	
<i>Tachybaptus ruficollis</i>	Little Grebe	2.130	4	InAq	
<i>Streptopelia turtur</i>	Eurasian Turtle-Dove	2.135	4	GrTe	X
<i>Coracias garrulus</i>	European Roller	2.167	4	InAe	X
<i>Ixobrychus minutus</i>	Little Bittern	2.169	4	CaAq	
<i>Falco naumanni</i>	Lesser Kestrel	2.182	4	InAe	X
<i>Clamator glandarius</i>	Great Spotted Cuckoo	2.186	4	InFo	
<i>Athene noctua</i>	Little Owl	2.196	4	CaAe	
<i>Garrulus glandarius</i>	Eurasian Jay	2.214	4	InFo	
<i>Picus viridis</i>	Green Woodpecker	2.244	4	InTe	
<i>Falco tinnunculus</i>	Eurasian Kestrel	2.304	4	CaAe	
<i>Accipiter nisus</i>	Eurasian Sparrowhawk	2.310	4	CaAe	
<i>Falco subbuteo</i>	Eurasian Hobby	2.324	4	CaAe	X
<i>Pterocles alchata</i>	Pin-tailed Sandgrouse	2.376	4	GrTe	X
<i>Corvus monedula</i>	Eurasian Jackdaw	2.376	4	InTe	
<i>Asio Otus</i>	Long-eared Owl	2.423	4	CaAe	
<i>Gallinula chloropus</i>	Common Moorhen	2.477	4	HeAq	
<i>Circus pygargus</i>	Montagu's Harrier	2.499	4	CaAe	X
<i>Columba livia</i>	Rock Pigeon	2.550	4	GrTe	
<i>Pterocles orientalis</i>	Black-bellied Sandgrouse	2.608	4	GrTe	X
<i>Strix aluco</i>	Tawny Owl	2.663	4	CaAe	
<i>Burhinus oedicephalus</i>	Eurasian Thick-knee	2.665	4	InAq	X
<i>Alectoris rufa</i>	Red-legged Partridge	2.679	4	GrTe	

APPENDIX H. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Columba palumbus</i>	Common Wood-Pigeon	2.689	4	HeTe	
<i>Corvus corone</i>	Carrion Crow	2.691	4	InTe	
<i>Tyto alba</i>	Barn Owl	2.719	4	CaAe	
<i>Podiceps cristatus</i>	Great Crested Grebe	2.889	5	CaAq	
<i>Falco peregrinus</i>	Peregrine Falcon	2.893	5	CaAe	
<i>Milvus migrans</i>	Black Kite	2.918	5	CaAe	X
<i>Hieraaetus pennatus</i>	Booted Eagle	2.925	5	CaAe	
<i>Ardea purpurea</i>	Purple Heron	2.941	5	CaAq	
<i>Buteo buteo</i>	Eurasian Buzzard	2.942	5	CaAe	
<i>Accipiter gentilis</i>	Northern Goshawk	2.967	5	CaAe	
<i>Milvus milvus</i>	Red Kite	3.020	5	CaAe	X
<i>Corvus corax</i>	Common Raven	3.054	5	CaTe	
<i>Circaetus gallicus</i>	Short-toed Eagle	3.230	6	CaAe	
<i>Hieraaetus fasciatus</i>	Bonelli's Eagle	3.312	6	CaAe	X
<i>Neophron percnopterus</i>	Egyptian Vulture	3.320	6	CaAe	X
<i>Bubo bubo</i>	Eurasian Eagle-Owl	3.347	6	CaAe	
<i>Aquila heliaca</i>	Imperial Eagle	3.514	6	CaAe	X
<i>Ciconia ciconia</i>	White Stork	3.538	6	CaTe	
<i>Aquila chrysaetos</i>	Golden Eagle	3.623	6	CaAe	X
<i>Otis tarda</i>	Great Bustard	3.862	6	InTe	X
<i>Gyps fulvus</i>	Eurasian Griffon	3.870	6	CaAe	
Non-Indigenous Species					
<i>Estrilda astrild</i>	Common Waxbill	0.875		GrFo	
<i>Amandava amandava</i>	Red Avadavat	0.996		GrTe	
<i>Passer montanus</i>	Tree Sparrow	1.342		GrTe	

APPENDIX H. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Psittacula krameri</i>	Ring-necked Parakeet	2.155		HeFo	
<i>Streptopelia decaocto</i>	Eurasian Collared-Dove	2.173		GrTe	
<i>Callipepla californica</i>	California Quail	2.238		GrTe	
<i>Oxyura jamaicensis</i>	Ruddy Duck	2.736		GrAq	
<i>Alectoris chukar</i>	Chukar	2.762		GrTe	
<i>Anas platyrhynchos</i>	Mallard	3.032		OmAq	
<i>Phasianus colchicus</i>	Pheasant	3.051		HeTe	

APPENDIX I. Mediterranean-climate ecosystem birds in South Africa. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass (g)	BCART	Functional Group	Endangered
<i>Estrilda astrild</i>	Common Waxbill	0.875	1	GrFo	
<i>Anthoscopus minutus</i>	Cape Penduline-Tit	0.875	1	InFo	
<i>Cinnyris chalybeus</i>	Lesser Double-collared Sunbird	0.937	1	NeAe	
<i>Cisticola fulvicapilla</i>	Neddicky	0.954	1	InTe	
<i>Anthobaphes violacea</i>	Orange-breasted Sunbird	0.964	1	NeFo	
<i>Cryptillas victorini</i>	Victorin's Warbler	1.000	1	InTe	
<i>Prinia maculosa</i>	Karoo Prinia	1.000	1	InFo	
<i>Cisticola subruficapilla</i>	Grey-backed Cisticola	1.021	1	InFo	
<i>Cisticola textrix</i>	Cloud Cisticola	1.021	1	InTe	
<i>Sylvietta rufescens</i>	Long-billed Crombec	1.053	1	InFo	
<i>Apalis thoracica</i>	Bar-throated Apalis	1.083	1	InFo	
<i>Batis capensis</i>	Cape Batis	1.107	1	InFo	
<i>Cisticola tinniens</i>	Levaillant's Cisticola	1.111	1	InTe	
<i>Crithagra totta</i>	Cape Siskin	1.117	1	GrTe	
<i>Riparia paludicola</i>	Brown-throated Martin	1.127	1	InAe	
<i>Zosterops pallidus</i>	Orange River White-eye	1.127	1	NeFo	
<i>Serinus canicollis</i>	Cape Canary	1.140	1	GrTe	
<i>Saxicola torquatus</i>	African Stonechat	1.185	2	InAe	
<i>Parisoma subcaeruleum</i>	Chestnut-vented Tit-Babbler	1.193	2	InFo	
<i>Hirundo rustica</i>	Barn Swallow	1.204	2	InAe	
<i>Euplectes orix</i>	Southern Red Bishop	1.211	2	GrTe	
<i>Crithagra flaviventris</i>	Yellow Canary	1.212	2	GrTe	
<i>Nectarinia famosa</i>	Malachite Sunbird	1.233	2	NeFo	

APPENDIX I. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Euplectes capensis</i>	Yellow Bishop	1.260	2	GrTe	
<i>Cercomela sinuata</i>	Sickle-winged Chat	1.270	2	InAe	
<i>Hirundo fuligula</i>	Rock Martin	1.279	2	InAe	
<i>Crithagra sulphuratus</i>	Brimstone Canary	1.283	2	GrTe	
<i>Parus afer</i>	Grey Tit	1.297	2	InBa	
<i>Cercotrichas coryphoeus</i>	Karoo Scrub-Robin	1.301	2	InTe	
<i>Calandrella cinerea</i>	Red-capped Lark	1.316	2	InTe	
<i>Motacilla capensis</i>	Cape Wagtail	1.318	2	InTe	
<i>Hirundo albicularis</i>	White-throated Swallow	1.328	2	InAe	
<i>Passer melanurus</i>	Cape Sparrow	1.340	2	GrTe	
<i>Cercomela familiaris</i>	Familiar Chat	1.342	2	InTe	
<i>Crithagra leucopterus</i>	Protea Seed eater	1.346	2	GrFo	
<i>Emberiza capensis</i>	Cape Bunting	1.350	2	GrTe	
<i>Sigelus silens</i>	Fiscal Flycatcher	1.408	3	InAe	
<i>Crithagra albogularis</i>	White-throated Canary	1.413	3	GrTe	
<i>Anthus leucophrys</i>	Plain-backed Pipit	1.431	3	InTe	
<i>Hirundo cucullata</i>	Greater Striped Swallow	1.431	3	InAe	
<i>Cossypha caffra</i>	Cape Robin-Chat	1.455	3	InTe	
<i>Mirafra apiata</i>	Cape Clapper Lark	1.487	3	InTe	
<i>Calendulauda albescens</i>	Karoo Lark	1.487	3	InTe	
<i>Sphenoeacus afer</i>	Cape Grassbird	1.497	3	InTe	
<i>Tricholaema leucomelas</i>	Acacia Pied Barbet	1.508	3	HeFo	
<i>Certhilauda curvirostris</i>	Cape Long-billed Lark	1.582	4	InTe	
<i>Pycnonotus capensis</i>	Cape Bulbul	1.597	4	HeFo	
<i>Promerops cafer</i>	Cape Sugarbird	1.606	4	NeFo	

APPENDIX I. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Oena capensis</i>	Namaqua Dove	1.608	4	GrTe	
<i>Colius colius</i>	White-backed Mousebird	1.617	4	HeFo	
<i>Lanius collaris</i>	Common Fiscal	1.618	4	InTe	
<i>Ploceus capensis</i>	Cape Weaver	1.627	4	InTe	
<i>Apus barbatus</i>	African Black Swift	1.631	4	InAe	
<i>Galerida magnirostris</i>	Large-billed Lark	1.643	4	GrTe	
<i>Caprimulgus pectoralis</i>	Fiery-necked Nightjar	1.674	4	InAe	
<i>Macronyx capensis</i>	Cape Longclaw	1.677	4	InTe	
<i>Laniarius ferrugineus</i>	Southern Boubou	1.688	4	InTe	
<i>Colius striatus</i>	Speckled Mousebird	1.708	4	HeFo	
<i>Urocolius indicus</i>	Red-faced Mousebird	1.751	4	HeFo	
<i>Chaetops frenatus</i>	Cape Rockjumper	1.756	4	InTe	
<i>Monticola rupestris</i>	Cape Rock-Thrush	1.778	4	InTe	
<i>Upupa africana</i>	African Hoopoe	1.788	4	InTe	
<i>Telophorus zeylonus</i>	Bokmakierie	1.797	4	InTe	
<i>Creatophora cinerea</i>	Wattled Starling	1.826	4	InTe	
<i>Turdus olivaceus</i>	Olive Thrush	1.868	4	InTe	
<i>Tachymarptis melba</i>	Alpine Swift	1.881	4	InAe	
<i>Streptopelia senegalensis</i>	Laughing Dove	2.004	5	GrTe	
<i>Spreo bicolor</i>	Pied Starling	2.021	5	InTe	
<i>Geocolaptes olivaceus</i>	Ground Woodpecker	2.079	5	InTe	
<i>Onychognathus morio</i>	Red-winged Starling	2.124	5	InTe	
<i>Streptopelia capicola</i>	Cape Turtle-Dove	2.152	5	GrTe	
<i>Vanellus coronatus</i>	Crowned Lapwing	2.223	5	InTe	
<i>Falco rupicolus</i>	Rock Kestrel	2.304	5	CaAe	

APPENDIX I. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Elanus caeruleus</i>	Black-shouldered Kite	2.522	6	CaAe	
<i>Bubulcus ibis</i>	Cattle Egret	2.529	6	InTe	
<i>Columba guinea</i>	Speckled Pigeon	2.547	6	GrTe	
<i>Scleroptila africanus</i>	Grey-winged Francolin	2.592	6	HeTe	
<i>Burhinus capensis</i>	Spotted Thick-knee	2.626	6	InTe	
<i>Circus ranivorus</i>	African Marsh-Harrier	2.705	6	CaAe	x
<i>Corvus albus</i>	Pied Crow	2.723	6	HeTe	
<i>Pternistis capensis</i>	Cape Spurfowl	2.814	6	HeTe	
<i>Afrotis afra</i>	Southern Black Korhaan	2.840	6	InTe	
<i>Corvus capensis</i>	Cape Crow	2.843	6	InTe	
<i>Buteo vulpinus</i>	Steppe Buzzard	2.942	6	CaAe	
<i>Corvus albicollis</i>	White-necked Raven	2.954	6	InTe	
<i>Ardea melanocephala</i>	Black-headed Heron	3.025	6	InTe	
<i>Buteo rufofuscus</i>	Jackal Buzzard	3.066	6	CaAe	
<i>Numida meleagris</i>	Helmeted Guinea-fowl	3.114	6	GrTe	
<i>Sagittarius serpentarius</i>	Secretarybird	3.557	7	InTe	x
<i>Aquila verreauxi</i>	Verreaux's Eagle	3.613	7	CaAe	
<i>Neotis denhami</i>	Denham's Bustard	3.615	7	InTe	x
Non-Indigenous Species					
<i>Delichon urbicum</i>	Common House Martin	1.161		InAe	
<i>Fringilla coelebs</i>	Chaffinch	1.330		GrTe	
<i>Passer domesticus</i>	House Sparrow	1.442		GrTe	
<i>Apus horus</i>	Horus Swift	1.447		InAe	
<i>Sturnus vulgaris</i>	European Starling	1.915		InTe	
<i>Cuculus clamosus</i>	Black Cuckoo	1.929		InFo	

APPENDIX I. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Streptopelia decaocto</i>	Eurasian Collared-Dove	2.173		GrTe	
<i>Corvus splendens</i>	House Crow	2.477		OmTe	
<i>Columba livia</i>	Rock Dove	2.550		GrTe	
<i>Anas platyrhynchos</i>	Mallard	3.032		OmAq	

APPENDIX J. Mediterranean-climate ecosystem birds in southwestern Australia. Log 10 body mass in grams (g), Bayesian CART (BCART) group, functional group, and the species status.

Latin Name	Common Name	Mass (g)	BCART	Functional Group	Endangered
<i>Smicrornis brevirostris</i>	Weebill	0.708	1	GrFo	
<i>Gerygone fusca</i>	Western Gerygone	0.783	1	InFo	
<i>Malurus leucopterus</i>	White-winged Fairywren	0.785	1	InFo	
<i>Acanthiza inornata</i>	Western Thornbill	0.845	2	InTe	
<i>Poephila guttata</i>	Zebra Finch	0.845	2	GrTe	
<i>Acanthiza uropygialis</i>	Chestnut-rumped Thornbill	0.874	2	InFo	
<i>Stipiturus malachurus</i>	Southern Emuwren	0.879	2	InTe	x
<i>Acanthiza apicalis</i>	Inland Thornbill	0.881	2	InFo	
<i>Certhionyx niger</i>	Black Honeyeater	0.892	2	NeFo	
<i>Dicaeum hirundinaceum</i>	Mistletoebird	0.903	2	HeTe	
<i>Malurus lamberti</i>	Variegated Fairywren	0.903	2	InFo	x
<i>Petroica goodenovii</i>	Red-capped Robin	0.903	2	InAe	
<i>Rhipidura fuliginosa</i>	Grey Fantail	0.903	2	InAe	
<i>Acanthiza chrysorrhoa</i>	Yellow-rumped Thornbill	0.944	2	InFo	
<i>Pardalotus xanthopygus</i>	Yellow-rumped Pardalope	0.944	2	InFo	
<i>Pardalotus punctatus</i>	Spotted Pardalote	0.964	2	InFo	
<i>Malurus pulcherrimus</i>	Blue-breasted Fairywren	0.978	2	InFo	
<i>Petroica multicolor</i>	Scarlet Robin	0.982	2	InTe	
<i>Malurus elegans</i>	Red-winged Fairywren	1.000	2	InFo	
<i>Malurus splendens</i>	Splendid Fairywren	1.000	2	InFo	
<i>Ephthianura tricolor</i>	Crimson Chat	1.024	2	InTe	
<i>Acanthorhynchus superciliosus</i>	Western Spinebill	1.033	2	NeFo	
<i>Sericornis brunneus</i>	Redthroat	1.052	2	InTe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Cecropis ariel</i>	Fairy Martin	1.053	2	InAe	
<i>Emblema oculata</i>	Red-eared Firetail	1.070	2	GrFo	
<i>Daphoenositta chrysoptera</i>	Varied Sittella	1.076	2	InBa	
<i>Ephthianura albifrons</i>	White-fronted Chat	1.079	2	InTe	
<i>Pardalotus striatus</i>	Striated Pardalote	1.086	2	InFo	
<i>Aphelocephala leucopsis</i>	Southern Whiteface	1.101	2	InTe	
<i>Sericornis frontalis</i>	White-browed Scrubwren	1.107	2	InTe	
<i>Lichmera indistincta</i>	Brown Honeyeater	1.114	2	NeFo	
<i>Sericornis cautus</i>	Shy Hylacola	1.153	3	InTe	
<i>Melithreptus brevirostris</i>	Brown-headed Honeyeater	1.164	3	NeFo	
<i>Hirundo neoxena</i>	Welcome Swallow	1.167	3	InAe	
<i>Melithreptus lunatus</i>	White-naped Honeyeater	1.167	3	NeFo	
<i>Cheramoeca leucosternum</i>	White-backed Swallow	1.170	3	InAe	
<i>Cecropis nigricans</i>	Tree Martin	1.175	3	InAe	
<i>Microeca fascinans</i>	Jacky Winter	1.196	3	InAe	
<i>Lichenostomus ornatus</i>	Yellow-plumed Honeyeater	1.250	4	NeFo	
<i>Phylidonyris albifrons</i>	White-fronted Honeyeater	1.255	4	NeFo	
<i>Pachycephala rufiventris</i>	Rufous Whistler	1.258	4	InTe	
<i>Phylidonyris nigra</i>	White-cheeked Honeyeater	1.262	4	NeFo	
<i>Phylidonyris melanops</i>	Tawny-crowned Honeyeater	1.267	4	NeFo	
<i>Lichenostomus cratitius</i>	Purple-gaped Honeyeater	1.292	4	InFo	
<i>Lichenostomus penicillatus</i>	White-plumed Honeyeater	1.297	4	HeFo	
<i>Phylidonyris novaehollandiae</i>	New Holland Honeyeater	1.301	4	NeFo	
<i>Sericornis fuliginosus</i>	Field Wren	1.319	4	InFo	
<i>Melanodryas cucullata</i>	Hooded Robin	1.326	4	InAe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Eopsaltria griseogularis</i>	Gray-breasted Robin	1.336	4	InTe	
<i>Amytornis textilis</i>	Thick-billed Grasswren	1.356	4	GrTe	x
<i>Chrysococcyx basalus</i>	Horsfield's Bronze Cuckoo	1.358	4	InFo	
<i>Lichenostomus leucotis</i>	White-eared Honeyeater	1.364	4	HeFo	
<i>Myiagra inquieta</i>	Restless Flycatcher	1.380	4	InAe	
<i>Anthus novaeseelandiae</i>	Australasian Pipit	1.384	4	InTe	
<i>Lichenostomus virescens</i>	Singing Honeyeater	1.391	4	NeFo	
<i>Chrysococcyx lucidus</i>	Shining Bronze-Cuckoo	1.394	4	InFo	
<i>Cinclorhampus mathewsi</i>	Rufous Songlark	1.398	4	GrTe	
<i>Certhionyx variegatus</i>	Pied Honeyeater	1.414	4	NeFo	
<i>Lalage tricolor</i>	White-winged Triller	1.415	4	InTe	
<i>Rhipidura leucophrys</i>	Willie-wagtail	1.442	4	InAe	
<i>Cinclorhampus cruralis</i>	Brown Songlark	1.447	4	InTe	
<i>Falcunculus frontatus</i>	Crested Shrike-tit	1.456	4	InBa	x
<i>Chrysococcyx osculans</i>	Black-eared Cuckoo	1.458	4	InFo	
<i>Merops ornatus</i>	Rainbow Bee-eater	1.459	4	InAe	
<i>Melopsittacus undulatus</i>	Budgerigar	1.462	4	GrTe	
<i>Pachycephala inornata</i>	Gilbert's Whistler	1.515	5	InTe	
<i>Pachycephala pectoralis</i>	Golden Whistler	1.515	5	InTe	
<i>Climacteris rufa</i>	Rufous Treecreeper	1.526	5	InBa	
<i>Artamus cinereus</i>	Black-faced Woodswallow	1.544	5	InAe	
<i>Pomatostomus superciliosus</i>	White-browed Babbler	1.544	5	InTe	x
<i>Artamus personatus</i>	Masked Woodswallow	1.549	5	InAe	
<i>Geopelia cuneata</i>	Diamond Dove	1.550	5	GrTe	
<i>Drymodes brunneopygia</i>	Southern Scrub Robin	1.568	5	InTe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Artamus cyanopterus</i>	Dusky Woodswallow	1.602	5	InAe	
<i>Turnix velox</i>	Little Buttonquail	1.613	5	GrTe	
<i>Halcyon sancta</i>	Sacred Kkingfisher	1.620	5	InTe	
<i>Neophema elegans</i>	Elegant Parrot	1.633	5	GrTe	
<i>Glossopsitta porphyrocephala</i>	Purple-crowned Lorikeet	1.641	5	HeFo	
<i>Acanthagenys rufogularis</i>	Spiny-cheeked Honeyeater	1.643	5	InAe	
<i>Cuculus pyrrhophanus</i>	Fan-tailed Cuckoo	1.679	5	InTe	
<i>Aegotheles cristatus</i>	Australian Owlet-Nightjar	1.699	5	InTe	
<i>Halcyon pyrrhopygia</i>	Red-backed Kingfisher	1.719	5	InTe	
<i>Psephotus varius</i>	Mulga Parrot	1.778	6	GrTe	
<i>Oreoica gutturalis</i>	Crested Bellbird	1.792	6	InTe	
<i>Platycercus icterotis</i>	Western Rosella	1.801	6	GrFo	x
<i>Manorina flavigula</i>	Yellow-throated Miner	1.829	6	NeFo	
<i>Cinclosoma castanotus</i>	Chestnut Quail-thrush	1.865	6	GrTe	
<i>Anthochaera chrysoptera</i>	Brush Wattlebird	1.871	6	NeFo	
<i>Colluricincla harmonica</i>	Grey Shrike-thrush	1.879	6	InFo	
<i>Cuculus pallidus</i>	Pallid Cuckoo	1.934	7	HeTe	
<i>Turnix varia</i>	Painted Buttonquail	1.944	7	GrTe	
<i>Caprimulgus guttatus</i>	Large-tailed Nightjar	1.945	7	InAe	
<i>Grallina cyanoleuca</i>	Magpie-lark	1.949	7	InTe	
<i>Nymphicus hollandicus</i>	Cockatiel	1.954	7	GrTe	
<i>Cracticus torquatus</i>	Grey Butcherbird	1.966	7	InFo	
<i>Peltohyas australis</i>	Inland Dotterel	1.966	7	InTe	
<i>Coracina novaehollandiae</i>	Black-faced Cuckoo-shrike	1.970	7	InTe	
<i>Coturnix australis</i>	Brown Quail	1.974	7	GrTe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Polytelis anthopeplus</i>	Regent Parrot	2.057	8	GrTe	
<i>Anthochaera carunculata</i>	Red Wattlebird	2.097	8	NeFo	
<i>Purpureicephalus spurius</i>	Red-capped Parrot	2.107	8	GrFo	
<i>Barnardius zonarius</i>	Port Lincoln Parrot	2.125	8	NeFo	
<i>Coracina maxima</i>	Ground Cuckoo-shrike	2.126	8	InTe	
<i>Cracticus nigrogularis</i>	Pied Butcherbird	2.193	9	InTe	
<i>Falco cenchroides</i>	Australian Kestrel	2.193	9	InAe	
<i>Ninox novaeseelandiae</i>	Morepork	2.241	9	InAe	
<i>Accipiter cirrhocephalus</i>	Collared Sparrowhawk	2.255	9	CaAe	
<i>Ocyphaps lophotes</i>	Crested Pigeon	2.264	9	GrTe	
<i>Vanellus tricolor</i>	Banded Lapwing	2.265	9	InTe	
<i>Phaps elegans</i>	Brush Bronzewing	2.301	9	GrTe	
<i>Geophaps lophotes</i>	Crested Pigeon	2.316	9	GrTe	
<i>Strepera versicolor</i>	Grey Currawong	2.370	9	OmTe	
<i>Elanus notatus</i>	Black-shouldered Kite	2.398	9	CaAe	
<i>Falco longipennis</i>	Australian Hobby	2.403	9	CaAe	
<i>Cacatua leadbeateri</i>	Pink Cockatoo	2.491	10	HeFo	
<i>Phaps chalcoptera</i>	Common Bronzewing	2.491	10	GrFo	
<i>Gymnorhina tibicen</i>	Australian Magpie	2.497	10	InTe	
<i>Cacatua roseicapilla</i>	Galah	2.505	10	GrTe	
<i>Podargus strigoides</i>	Tawny Frogmouth	2.544	10	InTe	
<i>Circus assimilis</i>	Spotted Harrier	2.623	11	InAe	
<i>Ninox connivens</i>	Barking Owl	2.665	11	CaAe	
<i>Lophoictinia isura</i>	Square-tailed Kite	2.700	11	CaAe	
<i>Accipiter fasciatus</i>	Brown Goshawk	2.708	11	CaAe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Tyto alba</i>	Barn Owl	2.719	11	CaAe	
<i>Cacatua sanguinea</i>	Little Corella	2.720	11	GrTe	
<i>Falco berigora</i>	Brown Falcon	2.740	11	CaTe	
<i>Ardea novaehollandiae</i>	White-faced Heron	2.742	11	CaAq	
<i>Tyto novaehollandiae</i>	Australian Masked-Owl	2.785	11	CaAe	
<i>Calyptorhynchus magnificus</i>	Red-tailed Black Cockatoo	2.796	11	HeFo	
<i>Ardea pacifica</i>	Pacific Heron	2.813	11	CaAq	
<i>Corvus coronoides</i>	Australian Raven	2.829	11	CaTe	
<i>Burhinus magirostris</i>	Beach Thick-knee	2.836	11	InTe	x
<i>Cacatua tenuirostris</i>	Long-billed Corella	2.869	11	HeTe	
<i>Falco peregrinus</i>	Peregrine Falcon	2.893	11	CaAe	
<i>Haliastur spheurnus</i>	Whistling Kite	2.903	11	CaAe	
<i>Calyptorhynchus funereus</i>	Yellow-tailed Black-Cockatoo	2.904	11	GrFo	
<i>Hieraaetus morphnoides</i>	Little Eagle	2.924	11	CaAe	
<i>Chenonetta jubata</i>	Maned Duck	2.940	11	HeTe	
<i>Tadorna tadornoides</i>	Australian Shelduck	3.111	12	HeAq	
<i>Threskiornis spinicollis</i>	Straw-necked Ibis	3.255	12	InTe	
<i>Leipoa ocellata</i>	Malleefowl	3.273	12	HeTe	x
<i>Aquila audax</i>	Wedge-tailed Eagle	3.544	12	CaAe	
<i>Ardeotis australis</i>	Australian Bustard	3.799	12	InTe	x
<i>Dromaius novaehollandiae</i>	Emu	4.494	12	HeTe	
Non-Indigenous Species					
<i>Neochmia temporalis</i>	Red-browed Firetail	1.041		GrTe	
<i>Taeniopygia guttata</i>	Zebra Finch	1.079		GrTe	
<i>Streptopelia senegalensis</i>	Laughing Turtledove	2.004		GrTe	

APPENDIX J. Continued.

Latin Name	Common Name	Mass(g)	BCART	Functional Group	Endangered
<i>Coturnix pectoralis</i>	Stubble Quail	2.019		GrTe	
<i>Streptopelia chinensis</i>	Spotted Dove	2.199		GrTe	
<i>Dacelo gigas</i>	Laughing Kookaburra	2.520		CaTe	
<i>Ardeola ibis</i>	Cattle Egret	2.529		InTe	
<i>Columba livia</i>	Domestic Pigeon	2.550		GrTe	
<i>Egretta alba</i>	Large Egret	2.924		PiAq	
<i>Cacatua galerita</i>	Sulphur-crested Cockatoo	2.950		GrFo	
<i>Threskiornis molucca</i>	White Ibis	3.255		InAq	
<i>Cygnus olor</i>	Mute Swan	4.031		HeAq	

APPENDIX K. Body mass of mammals reconstructed in Bridger Zone 1 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Pontifactor	21	1
Mysops	40	1
Uintasorex	40	1
Entomolestes	49	1
Centetodon	54	1
Nyctitherium	73	1
Apatemys	79	1
Talpavus	107	1
Trogolemur	121	1
Uintanius	158	2
Gazinius	229	2
Scenopagus	242	2
Microparamys	393	2
Omomys	440	2
Sciuravus	794	3
Pauromys	857	3
Oodectes	858	3
<i>Leptotomus bridgerensis</i>	1160	3
Paramys delicatus	1355	3
Peradectes	1728	3
Hemiacodon	1862	3
Pantolestes	2228	3
Antiacodon	2650	3
Homacodon	5386	4
Notharctus	5500	4
Peratherium	5568	4
Orohippus	8240	4
Viverravus	10203	4
Helohyus	13251	4
<i>Thisbemys corrugatus</i>	14500	4
Helaletes	15748	4
<i>Hyrachyus eximius</i>	127880	5
Mesonyx	303000	5
Palaeosyops	418000	5
Uintatherium	524283	5

APPENDIX L. Body mass of mammals reconstructed in Bridger Zone 2 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Entomolestes	49	1
Apatemys	79	1
Talpavus	107	1
<i>Washakius insignis</i>	222	1
Scenopagus	242	1
Omomys	440	2
Sciuravus	794	2
Pauromys	857	2
<i>Leptotomus bridgerensis</i>	1160	2
<i>Paramys delicatus</i>	1355	3
Peradectes	1728	3
Hemicacodon	1862	3
Microsus	2188	3
Pantolestes	2228	3
Antiacodon	2650	3
Pseudotomus	3378	3
Homacodon	5386	4
Notharctus	5500	4
Peratherium	5568	4
Orohippus	8230	4
Viverravus	10203	4
<i>Thisbemys corrugatus</i>	14500	5
Helaletes	15748	5
Didelphodus	18446	5
Patriofelis	21079	5
<i>Ischyrotomus oweni</i>	32000	5
<i>Hyrachyus eximius</i>	127880	6
Uintatherium	524283	6

APPENDIX M. Body mass of mammals reconstructed in Bridger Zone 3 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Entomolestes	49	1
Centetodon	54	1
Taxymys	60	1
Nyctitherium	73	1
Apatemys	79	1
Uintanius	158	1
Scenopagus	242	2
Microparamys	393	2
Omomys	440	2
Sciuravus	794	2
Oodectes	858	2
<i>Paramys delicatus</i>	1355	3
Hemiacodon	1862	3
Smilodectes	2001	3
Microsus	2188	3
Pantolestes	2228	3
Microsyops	2300	3
Antiacodon	2650	3
Notharctus	5500	4
Peratherium	5568	4
<i>Limnocyon verus</i>	6290	4
Orohippus	8230	4
Viverravus	10203	4
Helohyus	13251	4
<i>Thisbemys corrugatus</i>	14500	4
Helaletes	15748	4
<i>Hyrachyus eximius</i>	127880	5
Isectolophus	327000	5

APPENDIX N. Body mass of mammals reconstructed in Bridger Zone 4 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Nyctitherium	73	1
Apatemys	79	1
Scenopagus	242	1
Omomys	440	1
Hemiacodon	1862	2
Pantolestes	2228	2
Microsypops	2300	2
Antiacodon	2650	2
Notharctus	5500	3
Peratherium	5568	3
Orohippus	8230	3
Thinocyon	9072	3
Viverravus	10203	3
Helohyus	13251	3
<i>Thisbemys corrugatus</i>	14500	3
Helalestes	15748	3
<i>Hyrachyus eximius</i>	127880	4
Mesonyx	303000	4

APPENDIX O. Body mass of mammals reconstructed in Bridger Zone 5 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Pontifactor	21	1
Uintasorex	40	1
Entomolestes	49	1
Centetodon	54	1
Taxymys	60	1
Nyctitherium	73	1
Apatemys	79	1
Tillomys	81	1
Trogolemur	121	1
Namatomys	190	2
Scenopagus	242	2
Omomys	440	2
Sciuravus	794	3
Pauromys	857	3
<i>Paramys delicatus</i>	1355	3
Peradectes	1728	3
Pantolestes	2228	3
Microsyops	2300	3
Notharctus	5500	4
Peratherium	5568	4
Triplopus	31000	5
Isectolophus	327000	5

APPENDIX P. Body mass of mammals reconstructed in Uinta Zone 1 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Protoptychus	98	1
Microparamys	393	1
Sciuravus	794	2
Pareumys	963	2
Mesomeryx	1325	2
<i>Ischyrotomus compressidens</i>	1673	2
<i>Leptotomus leptodus</i>	2508	2
Bunomeryx	2793	2
Peratherium	5568	3
Leptoreodon	7557	3
Epihippus	11075	3
Tapocyon	11210	3
Protoreodon	14219	3
<i>Thisbemys medius</i>	20600	4
<i>Isectolophus annectens</i>	27500	4
Triplopus	31000	4
Epitriplopus	58500	4
Achaenodon	334000	5
Amynodon	375000	5

APPENDIX Q. Body mass of mammals reconstructed in Uinta Zone 2 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Protoptychus	98	1
Trogolemur	121	1
Spurimus	150	1
Uintamys	404	2
Mytonolagus	780	2
Sciuravus	794	2
Pareumys	963	2
Mesomeryx	1325	2
Ourayia	1832	3
<i>Leptotomus leptodus</i>	2508	3
Bunomeryx	2793	3
Peratherium	5568	4
<i>Reithroparamys gidleyi</i>	6900	4
Epihippus	11075	4
Tapocyon	11210	4
Protoreodon	14219	4
<i>Thisbemys medius</i>	20600	5
<i>Isectolophus annectens</i>	27500	5
Triplopus	31000	5
<i>Oxyaenodon dysodus</i>	41075	5
Simidectes	48888	5
Epitriplopus	58500	5
<i>Limnocyon potens</i>	88775	6
<i>Ischyrotomus eugenei</i>	132000	6
Amynodon	375000	6

APPENDIX R. Body mass of mammals reconstructed in Uinta Zone 3 faunal assemblage. Bayesian CART groups used to identify each body mass aggregation.

Latin Name	Mass(g)	CART
Microparamys	393	1
Mytonolagus	780	1
Sciuravus	794	1
Pareumys	963	1
Ourayia	1832	2
Mytonomys	2144	2
<i>Leptotomus leptodus</i>	2508	2
Pentacemylus	4325	3
Auxontodon	5278	3
Leptoreodon	7557	3
Epihippus	11075	4
Tapocyon	11210	4
Protoreodon	14219	4
Diplobunops	18800	4
<i>Isectolophus annectens</i>	27500	5
Triplopus	31000	5
Colodon	40500	5
Simidectes	48888	5
Epitriplopus	58500	5
Proviverra	63772	5

APPENDIX S. The data of one hundred countries used in Chapter 6 analyses. Environmental Sustainability Index (ESI) and Environmental Vulnerability Index (EVI).

Country	Endangered Birds (%)	Endangered Mammals (%)	Invasive Birds (%)	Invasive Mammals (%)	Resilience (%)	ESI	EVI	Total Population (1000)
Albania	1.714	4.054	0.286	1.351	2.594	58.8	330	3,130
Algeria	6.349	14.286	1.003	1.905	2.757	46	275	32,850
Argentina	4.776	9.626	0.682	4.011	7.643	62.7	287	38,750
Armenia	3.343	10.000	0.000	2.222	5.122	53.2	247	3,020
Australia	6.349	22.350	3.515	14.040	17.384	61	238	20,160
Austria	2.069	4.444	1.839	12.222	6.095	62.7	369	8,190
Azerbaijan	4.076	6.796	0.000	4.854	5.732	45.4	354	8,410
Bolivia	2.021	5.234	0.139	0.275	2.836	59.5	250	9,180
Brazil	7.025	13.117	0.228	0.772	9.045	62.2	281	186,400
Bulgaria	3.023	7.447	0.504	2.128	4.684	50	323	7,730
Burundi	1.168	6.618	0.146	0.000	2.192	40	288	7,550
Cameroon	1.567	12.239	0.000	0.000	4.334	52.5	229	16,320
Canada	2.719	5.941	1.208	5.446	5.671	64.4	251	32,270
Central African Republic	0.646	3.196	0.000	0.000	1.208	58.7	193	4,040
Chile	6.4	14.685	1.000	8.392	10.886	53.6	287	16,300
China	6.523	13.612	0.153	1.089	9.061	38.6	360	1,315,840
Colombia	4.567	11.991	0.106	0.905	6.237	58.9	296	45,600
Costa Rica	1.917	3.524	0.225	0.881	2.603	59.6	354	4,330
Côte d'Ivoire	1.884	9.524	0.000	0.000	3.819	47.3	248	18,150
Cuba	5.163	30.769	2.446	15.385	13.395	52.3	329	11,270
Czech Republic	1.474	2.564	1.966	14.103	5.567	46.6	315	10,220
Democratic Republic of Congo	2.661	6.744	0.000	0.000	3.762	44.1	288	57,550
Dominican Republic	5.19	28.571	3.460	0.000	11.538	43.7	324	8,890

APPENDIX S. Continued.

Country	Endangered Birds (%)	Endangered Mammals (%)	Invasive Birds (%)	Invasive Mammals (%)	Resilience (%)	ESI	EVI	Total Population (1000)
Ecuador	4.169	12.366	0.181	0.806	5.969	52.4	304	13,230
Egypt	2.132	17.143	0.853	3.810	6.272	44	298	74,030
El Salvador	0.513	3.086	0.684	0.000	1.606	43.8	348	6,880
Finland	0.887	1.613	0.887	9.677	2.924	75.1	265	5,250
France	1.056	8.130	2.465	11.382	6.368	55.2	361	60,500
Gabon	0.667	7.143	0.000	1.099	2.146	61.7	211	1,380
Gambia	0.87	6.767	0.174	0.000	2.119	50	277	1,520
Georgia	2.793	9.804	0.000	0.980	4.565	51.5	261	4,470
Germany	1.174	5.941	2.153	14.851	6.209	57	357	82,690
Ghana	1.083	6.615	0.000	0.000	2.510	52.8	279	22,110
Greece	2.477	9.901	0.676	3.960	5.138	50.1	353	11,120
Guinea-Bissau	0.405	8.943	0.000	0.000	2.107	48.6	271	1,590
Honduras	0.947	3.774	0.271	0.000	1.788	47.4	273	7,200
Hungary	2.261	2.500	1.005	5.000	3.975	52	363	10,100
India	6.194	23.301	0.081	1.214	10.860	45.2	385	1,103,370
Indonesia	7.241	27.463	0.375	2.985	14.349	48.8	316	222,780
Iran	3.839	8.602	0.576	1.075	5.799	39.8	313	69,520
Ireland	0.209	10.000	0.626	14.000	3.025	59.2	318	4,150
Israel	2.403	13.889	3.327	0.926	7.242	50.9	380	6,720
Italy	1.495	7.080	0.748	5.310	4.012	50.1	386	58,090
Jamaica	3.988	16.327	4.601	6.122	10.400	44.7	381	2,650
Japan	7.12	19.444	1.942	8.333	12.598	57.3	389	128,080
Jordan	1.942	16.049	0.485	0.000	4.665	47.8	310	5,700

APPENDIX S. Continued.

Country	Endangered Birds (%)	Endangered Mammals (%)	Invasive Birds (%)	Invasive Mammals (%)	Resilience (%)	ESI	EVI	Total Population (1000)
Kazakhstan	4.158	10.323	0.000	2.581	6.212	48.6	215	14,830
Kenya	2.4	7.181	0.444	0.532	4.064	45.3	262	34,260
Kyrgyzstan	3.183	8.451	0.265	7.042	5.357	48.4	234	5,260
Laos	3.286	21.395	0.286	0.000	7.760	52.4	243	5,920
Lebanon	4.556	14.925	1.075	0.000	1.613	40.5	387	3,580
Malaysia	5.534	20.833	0.515	1.190	10.872	54	312	25,350
Mali	0.982	8.824	0.000	0.000	2.410	53.7	215	13,520
Mauritania	1.476	13.274	0.000	0.000	3.511	42.6	233	3,070
Mexico	5.341	20.268	0.552	1.147	10.938	46.2	306	107,030
Mongolia	4.907	8.088	0.000	0.735	5.851	50	208	2,650
Morocco	2.041	14.286	0.204	4.511	5.778	44.8	315	31,480
Namibia	3.043	5.446	0.000	1.485	3.924	56.8	200	2,030
Nepal	3.556	17.582	0.111	0.549	6.100	47.7	305	27,130
Netherlands	0.433	5.128	1.515	11.538	4.074	53.7	388	16,300
New Zealand	24.047	18.182	11.730	70.455	41.818	61	292	4,030
Nicaragua	1.202	2.463	0.267	0.000	1.681	50.2	272	5,490
Nigeria	1.268	9.825	0.000	0.000	3.249	45.4	336	131,530
Norway	0.425	9.333	0.425	5.333	2.747	73.4	273	4,620
Pakistan	3.659	12.042	0.000	2.618	5.920	39.9	373	157,940
Panama	1.763	5.691	0.415	0.407	2.975	57.7	247	3,230
Paraguay	3.808	4.848	0.282	1.212	4.462	59.7	260	6,160
Peru	5.07	11.777	0.162	1.285	6.807	60.4	268	27,970
Philippines	11.337	18.841	0.508	2.415	14.286	42.3	402	83,050

APPENDIX S. Continued.

Country	Endangered Birds (%)	Endangered Mammals (%)	Invasive Birds (%)	Invasive Mammals (%)	Resilience (%)	ESI	EVI	Total Population (1000)
Poland	1.33	6.667	1.109	6.667	4.251	45	354	38,530
Portugal	1.782	12.222	1.782	2.222	5.210	54.2	335	10,490
Romania	3.158	7.368	0.263	5.263	5.263	46.2	335	21,710
Rwanda	1.389	10.326	0.000	0.000	3.208	44.8	298	9,040
Saudi Arabia	5.429	11.905	1.027	2.381	2.875	37.8	274	24,570
Senegal	1.196	8.466	0.000	0.000	2.797	51.1	277	11,660
Sierra Leone	1.504	8.421	0.000	0.000	3.041	43.4	283	5,530
Slovakia	1.939	3.529	0.554	0.000	2.691	52.8	303	5,400
Slovenia	1.061	4.651	1.061	1.163	2.808	57.5	362	1,970
South Africa	4.118	8.081	1.059	3.030	6.713	46.2	324	47,430
South Korea	5.693	13.889	0.380	1.389	7.179	43	373	47,820
Spain	2.98	13.913	2.607	6.957	8.282	48.8	352	43,060
Sri Lanka	2.921	25.641	0.449	2.564	8.541	48.5	331	20,740
Sudan	1.304	5.338	0.100	1.423	2.582	35.9	274	36,230
Sweden	0.63	1.389	0.210	16.667	3.102	71.7	311	9,040
Switzerland	0.495	2.381	1.238	0.000	1.844	63.7	348	7,250
Syria	6.224	17.021	0.000	1.064	3.351	43.8	350	19,040
Tajikistan	2.535	11.765	0.000	2.941	4.492	38.6	271	6,510
Tanzania	3.636	9.471	0.273	0.557	5.415	50.3	257	38,330
Thailand	4.427	18.650	0.101	0.643	8.046	49.8	308	64,230
Togo	0.302	5.714	0.000	0.000	1.434	44.5	293	6,150
Trinidad & Tobago	0.429	1.754	0.644	2.632	1.724	36.3	381	1,310
Turkey	3.049	11.409	0.610	1.342	5.772	46.6	353	73,190

APPENDIX S. Continued.

Country	Endangered Birds (%)	Endangered Mammals (%)	Invasive Birds (%)	Invasive Mammals (%)	Resilience (%)	ESI	EVI	Total Population (1000)
Uganda	1.713	6.583	0.000	0.627	2.993	51.3	283	28,820
Ukraine	2.804	9.821	1.168	8.036	6.852	44.7	317	46,480
United Arab Emirates	6.029	15.217	3.218	0.000	1.839	44.6	293	4,500
United Kingdom	0.334	6.757	1.836	47.297	7.875	50.2	373	59,670
United States	9.209	9.091	4.372	6.136	14.059	53	300	298,210
Venezuela	1.849	9.091	0.427	0.551	3.787	48.1	291	26,750
Viet Nam	4.615	18.815	0.355	0.697	8.657	42.3	357	84,240
Zambia	1.435	3.419	0.000	1.282	2.150	51.1	210	11,670

APPENDIX T. The data of one hundred countries used in Chapter 6 analyses. Life Expectancy (LE) and Pesticide Regulations (PR).

Country	GDP per capita (\$)	Total Land Area (km ²)	Latitude (°)	LE	Water Stress (%)	Agricultural Intensity (%)	PR	Adult Literacy (%)	Tourism (1000 people)
Albania	4,955	28,300	41	78	0.0	6.2	2	98.7	748
Algeria	6,376	2,302,500	28	74	24.5	55.9	15	69.9	1,443
Argentina	13,652	2,736,300	34	77	24.1	13.7	20	97.2	3,895
Armenia	5,011	28,300	40	73	68.6	3.5	22	99.4	319
Australia	30,678	7,634,600	27	82	45.7	12.9	22	99	5,497
Austria	30,736	83,200	47	80	0.0	23.3	22	99	19,952
Azerbaijan	5,953	85,400	40	67	31.4	5.6	1	98.8	1,177
Bolivia	2,579	1,069,100	17	67	2.1	0	4	86.7	413
Brazil	7,826	8,511,000	10	72	2.3	2	20	88.6	5,358
Bulgaria	8,754	111,300	43	73	36.5	18.4	22	98.2	7,282
Burundi	630	25,200	3	52	0.0	5.1	22	59.3	148
Cameroon	2,079	465,800	6	54	0.0	12.8	2	67.9	176
Canada	30,278	9,458,900	60	81	1.7	25.6	22	99	18,770
Central African Republic	1,111	622,900	7	45	0.5	0	13	48.6	12
Chile	10,939	721,200	30	77	16.5	0.4	22	95.7	2,027
China	6,621	9,198,100	35	74	19.6	10.7	13	90.9	120,292
Colombia	6,886	1,141,200	4	73	2.8	0	19	92.8	933
Costa Rica	9,646	51,000	10	78	0.0	4.1	16	94.9	1,679
Côte d'Ivoire	1,471	320,300	8	56	1.8	1.1	17	48.7	180
Cuba	4,100	111,200	21	78	28.7	34.2	14	99.8	2,319
Czech Republic	19,700	78,600	49	77	2.6	28.6	22	99	6,336
Democratic Republic of Congo	700	2,313,400	0	54	0.0	0.1	3	67.2	61
Dominican Republic	7,618	48,100	19	74	20.4	13.8	21	87	3,691

APPENDIX T. Continued.

Country	GDP per capita (\$)	Total Land Area (km ²)	Latitude (°)	LE	Water Stress (%)	Agricultural Intensity (%)	PR	Adult Literacy (%)	Tourism (1000 people)
Ecuador	3,982	256,300	2	75	19.2	1	19	91	861
Egypt	4,031	968,100	27	72	25.5	45.7	19	71.4	8,608
El Salvador	4,776	20,300	14	72	0.0	31.7	17	80.6	1,154
Finland	30,420	317,000	64	79	0.4	15.3	22	99	2,080
France	28,877	547,100	46	81	8.4	29	21	99	76,001
Gabon	5,835	265,100	1	53	0.0	0.8	3	84	222
Gambia	1,745	10,800	13	55	0.0	6	21	99	90
Georgia	3,304	69,200	42	77	7.0	3	3	100	560
Germany	27,438	356,000	51	79	15.9	17.2	22	99	21,500
Ghana	2,299	231,700	8	60	0.0	10.6	17	57.9	429
Greece	21,675	131,900	39	80	4.5	9.4	21	96	14,276
Guinea-Bissau	745	34,100	12	48	0.0	0	1	99	5
Honduras	3,170	112,100	15	69	2.3	1.3	1	80	673
Hungary	16,928	92,000	47	73	24.5	40.7	21	99	3,446
India	3,308	3,208,100	20	70	33.5	50.6	3	61	3,919
Indonesia	3,570	1,897,800	5	71	0.2	10.9	19	90.4	5,002
Iran	7,405	1,590,400	32	71	25.3	13.2	20	82.4	1,659
Ireland	36,238	69,500	53	78	0.0	2.9	21	99	7,334
Israel	23,020	21,900	31	81	75.3	29.4	1	97.1	1,903
Italy	26,496	299,300	43	80	17.7	21.9	21	98.4	36,513
Jamaica	3,907	11,100	18	74	0.0	10.2	20	79.9	1,479
Japan	27,992	371,700	36	82	5.6	1.7	22	99	6,728
Jordan	5,176	88,400	31	79	75.0	23.7	22	91.1	2,987

APPENDIX T. Continued.

Country	GDP per capita (\$)	Total Land Area (km ²)	Latitude (°)	LE	Water Stress (%)	Agricultural Intensity (%)	PR	Adult Literacy (%)	Tourism (1000 people)
Kazakhstan	7,652	2,619,400	48	68	20.1	8.7	10	99.5	4,365
Kenya	1,137	579,600	1	58	13.9	1.3	4	73.6	1,146
Kyrgyzstan	1,749	185,300	41	69	20.5	0	18	98.7	315
Laos	2,013	230,200	18	57	0.0	0.3	19	68.7	1,095
Lebanon	4,876	10,300	34	74	10.0	14.5	20	99	1,140
Malaysia	10,091	330,800	2	73	0.7	1.8	20	88.7	16,431
Mali	942	1,248,100	17	50	13.5	0	4	24	143
Mauritania	2,161	1,036,900	20	60	15.8	0	3	51.2	30
Mexico	9,967	1,943,100	23	76	31.5	9.7	18	91.6	21,915
Mongolia	2,034	1,546,300	46	68	11.3	0.2	17	97.8	338
Morocco	4,346	403,800	32	72	47.6	58.7	19	52.3	5,843
Namibia	7,038	820,000	22	51	52.0	0	3	85	778
Nepal	1,379	139,100	28	66	0.9	7.9	13	48.6	375
Netherlands	31,306	35,000	52	79	24.1	9.4	21	99	10,012
New Zealand	23,109	265,300	41	80	1.2	1.7	22	99	2,366
Nicaragua	3,539	118,800	13	72	0.0	4.9	5	76.7	712
Nigeria	1,008	904,200	10	47	4.7	27.2	3	69.1	2,778
Norway	37,667	318,500	62	80	0.0	8.7	22	99	3,859
Pakistan	2,206	785,300	30	65	33.4	34.3	2	49.9	798
Panama	7,234	74,500	9	77	2.6	0	21	91.9	576
Paraguay	4,368	395,900	23	76	23.5	3.1	21	93.5	341
Peru	5,725	1,288,300	10	71	16.7	0.1	21	87.9	1,486
Philippines	4,731	295,400	13	71	3.0	6.9	18	92.6	2,623

APPENDIX T. Continued.

Country	GDP per capita (\$)	Total Land Area (km ²)	Latitude (°)	LE	Water Stress (%)	Agricultural Intensity (%)	PR	Adult Literacy (%)	Tourism (1000 people)
Poland	13,349	311,200	52	76	5.6	37.5	21	99	4,310
Portugal	18,966	91,400	39	78	10.0	19.5	21	93.8	5,676
Romania	8,722	237,100	46	73	17.2	42.3	22	97.3	5,839
Rwanda	1,105	24,300	2	51	0.0	13.4	4	64.9	113
Saudi Arabia	14,769	1,942,700	25	76	51.6	24.6	20	82.9	8,037
Senegal	1,599	196,200	14	59	13.4	2.4	4	39.3	387
Sierra Leone	753	72,600	8	41	0.0	0	1	34.8	40
Slovakia	15,409	48,900	48	75	0.0	30.4	22	99	1,515
Slovenia	20,890	20,200	46	77	0.0	2.3	19	99.7	1,545
South Africa	10,338	1,217,600	29	49	54.8	4.8	14	82.4	7,369
South Korea	20,572	99,000	37	79	9.7	4.2	15	99	6,023
Spain	24,681	505,300	40	80	37.1	31.6	21	99	55,914
Sri Lanka	4,391	65,800	7	75	16.5	13	18	90.7	549
Sudan	2,050	2,492,400	15	51	10.7	1.1	21	60.9	246
Sweden	30,392	431,700	62	81	0.4	15.8	22	99	7,627
Switzerland	32,775	39,000	47	81	0.0	4.3	22	99	7,229
Syria	3,497	184,400	35	71	55.6	58.2	21	80.8	3,368
Tajikistan	1,257	130,100	39	65	14.0	0.7	3	99.5	5
Tanzania	650	891,000	6	52	10.8	0.1	4	69.4	613
Thailand	8,065	513,600	15	73	8.8	11.7	20	92.6	11,567
Togo	1,306	57,300	8	59	0.0	33.5	16	53.2	81
Trinidad & Tobago	14,708	5,200	11	71	0.0	16	19	98.4	463
Turkey	7,842	768,700	39	72	13.9	14.2	19	87.4	20,273

APPENDIX T. Continued.

Country	GDP per capita (\$)	Total Land Area (km ²)	Latitude (°)	LE	Water Stress (%)	Agricultural Intensity (%)	PR	Adult Literacy (%)	Tourism (1000 people)
Uganda	1,313	207,100	1	53	1.4	31.9	1	66.8	468
Ukraine	6,605	588,400	49	68	24.2	62.3	16	99.4	12,514
United Arab Emirates	22,698	74,800	24	76	41.6	0	3	88.7	5,871
United Kingdom	30,237	247,200	54	79	8.4	20.5	21	99	29,970
United States	38,165	9,210,800	38	78	21.3	16.8	19	99	49,206
Venezuela	6,485	911,800	8	74	9.7	0.9	3	93	706
Viet Nam	2,925	328,800	16	72	3.0	11.8	20	90.3	3,468
Zambia	949	745,300	15	39	0.1	0.1	9	68	669

APPENDIX U. The data of one hundred countries used in Chapter 6 analyses. Under Nourishment (UN), Total Biodiversity (TB) and Political Stability (PS).

Country	UN (%)	Annual Rainfall (mm)	Energy Efficiency (Terajoules/million GDP)	Wilderness Protection (%)	TB	Exports/Imports (million \$)	PS	Women in Govt. (%)
Albania	6	1,136	6,751	8.1	3,569	2269/596	35	7
Algeria	5	257	6,797	5.2	3,846	18200/32300	13	7
Argentina	2.5	1,062	6,120	6.6	11,387	22445/34550	50	40
Armenia	29	497	15,417	8.6	4,010	1351/723	42	8
Australia	2.5	527	8,960	10.6	19,463	96507/80218	79	30
Austria	2.5	1,230	5,833	28.2	3,698	117708/117360	94	27
Azerbaijan	10	625	21,371	7.4	4,849	3504/3614	24	11
Bolivia	23	1,507	8,241	21.8	19,611	1844/2146	18	15
Brazil	8	1,940	6,402	29.7	60,322	65946/96475	37	9
Bulgaria	9	730	15,195	10.1	4,196	14400/9912	61	22
Burundi	67	1,042	1,650	6.1	3,324	176/47	9	32
Cameroon	25	1,778	2,300	10.3	10,059	2400/2600	31	14
Canada	2.5	641	14,227	8.7	4,164	273526/316735	85	25
Central African Republic	45	1,309	1,362	18.2	4,614	120/130	7	11
Chile	4	2,287	6,832	19.7	6,222	24871/32025	66	13
China	12	1,119	7,079	15.8	35,082	1003665/1029408	32	21
Colombia	14	2,708	3,805	26.2	54,967	16723/16431	8	10
Costa Rica	4	2,926	4,462	31.1	13,864	8268/6297	78	37
Côte d'Ivoire	14	1,912	4,027	21.3	4,887	3588/6243	3	9
Cuba	2.5	1,103	14,968	18.7	7,448	5000/1700	49	43
Czech Republic	2.5	818	9,418	15.8	2,448	69476/68624	77	16

APPENDIX U. Continued.

Country	UN (%)	Annual Rainfall (mm)	Energy Efficiency (Terajoules/million GDP)	Wilderness Protection (%)	TB	Exports/Imports (million \$)	PS	Women in Govt. (%)
Democratic Republic of Congo	72	1,566	2,139	11.8	13,208	600/400	2	8
Dominican Republic	27	821	4,699	28.8	6,250	5370/1334	49	17
Ecuador	5	2,320	8,247	28.1	22,311	7861/7634	19	25
Egypt	3	107	8,732	8.0	3,077	17975/10453	22	4
El Salvador	11	1,305	3,643	1.4	3,731	5989/3382	44	17
Finland	2.5	410	8,349	9.9	1,653	50799/61304	99	42
France	2.5	818	6,685	15.5	5,549	465229/448498	65	20
Gabon	5	1,504	4,561	16.7	7,724	1370/4200	52	16
Gambia	27	598	1,550	2.0	1,816	230/30	38	9
Georgia	13	1,321	11,490	3.9	4,841	1848/648	24	6
Germany	2.5	945	6,382	56.3	3,425	716580/911821	81	31
Ghana	12	1,112	2,762	17.1	5,000	5214/2327	53	11
Greece	2.5	748	6,215	3.4	5,758	52552/15190	63	15
Guinea-Bissau	37	1,223	4,960	19.3	1,730	69/69	30	14
Honduras	22	1,234	5,355	21.0	7,119	3922/1534	31	23
Hungary	2.5	559	6,909	5.7	2,751	59303/54830	68	11
India	20	1,432	4,571	4.9	26,769	94070/71798	18	9
Indonesia	6	2,749	6,485	15.8	36,760	52076/71261	15	12
Iran	4	258	13,048	7.2	9,011	37739/40710	11	3
Ireland	2.5	1,273	4,014	1.1	1,796	60621/104231	89	16
Israel	2.5	427	5,780	34.9	3,251	41036/38520	13	14

APPENDIX U. Continued.

Country	UN (%)	Annual Rainfall (mm)	Energy Efficiency (Terajoules/million GDP)	Wilderness Protection (%)	TB	Exports/Imports (million \$)	PS	Women in Govt. (%)
Italy	2.5	932	5,090	7.1	6,477	350865/348984	62	20
Jamaica	10	1,628	15,398	20.8	3,914	4071/1586	43	14
Japan	2.5	1,551	6,248	14.4	7,491	455254/565822	85	12
Jordan	7	179	10,528	10.6	2,744	8179/3883	34	9
Kazakhstan	8	277	21,143	2.9	6,746	12781/20096	58	12
Kenya	31	998	4,642	12.3	8,667	4553/2684	16	10
Kyrgyzstan	4	419	20,967	3.4	4,803	941/719	15	26
Laos	21	1,933	4,833	16.3	9,460	506/361	43	25
Lebanon	3	656	10,179	0.4	3,571	9397/1747	4	5
Malaysia	3	2,822	9,851	20.2	17,539	105284/126512	52	15
Mali	28	536	1,318	2.1	2,640	1131/932	39	10
Mauritania	10	199	9,024	0.9	1,909	480/400	32	20
Mexico	5	1,052	7,153	8.0	29,510	128723/101252	25	22
Mongolia	28	156	20,733	14.1	3,382	801/616	68	4
Morocco	6	340	4,117	1.4	4,486	17525/9667	27	6
Namibia	23	339	4,102	15.1	4,347	2180/1830	79	27
Nepal	17	1,648	1,781	17.6	8,226	1890/716	3	33
Netherlands	2.5	893	8,000	23.4	1,857	319176/358015	76	38
New Zealand	2.5	1,248	9,838	30.1	3,094	22163/19830	94	33
Nicaragua	27	1,869	3,584	18.5	8,883	1879/605	36	19
Nigeria	9	1,324	6,931	16.3	6,227	14164/31148	4	7
Norway	2.5	1,241	10,689	6.3	2,321	47929/81544	93	36

APPENDIX U. Continued.

Country	UN (%)	Annual Rainfall (mm)	Energy Efficiency (Terajoules/million GDP)	Wilderness Protection (%)	TB	Exports/Imports (million \$)	PS	Women in Govt. (%)
Pakistan	23	438	6,160	9.1	6,114	17950/13380	1	21
Panama	25	2,619	9,763	28.5	11,759	3594/944	50	17
Paraguay	15	1,205	15,943	6.1	8,953	3004/1985	28	14
Peru	12	1,988	3,997	13.8	20,247	982/12617	20	29
Philippines	19	2,199	3,603	17.4	11,079	42348/39700	10	20
Poland	2.5	588	8,036	24.4	3,065	89131/74818	67	18
Portugal	2.5	1,039	5,618	6.6	5,872	54888/35750	73	28
Romania	2.5	620	9,464	10.7	3,993	32664/23485	51	10
Rwanda	36	1,115	1,298	8.2	3,306	284/98	37	51
Saudi Arabia	4	151	18,749	42.4	2,831	44500/95369	25	0
Senegal	23	576	3,905	25.1	3,163	2860/1270	38	29
Sierra Leone	50	2,546	3,451	4.1	3,127	607/180	33	13
Slovakia	6	747	11,138	19.7	3,620	29457/27534	80	19
Slovenia	3	989	7,898	6.6	3,787	17189/15823	84	10
South Africa	4	640	10,129	6.2	25,681	48240/45720	51	34
South Korea	2.5	1,142	9,432	4.5	3,544	224463/253845	62	14
Spain	2.5	645	6,229	9.5	5,983	249187/178521	45	34
Sri Lanka	22	1,700	2,724	20.5	4,440	7973/5757	6	6
Sudan	27	741	2,156	4.6	4,698	2711/2609	2	17
Sweden	2.5	561	8,238	10.9	2,390	99276/122478	95	47
Switzerland	2.5	1,502	5,361	30.3	3,599	111740/118673	99	27
Syria	4	366	14,076	0.7	3,592	5300/7000	25	12

APPENDIX U. Continued.

Country	UN (%)	Annual Rainfall (mm)	Energy Efficiency (Terajoules/million GDP)	Wilderness Protection (%)	TB	Exports/Imports (million \$)	PS	Women in Govt. (%)
Tajikistan	61	440	37,289	15.1	5,491	1375/915	19	20
Tanzania	44	1,116	3,493	41.6	12,237	2508/1330	40	30
Thailand	21	1,634	6,654	20.3	13,648	94382/96064	17	13
Togo	25	1,117	2,316	11.0	4,034	556/367	27	11
Trinidad & Tobago	11	1,983	37,706	34.5	2,961	4858/6373	44	33
Turkey	3	615	6,690	2.0	9,549	97540/63121	21	9
Uganda	19	1,401	1,010	17.5	6,541	2014/886	14	31
Ukraine	3	525	23,643	3.5	5,678	28996/32672	50	8
United Arab Emirates	2.5	52	30,690	0.3	380	58000/88000	73	23
United Kingdom	2.5	1,130	5,668	22.0	2,738	463244/346695	66	20
United States	2.5	939	9,112	14.7	22,575	1525481/818520	56	17
Venezuela	18	1,813	22,593	71.4	23,675	16700/33929	12	19
Viet Nam	17	1,740	4,880	5.7	12,291	25227/20176	56	26
Zambia	47	1,049	11,906	41.5	5,989	1691/1198	54	15