

Relative effects of landscape structure variables, and interactions with life-history traits, on the abundance and distribution of wetland-dependent vertebrate species

by

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Abstract

Wetland-dependent species are undergoing the largest wildlife population declines worldwide primarily due to habitat loss, but are a relatively understudied ecological group in landscape ecology. The overarching goal of my dissertation is to advance our understanding of the effects of landscape structure on wetland-dependent vertebrate populations and the life-history traits that determine these responses. In Chapter 2, I used meta-analysis to combine data on the relationship between species abundance and wetland habitat loss from empirical studies conducted worldwide to identify the traits that influence species response to habitat loss. I show that species with low reproductive rates are more sensitive to habitat loss at the population level, whereas species mobility has no apparent effect. In Chapter 3, I conducted extensive field surveys to determine the relative effects of landscape structure variables on the distribution of wetland birds and turtles. I show that birds are more strongly affected by habitat loss whereas turtles are more strongly affected by the surrounding landscape matrix, specifically forest cover. I also show that habitat fragmentation independent of habitat loss was not an important predictor for any species. In Chapter 4, I investigated whether a similar pattern is found in other studies and vertebrate taxa by building upon the meta-analysis in Chapter 2 and comparing wetland species responses to wetland loss and landscape matrix quality at the population level. I show that wetland loss is more important for mammals and birds whereas matrix quality, indexed as forest cover, is more important for amphibians. I also show that these results are not scale-dependent. The results of this thesis suggest that conservation priority should be placed on species

with low reproductive rates and on actions that increase reproductive output rather than movement. My results also suggest that landscape-scale requirements vary among wetland-dependent taxa: mammals and birds are more limited by the amount of wetland habitat in a landscape, whereas amphibians and reptiles are more limited by the quality of the matrix surrounding wetlands, likely due to access to and/or availability of complementary habitats. Therefore, conservation policies that focus only on wetland protection will not conserve wetland biodiversity.

Preface

Co-authorship Statement

This thesis is formatted using the integrated thesis format, and therefore each data chapter was written as an independent manuscript. Chapters 2 and 3 have been published in, and Chapter 4 has been submitted to, a peer-reviewed journal when this thesis was completed. The text of each published chapter was reproduced in whole, but supplementary figures were added. There is some repetition in the introductions, methods and discussions; however I have cross-referenced between chapters to reduce repetition as much as possible.

I performed the majority of work described in this thesis as follows. I proposed and developed the research questions in cooperation with Lenore Fahrig and Kathryn Lindsay, and I was primarily responsible for the design of the studies used to address these questions. I carried out all field work for Chapter 3 and collected all data for Chapters 2 and 4. I analysed all of the data and wrote all first drafts of the data chapters.

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Chapter 1: General Introduction

Loss of habitat due to human land conversion is the single largest threat to the survival of many species and to global biodiversity, in general (Kerr and Deguise 2004, Millennium Ecosystem Assessment 2005). Land conversion also results in the breaking apart of habitat, or habitat fragmentation, and the creation of land cover types, referred to as the matrix or the non-habitat portion of a landscape. Since its inception in the 1980s, landscape ecology has led to major strides in understanding the impacts of human land conversion on biodiversity (Turner 2005). A central question in landscape ecology is to understand how landscape structure affects the abundance and distribution of species. Landscape structure is generally characterized by landscape composition (the number and amount of habitat and matrix cover types) and landscape configuration (the spatial arrangement of those cover types) (Turner 2005). Given that land conversion to agriculture and urban development is projected to dramatically increase in the next 50 years (FAO 2003), landscape ecologists are tasked with providing a scientific basis for land management and planning that will mitigate impacts to biodiversity.

Determining the relative effects of landscape structure variables is important to landscape ecology and conservation management (Lindenmayer and Fischer 2007, Smith et al. 2011). First, habitat loss, habitat fragmentation and changes to the surrounding matrix represent different landscape-scale processes and have distinct effects on biodiversity which may differ in magnitude and direction (McGarigal and Cushman 2002, Fahrig 2003), and second, they may be managed independently to some degree (e.g. decrease habitat fragmentation or increase quality of the surrounding matrix, for a given amount of habitat) (Lindenmayer and Fischer 2007). Therefore, assessing the

relative importance of these landscape factors allows the ranking of competing conservation management options by their potential benefit to biodiversity to effectively maximize the use of limited conservation resources (Wilson et al. 2007). However, because habitat loss, habitat fragmentation and changes to the surrounding matrix can all result from human land conversion, they are generally highly correlated, making it difficult to determine which has the larger effect on biodiversity. For example, landscapes with high habitat amount are generally unfragmented and surrounded by high-quality matrix. Therefore, careful study design, such that the landscape variables of interest vary independently, is needed in order to separate the effects of these landscape-scale processes on biodiversity (Fahrig 2003).

The relative effects of habitat loss and habitat fragmentation in landscape ecology have long been debated in landscape ecology. Reviews of over a decade of empirical research (Fahrig 2003, Smith et al. 2009) found that when fragmentation was measured independently of habitat amount (either by controlling for habitat amount in study design or statistically), habitat amount had a stronger, positive effect on biodiversity than fragmentation per se. The effect on biodiversity of fragmentation per se (i.e. configuration of habitat in a landscape, independent of habitat amount) was generally weak and was just as likely to be positive as negative (Fahrig 2003). Although these reviews were primarily based on temperate forest birds, several recent studies in less-studied regions (e.g. Mediterranean) and taxa (e.g. small mammals) confirm that habitat amount is more important than habitat configuration (Ritchie et al. 2009, Melles et al. 2011, Mortelliti et al. 2010 and 2011, With and Pavuk 2011), but there is still ongoing debate for the tropics (e.g. Cerezo et al. 2010, Zurita and Bellocq 2010) and some taxa

(e.g. mid- and large-sized mammals; Thornton et al. 2011a). Taken together, these results suggest that management of habitat configuration in a landscape (e.g. reduced habitat subdivision) is unlikely to mitigate the effects of habitat loss on biodiversity, and that conservation actions should focus on habitat protection and restoration.

The relative effects of habitat loss and the quality of the surrounding landscape matrix on biodiversity are less understood. In a recent review of empirical studies, Prevedello and Vieira (2010) found that the type of matrix surrounding habitat patches had a weaker effect on ecological responses compared to measures of habitat amount in a landscape in over half (56%) of the studies; in the remaining studies (44%), the effects of matrix quality were equivalent or greater than the effects of habitat amount. Studies since then have found disparate results for some matrix types. For example, the effects of road density in a landscape (index of poor-quality matrix) were considerably weaker than the effects of habitat amount for the richness and occurrence of temperate forest-breeding birds (Smith et al. 2011), but were considerably stronger for the abundance of temperate wetland-breeding amphibians (Veysey et al. 2011). Taken together, these results suggest that protection and restoration of habitat in landscape is generally more effective for conserving biodiversity than improvements to the surrounding matrix; however this may not be applicable to all species or species groups.

Given that habitat loss generally has the greatest impact on biodiversity relative to fragmentation per se and landscape matrix quality, it is important to landscape ecology and conservation management to identify those species or species groups whose populations are most negatively affected by habitat loss. Theory suggests that species responses to habitat loss are influenced by several life-history traits (reviewed in Henle et

al. 2004, Ewers and Didham 2006). Firstly, species with greater dispersal ability are predicted to be less sensitive to habitat loss because they are able to access more habitat in a landscape and thereby increase re-colonization rates (Baguette et al 2000). However, some empirical studies have found the opposite, that more mobile species are more sensitive to habitat loss, possibly because they incur higher dispersal mortality in the matrix (Gibbs 1998, León-Cortés et al. 2003, van Houtan et al. 2007). Similarly, species with larger territories or home ranges, and larger body size, should be more sensitive to habitat loss because these traits are highly correlated with dispersal distance (Sutherland 2000, Bowman et al. 2002, Bowman 2003). Secondly, species with high reproductive rates are predicted to be less sensitive to habitat loss because they will rebound more quickly from population declines (Gibbs and Shriver 2002) and a large number of offspring increases the number of potential colonists which increases colonization rates of unoccupied patches (Öckinger et al. 2012). Similarly, species with later sexual maturity and longer generation times are predicted to be more vulnerable to habitat loss since these traits are generally negatively correlated with reproductive rate. Moreover, species that occur at low natural densities, such as rare species, are predicted to be more sensitive to habitat loss because small populations are more susceptible to extinction via demographic stochasticity than large populations (Davies et al. 2000). Lastly, species at higher trophic levels (predators) or that have highly specialized niches (habitat specialists) are expected to be more sensitive to habitat loss than species at lower trophic levels (herbivores) or habitat generalists (Ewers and Didham 2006). Habitat generalists are predicted to be less sensitive to habitat loss because they are able to supplement resources using other cover types (Devictor et al. 2008). Species at higher trophic levels

are more vulnerable to habitat loss because they often have unstable population dynamics (Davies et al. 2000) and strong dependence on resource populations (Cagnolo et al. 2009).

Although there have been several empirical tests of these life-history predictions, these studies are limited for several reasons. First, the majority of studies were conducted at the patch scale (Cagnolo et al. 2009, Kennedy et al. 2011, Thornton et al. 2011a). Since habitat loss is a landscape-scale process (McGarigal and Cushman 2002), these predictions need to be tested at the landscape scale. Moreover, many of these studies were conducted within a narrow range of geographical locations or specific taxonomic groups (e.g. eastern US forest breeding birds, Vance et al. 2003). Finally, understanding the effects of life-history traits on species responses to habitat loss is often (unavoidably) confounded by correlations or synergistic interactions among the traits themselves. Overall, the predicted relationships between life-history traits and tolerance to habitat loss have not been generally validated at the landscape scale.

To investigate the relative effects of landscape structure variables on biodiversity and the role of life-history traits in determining species responses to landscape structure, I focused my dissertation on wetland-dependent vertebrates for several reasons. First, these organisms are undergoing the largest wildlife population declines worldwide (Millennium Ecosystem Assessment 2005). The primary driver of these declines is wetland habitat loss and fragmentation (Eddleman et al. 1998, Conway et al. 1994, Gibbons et al. 2000, Cushman 2006), with anthropogenic land-uses such as roads and development also implicated (Findlay and Houlihan 1997, De Luca et al. 2004 and 2008, Roe et al. 2006, Gibbs et al. 2005, Eigenbrod et al. 2008, Steen et al. 2012). Surprisingly,

however, these organisms are a relatively understudied ecological group in landscape ecology (McGarigal and Cushman 2002). In fact, a global survey of fragmentation studies by Gardner et al. (2007) found almost an order of magnitude more research for terrestrial species than wetland species.

Second, scientific recommendations for conservation management of wetland-dependent species have traditionally focused on wetland management (e.g. amphibians, Semlitsch 2000). Consequently, current conservation mitigation for these declining species relies on policies and regulations that generally focus on preserving or increasing wetland habitat in a region (Haig et al. 1998, Semlitsch and Bodie 2003, Bauer et al. 2010, MacPherson and Bright 2011). Although wetland loss undoubtedly impacts populations of wetland-dependent species, in a conservation context, it is not enough to know that wetland loss has an effect. We must also know how large this effect is in comparison with other landscape factors of population declines in these organisms (e.g. roads; Shriver et al. 2004, Rytwinski and Fahrig 2012); however, the relative importance of wetland loss with other landscape structure variables is generally unknown.

Finally, wetland-dependent species are generally considered to function as metapopulations, with demonstrated examples in mammals (Schooley and Branch 2009), birds (Foppen et al. 2000), amphibians (Werner et al. 2009) and reptiles (Cosentino et al. 2010). This is likely because wetlands exist as spatially discrete patches (Gibbs 2000, Marsh and Trenham 2001, Vermaat et al. 2008). Moreover, local wetland populations often undergo frequent extinctions (e.g. Skelly et al. 1999, Foppen et al. 2000, Schooley and Branch 2009), thereby requiring re-colonization through dispersal. Therefore, wetlands provide an appropriate system in which to test the relative effects of landscape

composition (habitat amount, matrix quality) and landscape configuration (habitat fragmentation per se), as well as the influence life-history traits, such as dispersal ability, on species responses to these landscape factors. Overall, there is a need to advance landscape scale research for wetland communities and provide a general synthesis on how wetland-dependent vertebrates respond to landscape structure to provide a basis for effective conservation management of these declining organisms (Semlitsch and Bodie 2003, Gardner et al. 2007).

The overarching goal of my dissertation was to advance our understanding of how wetland-dependent vertebrates respond to landscape structure and the life-history traits that determine these responses, and to provide conservation management recommendations to mitigate population declines of these species. In my second chapter, I addressed the question: *Do life-history traits influence wetland species responses to wetland habitat loss?* To overcome the limitations associated with existing empirical tests of life-history trait effects on species responses to habitat loss, specifically patch-scale inferences, limited taxonomic groups and geographic locations, I conducted a meta-analysis and combined results from 90 studies conducted worldwide that quantified the relationship between wetland habitat amount in a landscape and population abundance of at least one wetland-dependent vertebrate species. These 90 studies were from 16 countries and generated 426 effect sizes across 220 species, including mammals, birds, amphibians and reptiles. I converted the various test statistics reporting the relationship between wetland amount in a landscape and population abundance of a single species from each study into a common estimate of effect size (Pearson correlation coefficient, r). I then independently collected data on life-history traits for each species included in

the meta-analysis to test the prediction that species with greater mobility and/or higher reproductive rates are less sensitive to habitat loss than species with lower mobility and/or reproductive rates. I had initially hoped to include other life-history traits, including natural population density, trophic status and niche breadth; however there was too little information available for too few species, and these traits were therefore excluded.

My third chapter was an empirical study aimed to advance the understanding of the relative effects of landscape structure variables on two of the least studied taxonomic groups of wetland-dependent organisms, birds and turtles. In this chapter, I focused on eight wetland breeding birds and two freshwater turtles that are undergoing regional declines to ask the question: *What are the relative effects of habitat loss, habitat configuration and matrix composition on declining wetland species?* I assessed landscape structure using five variables: habitat loss and habitat configuration were measured as the proportion of wetland amount and number of wetland patches in a landscape, respectively, and matrix composition was measured as the proportion of forest cover, cropland cover and road density in a landscape. I conducted extensive field surveys over two field seasons (2009 and 2010) to determine the occurrence of my focal bird and turtle species in a set of landscapes selected to overcome common pitfalls in landscape ecology study design (Eigenbrod et al. 2011). I selected non-overlapping landscapes to minimize the correlation among these landscape variables, while maximizing the range in variation of these variables to the extent possible in the study area. I found that wetland amount was more important than the other landscape scale predictors of wetland breeding bird distribution, whereas forest cover, a measure of the landscape matrix surrounding a

wetland, was more important than the other measures of landscape structure for both turtle species.

In my fourth chapter, I investigated the generality of my empirical results in third chapter, and whether wetland species responses to landscape structure varied by taxonomic group. More specifically, I was interested in whether the amount of wetland in a landscape was more important than other measures of landscape structure for birds or any other wetland taxonomic group, and similarly whether the amount of forest in a landscape was more important than wetland amount for turtles or any other wetland taxonomic group, which led me to ask the question: *What are the relative effects of landscape-scale wetland amount and forest amount on wetland vertebrates?* I answered this question by conducting a second meta-analysis and building upon the data set of population-level responses of wetland vertebrates to wetland amount in a landscape that I generated from studies included in my Chapter 2 meta-analysis. I obtained information on forest cover around each wetland sampled in each of these studies to generate an effect size representing the population-level response to forest amount in the same landscape for each species. I then compared the strength of the relationship between species abundance and wetland amount to the strength of the relationship between species abundance and forest amount. I had initially hoped to include all studies from Chapter 2, but I could not obtain information on forest cover for all studies either because the study area did not contain forest (e.g. prairies), or the study did not measure forest cover and I could not obtain geographical coordinates from the authors to independently measure forest. I was able to extract forest amount effects sizes for 63 of the 90 studies included in Chapter 2 for 155 species across mammals, birds, amphibians and reptiles.

Chapter 2: Low reproductive rate predicts species sensitivity to habitat loss: a meta-analysis of wetland vertebrates.

Abstract

I tested the hypotheses that species with greater mobility and/or higher reproductive rates are less sensitive to habitat loss than species with lower mobility and/or reproductive rates, by conducting a meta-analysis of wetland vertebrate responses to wetland habitat loss. I combined data from 90 studies conducted worldwide that quantified the relationship between wetland amount in a landscape and population abundance of at least one wetland species to determine if mobility (indexed as home range size and body length) and annual reproductive rate influence species responses to wetland loss. When analyzed across all taxa, animals with higher reproductive rates were less sensitive to wetland loss. Surprisingly, I did not find an effect of mobility on response to wetland loss. Overall, wetland mammals and birds were more sensitive to wetland loss than were reptiles and amphibians. My results suggest that dispersal between habitat patches is less important than species' reproductive rates for population persistence in fragmented landscapes. This implies that immigration and colonization rate is most strongly related to reproduction, which determines the total number of potential colonists.

Introduction

Habitat loss is the primary threat to biodiversity worldwide (Vié et al. 2009), but species show wide variation in their responses to habitat loss. This variation is often attributed to differences in species traits (Henle et al. 2004, Ewers and Didham 2006). However, most studies evaluating species responses to habitat loss have measured habitat amount as patch size (Prugh et al. 2008), rather than evaluating the effects of habitat loss over the landscape (i.e. landscape scale study). In addition, most are limited to a narrow range geographical locations (Thornton et al. 2011a). Moreover, understanding the effects of species traits on species responses to habitat loss is often (unavoidably) confounded by correlations or synergistic interactions among the traits themselves (Meyer et al. 2008). Therefore, we still do not know, in general terms, why some species or species groups are more sensitive to habitat loss than others (Thornton et al. 2011a, Vetter et al. 2011, Newbold et al. 2012).

Dispersal ability is generally considered an important species trait influencing species response to habitat loss. Species with greater movement ranges are predicted to have higher colonization rates because they are able to access more habitat in a landscape (Baguette et al. 2000). Moreover, since colonization rates are assumed to be correlated to immigration rates, local extinction probability is predicted to be lower for species with higher movement ranges, due to rescue of populations from low numbers by immigration. Metapopulation studies therefore typically assume that more mobile species should be less susceptible to habitat loss than less mobile species (Hanski and Thomas 1994, Tscharrntke et al. 2002, Uezu et al. 2005, Holland and Bennett 2009). However, some empirical studies have found the opposite, that more mobile species are more sensitive to

habitat loss, possibly because they incur higher dispersal mortality (Gibbs 1998, León-Cortés et al. 2003, van Houtan et al. 2007). Therefore, the general relationship between mobility and tolerance to habitat loss is not clear.

Reproductive rate could also influence species responses to habitat loss, but it is not often considered, at least in empirical studies. Simulation studies suggest that reproductive rate has a much larger effect on the amount of habitat required for population persistence than the per capita rate of emigration (Fahrig 2001) or dispersal ability (With and King 1999); lower reproductive rates are predicted to increase the amount of habitat required for population persistence. To my knowledge, there are only two empirical tests of this prediction in which habitat amount is measured at a landscape scale. Vance et al. (2003) found that forest bird species with lower reproductive rates require more habitat for a 50% probability of occurrence than do forest birds with higher reproductive rates. Similarly, Holland et al. 2005 found a negative association between reproductive rate and minimum habitat amount required for presence across a group of dead wood boring beetle species. These studies suggest that species with lower reproductive rates require more habitat in a landscape for population persistence than do species with higher reproductive rates. This is likely because for a given amount of habitat in a landscape, species with higher reproductive rates will rebound more quickly from population declines than species with lower reproductive rates. Moreover, a large number of offspring increases the number of potential colonists which increases colonization rates of unoccupied patches (Öckinger et al. 2010). Therefore, I predicted that species with lower reproductive rates are more sensitive to habitat loss than species with higher reproductive rates.

The objective of this study was to determine the importance of mobility and reproductive rate in determining the responses of wetland species to wetland habitat loss. I used a meta-analytical approach to quantitatively synthesize the results of 90 studies conducted across the world that quantified the relationship between wetland amount in a landscape and wetland animal abundance. From these I obtained 426 responses to wetland loss for 220 wetland species including mammals, birds, reptiles and amphibians. I selected wetlands and wetland species for several reasons. First, wetlands are generally thought to function as metapopulations (Schooley and Branch 2009, Cosentino et al. 2010, Heard et al. 2012) and thus they provide an appropriate system to test whether mobility drives species responses to habitat loss. Moreover, wetland species are undergoing the largest wildlife population declines worldwide, primarily due to habitat loss (Millennium Ecosystem Assessment 2005), but are a relatively understudied ecological group in landscape ecology. Therefore, there is a need to provide a general synthesis on how wetland species are responding to wetland habitat loss, to prioritize landscape-scale conservation action. For example, if mobility is driving the response to habitat loss, the conservation focus should be on facilitating movement during high dispersal events or increasing connectivity in landscapes. In contrast, if reproductive rate is driving the response, the focus should be on supporting critical reproductive stages. To my knowledge, this is the first meta-analysis of population responses to habitat amount at the landscape scale (previous reviews were conducted at the patch scale), across taxa, and including the relationship to species' mobility and reproductive rate.

Methods

Study selection criteria

I measured wetland loss as the amount of wetland in a landscape. I included studies that measured wetland amount as the percent of wetland area in a landscape (area-based buffers) or wetland connectivity (or isolation). I searched for studies that quantified the relationship between the amount of wetland in a landscape and population abundance of at least one wetland species in the Web of Science and ProQuest dissertation and theses databases on 01 December 2011 using the following keyword string: (wetland* OR marsh* OR swamp* OR pond*) AND (amount OR area OR isolation* OR fragmentation) AND (amphib* OR turtle* OR reptile* OR mammal* OR bird*) AND (abundance* OR occurrence* OR occupancy* OR distribution) AND (species OR population*) AND (landscape*). No restriction on date was used. I limited my analyses to empirical studies that were conducted in wetlands, including natural wetlands (e.g. pond, marsh) and artificially created wetlands (e.g. stormwater basins, rice fields). For all studies, I assumed the authors accurately selected wetland habitat for each species; however if wetland amount at the landscape scale included land cover types other than wetlands (e.g. lakes), I contacted authors for clarification or excluded the study from analyses. I used a broad definition of “population abundance” to include population size (or relative abundance), population density (or relative density) and species occupancy (as an index of low vs. high abundance). I defined “wetland species” as any vertebrate (mammal, bird, reptile or amphibian) that uses wetlands as primary habitat for at least one part of its life cycle. I included species complexes that were fertile hybrids (e.g. *Pelophylax esculentus*) or two species that could not be distinguished (e.g. larval stages

of *Ambystoma* spp) as one species in the meta-analysis. During my literature search, I recorded the number of articles identified and the number of studies included and excluded according to the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) Statement (Figure 2.1).

Data extraction

In my meta-analysis, an effect size represents the quantitative relationship between the amount of wetland in a landscape and population abundance for a given wetland species. To extract an effect size from each study, I first searched the paper for a test statistic for the effect of wetland amount on animal abundance, and/or summary statistics (e.g. mean and variance), and corresponding sample size that could be converted into an effect size. When these values were not reported, I calculated them using raw data if they were provided in the paper, or I could extract them from figures using GetData Graph Digitizer 2.25 (Fedorov, S. 2012, internet free software), or I could obtain them from the authors. When a single study reported results for more than one species, I entered each species effect size as an independent estimate. When a study combined abundance or occurrence data across species, such that values for individual species could not be extracted, I contacted the authors for raw data or excluded the study. I did not calculate an effect size for 36 (of 256) species that occurred in $\leq 10\%$ of sampled landscapes or locations in each study, because effect size estimation is biased when there is a high proportion of absences (Rücker et al. 2009). While this may have resulted in exclusion of species sensitive to habitat loss, in the absence of sufficient sample sizes I cannot include them in the analyses. When a single study presented more than one effect

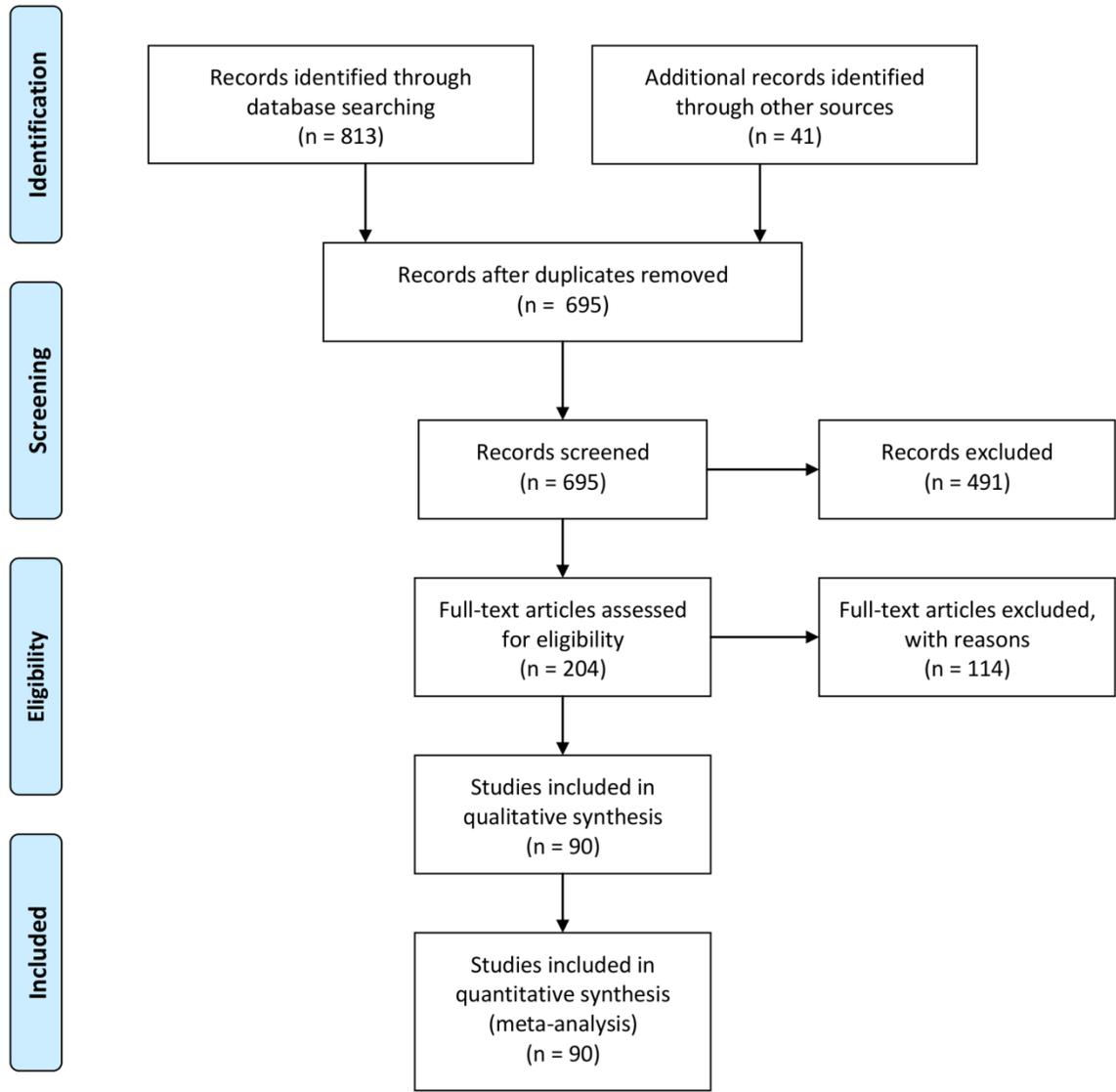


Figure 2.1: PRISMA literature search flow diagram.

size for a given species such that different effect sizes representing responses of the same species to different wetland types, I averaged these estimates across wetland types to extract a single effect size for that species, to avoid non-independence (4 studies). When a single study presented data in multiple years using the same study design, I averaged estimates across years for continuous data or tallied the numbers of years present for occupancy data. When studies presented effects of wetland amount at multiple landscape scales (buffer sizes), I selected the largest estimate, on the assumption that this scale was closest to the scale at which wetland loss best predicts the species' response (i.e. scale of effect, *sensu* Jackson and Fahrig (2012)).

Study design moderators

I identified three study design moderators to test if differences in study design influenced the magnitude and direction of the effect size, and to statistically control for such effects in remaining analyses (Table 2.1). First, the effect size might vary depending on how wetland amount in a landscape was measured. I combined several measures of wetland amount, including simple area-based buffers and nearest-neighbour distances to more complex connectivity indexes based on the incidence function model, because these measures have been shown to be highly correlated and have similar performance in predicting ecological responses (Moilanen and Nieminen 2002, Bender et al. 2003, Tischendorf et al. 2003, Prugh 2009, Ranius et al. 2010, Thornton et al. 2011b, Martin and Fahrig 2012). These comparative studies also suggest that measures with more information about the amount of (occupied) habitat in the landscape are better predictors, and therefore I expected a priori studies using such measures would have

Table 2.1: Study design and species trait moderator variables used in the meta-analysis.

Moderator Type	Moderator Variable	Category or Range	Description
Study Design	Study Type	Amount-based	Wetland amount was measured as the percent wetland area in a landscape or buffer (area-based buffers)
		Configuration-based	Wetland amount was measured as the number of wetland patches in a landscape, or using wetland isolation (nearest-neighbour distances) or connectivity (incidence function model) metrics
	Sampling Effort	Area-dependent	Sampling effort increased in proportion to the sampled wetland area
		Area-independent	Sampling effort was consistent across sampled wetlands
		Unknown	Sampling effort was unknown
	Patch Area	Included	Sampled wetland area was included in the calculation of wetland amount in the landscape
Not included		Sampled wetland area was not included in the calculation of wetland amount in the landscape	
Species Trait	Home Range Size	0.001 - 35600	Mean annual home range or territory size (ha)
	Body Length	2.5 - 1200	Mean body length across both sexes (cm)
	Reproductive rate	2 – 16000	Mean litter or clutch size multiplied by the mean number of litters or clutches per year

larger effect sizes. I distinguished two study types, 1) amount-based studies, where wetland amount was calculated as the percent wetland area in a landscape or buffer surrounding the sampled wetland patch, or 2) configuration-based studies, where the configuration of wetland habitat was included in the calculation of the measure, such as the number of wetland patches in a landscape, nearest-neighbour distances, wetland proximity or wetland connectivity. For all study types, I applied the convention that each effect size extracted from a study should represent the population response of a species to increasing wetland amount in a landscape. However, for nearest-neighbour studies, a negative effect of increasing distance indicates that a species responded positively to closer wetlands, or equivalently, greater wetland habitat amount within the surrounding landscape. Therefore, I reversed the sign of the effect sizes extracted for nearest-neighbour studies to make them comparable to those extracted for all other studies representing the response to increasing wetland amount (Gurevitch and Hedges 1993).

Second, the relationship between sampling effort used to measure population abundance of a given species and the size of wetlands may influence the effect size observed. Studies where sampling effort increased in proportion to wetland size will observe a positive relationship between wetland size and abundance, simply because more area is searched in larger wetlands. If wetland patch size is positively correlated with total wetland amount in a landscape, this will inflate the effect of wetland amount because a greater amount of wetland was sampled in landscapes containing more wetland. Therefore, I categorized studies by the sampling effort as, 1) area-independent, where sampling effort was consistent across sampled wetlands, 2) area-dependent, where sampling effort increased in proportion to the wetland area, or 3) unknown, where the

sampling effort was unknown and could not be obtained by contacting authors. When a study used a combination of more than one of these methods, I selected the sampling method that accounted for the majority of the data.

Third, the effect of wetland habitat amount in a landscape could vary depending on whether the sampled wetland was included in the calculation of wetland area in a landscape (wetland amount). Prugh (2009) found that measures of habitat amount in a landscape, including area-based buffers, nearest-neighbour distances and connectivity, were better predictors of occupancy when focal patch area was included in the model. Therefore, I expected a priori that studies that did not include the sampled wetland patch area in wetland amount would have a lower effect size compared to studies that did include the sampled wetland patch area.

Effect size calculations

I selected the Pearson correlation coefficient, r , as my common estimate of effect size. When a study reported Spearman's rank correlation coefficient (ρ), I converted ρ to r following Lajeunesse (2013). If studies did not report a correlation coefficient, I transformed published test statistics as follows. For studies with continuous measures of population abundance, I extracted r values by taking the square root of reported R^2 values from univariate linear regressions, and adding the sign of the slope. Note that I did not use partial R^2 values (Hullett and Levine 2003). When raw data were available, I calculated r for species with occupancy rates ≥ 0.7 . Data sets with occupancy rates ≤ 0.7 did not meet normality assumptions of r ; in this case I converted continuous data to occupancy data to determine an effect size. For studies that measured species occupancy or reported means and variances between two groups (e.g. mean wetland amount in

occupied vs. unoccupied landscapes), I first calculated the standardized mean difference (ES_{sm}) following (Gurevitch and Hedges 1993). I then converted each ES_{sm} to r following (Borenstein et al. 2009). I transformed all correlation coefficients to Fisher's z -scale (ES_{Zr}) following (Borenstein et al. 2009).

The next step was to obtain accurate and comparable sample sizes across studies. Meta-analysis weights each study by its inverse variance, based on the assumption that studies with greater precision provide a more accurate estimate of the true effect. The variance of ES_{Zr} is approximated as $V_{Zr} = 1/(n-3)$, where n is the total sample size of the study (Borenstein et al. 2009). This gives more weight to studies with larger sample sizes; however, this may overweight studies with pseudoreplication, a common problem in landscape ecology (Eigenbrod et al. 2011). For example, within a given study area, studies that selected spatially independent wetlands in non-overlapping landscapes at a landscape size (i.e. scale) based on a species' biology may have a lower apparent sample size than studies that sampled as many wetlands as possible without consideration of spatial independence. In the latter, the sample size would be inflated due to non-independence of sample points because almost identical (i.e. overlapping) landscapes were incorrectly used as multiple independent observations (Eigenbrod et al. 2011). In addition, if neighbouring species sample points are closer together than the movement range of an individual, the same individual may be sampled more than once, again leading to violation of the assumption of independence. Therefore, I assessed the sample size of each study for pseudoreplication using process similar to that in Rytwinski and Fahrig (2012), as follows. My assessment was based on the assumption that each data point should represent a spatially independent sample. I assumed that an independent

sample was equivalent to an independent individual in a spatially independent sampling location, such that it was unlikely the same individual was sampled at more than one wetland. I considered studies to have independent samples, and therefore accurate sample sizes, in two situations. First, studies that selected non-overlapping landscapes a priori based on the movement range of the species were assumed to be independent samples because it is unlikely that the same individual would be sampled in neighbouring sites. Second, in studies where each landscape represented the area around a sampled individual (e.g. a nest), the number of landscapes was already equivalent to the number of independent individuals. For all other studies, I adjusted sample size. When the distance between two sampling locations (e.g. wetlands) was less than the linear home range or territory size of the species, the two locations were counted as a single sample. For studies that compared population abundance in sampling locations to randomly selected locations where the species was known to be absent, and spatial information on these random locations was not available, the sample size was the number of spatially independent sampling locations plus one (to account for all random locations).

After determining the adjusted sample size ($n_{adjusted}$) of each study, I calculated the inverse variance weight for each ES_{Zr} as $w = n_{adjusted} - 3$ (Lipsey and Wilson 2001). Studies with $w < 1$ were excluded from the meta-analysis. Refer to Appendix A for studies included in the meta-analysis and associated country, species, effect sizes, adjusted sample sizes and study-design categories.

Species traits

My main objective was to test if mobility and reproductive rate could explain variation in species responses to wetland habitat loss (Table 2.1). I collected data on

these species traits from primary literature, theses and published species guides. I estimated mobility using two species traits, home range size and body size, which are both strongly correlated to dispersal distance across species of mammals and birds, independent of their migratory status (Sutherland et al. 2000, Bowman et al. 2002, Bowman 2003). Home range size for mammals was indexed as mean annual home range area (ha) (Bowman et al. 2003). For birds, the mean annual breeding territory size (ha) was used for species that forage primarily within their breeding territory (Bowman 2003). For species that travel away from the nest site to forage (e.g. great blue heron, *Ardea herodias*), home range or foraging distance was reported in the literature rather than breeding territory. In these cases, mean annual home range area (ha) during the breeding season was used as a measure of territory size (Bowman 2003). When annual home range estimates were not reported, I used the mean foraging distance as a diameter to calculate a circular home range area (ha). For studies conducted during the non-breeding season (e.g. migration, overwintering), I used territory size or foraging distances during the same season when available. For reptiles (turtles and snakes) and amphibians (anurans and salamanders), a cross-species relationship between either home range or body size and dispersal distance has not yet been demonstrated (Semlitsch 2008). Therefore, I used adult annual movement ranges, which represented seasonal migration distances between breeding and summer (foraging) or overwintering sites (Semlitsch 2008). This is often called a home range, but is conceptually different than a breeding home range or territory in mammals or birds. However, it is the only measure of space use that I could consistently obtain from the literature across all species of reptiles and amphibians. I assumed that species with larger adult annual movement ranges are more

mobile in general and thus have larger dispersal distances than species with smaller adult annual movement ranges. Moreover, since body size is generally correlated with dispersal distance across several vertebrate taxa (including amphibians; Jenkins et al. 2007), it is also reasonable to assume that larger bodied reptile and amphibian species disperse greater distances than smaller bodied species (Perry and Garland 2002; but see Lemckert 2004). Therefore, for reptiles and amphibians, I used mean annual home range area (ha) as an index of mobility, preferably estimated using the minimum convex polygon method (Row and Blouin-Demers 2006). When minimum convex polygon estimates were not available, I estimated home range as a circular area (ha) with diameter equal to the mean seasonal migration distance (Semlitsch 2008). All home range estimates were averaged between the sexes. Body size was measured as body length (mean body length across both sexes in centimeters); body mass was not used as it was unavailable for 37 % of species (primarily amphibians), which represented half of the effect sizes included in the meta-analysis. I estimated reproductive rate as the mean litter or clutch size multiplied by the mean number of litters or clutches per year. Species traits were taken from sources as close to the study region of each study as possible. When trait data were available over multiple years, I used the mean value. Where studies reported a range in values, I used the mid-point of these ranges. For species complexes that were comprised of two species that could not be distinguished, I took the average of the traits of the species comprising the complex.

I classified effect sizes by taxonomic group at the class level: mammal, bird, reptile and amphibian, and by order within each taxonomic group. Since there were only

two orders within amphibians, I also classified effect sizes by family for amphibians. Refer to Appendix B for life-history trait values and for information sources.

Meta-analysis

I first conducted a random-effects meta-analysis using the DerSimonian-Laird method to determine the summary weighted-mean effect size of the overall response of wetland species abundance to wetland loss at the landscape scale. Under the random-effects model, the weight assigned (w^*) to each effect size is the inverse of the sum of two variance components $w^* = 1/(w + T^2)$, where w (see above) is the unique sampling variance for each study (within-study error) and T^2 is the pooled variance of the true effects across all randomly selected studies (between-studies variance (Borenstein et al. 2009). I also calculated the heterogeneity in true effects (Q statistic), which I compared against a chi-square distribution, to test whether the total variation in observed effect sizes (Q_T) was significantly greater than that expected from sampling error (Q_E). I then tested if moderator variables can explain true variation in the effect sizes (Q_M), i.e. $Q_T = Q_M + Q_E$. All analyses were conducted using the ‘metafor’ package (version 1.7-0) in R 3.0 (R Development Core Team 2013).

To test whether mobility, reproductive rate and/or study-level moderators explained a significant amount of heterogeneity in effect sizes, I performed a mixed-effects meta-regression using restricted maximum-likelihood estimation of heterogeneity. Since mobility and reproductive rate information was not available for all species, I removed all effect sizes with missing species trait data to have equal datasets. All species traits were log-transformed to meet test assumptions. I first tested if study design or

taxonomic moderators influenced the effect sizes by performing univariate mixed-effects meta-analysis where, if study design or taxonomy explained significant heterogeneity in the effect sizes, I would then subset my data by that moderator variable to control for the effect of study design or taxonomy in analyses of the effects of mobility and reproductive rate. I then performed univariate mixed-effects meta-regression for home range, body length and reproductive rate.

I assessed publication bias by a rank correlation (Kendall's tau) test of the relationship between ES_{Z_r} and n in association with visual inspection of a scatterplot between these two variables following Jennions et al. (2013).

Results

Although I found more than 200 studies that examined the effect of wetland habitat amount in a landscape on population abundance of wetland vertebrates, only 90 studies met the inclusion criteria (Figure 2.1). These 90 studies were published from 1991 to 2013, and generated 426 effect sizes across 220 species and 16 countries (Appendix A). Studies were predominately from North America (60) and Europe (19), with remaining studies from Australia (5), Central and South America (3), Asia (2) and Africa (1). After removing effect sizes due to lack of information on mobility or reproductive rate, the total number of effect sizes was reduced to 334 across 137 species. The summary weighted-mean effect size from a random-effects meta-analysis across all taxa was 0.11 (95% CI: 0.089, 0.137; $n = 334$), indicating an overall weak, positive effect of wetland amount in a landscape on wetland animal population abundance. The overall heterogeneity was $Q = 712.03$ ($p < 0.0001$), indicating highly significant variation in

species responses to wetland amount. There was no strong evidence of publication bias as there was a weak relationship between effect size and sample size (Kendall's tau = 0.03, $p = 0.36$), and a scatterplot between these two variables showed effect sizes were symmetrically distributed around the summary effect and produced a funnel-shape with greater variation in studies at low sample sizes (Figure 2.2).

Mixed-effects meta-analysis across all taxa ($n = 334$) showed that no study design moderator explained any significant heterogeneity in the effects (study type: $Q_M = 0.65$, $p = 0.42$; sampling effort: $Q_M = 4.34$, $p = 0.11$; sampled wetland area: $Q_M = 1.26$, $p = 0.26$; Appendix C). Therefore I did not control for study design in analyses of the influence of mobility and reproductive rate on species responses to habitat amount. Reproductive rate and body length explained significant heterogeneity in the effects of wetland amount on animal population abundance across all taxa ($Q_M = 18.83$, $p < 0.0001$; $Q_M = 16.02$, $p < 0.0001$, respectively; Appendix C). Since the correlation between reproductive rate and body length was high ($r = -0.72$, $p < 0.0001$, $n = 334$), I performed a multiple meta-regression to test for independent effects of each moderator while accounting for the presence of the other ($Q_M = 20.76$, $p < 0.0001$; Appendix C). After controlling for the effect of body length, reproductive rate was negatively related to the effect of wetland amount on population abundance ($ES_{Zr} = -0.03$; 95% CI: -0.061, -0.002; $p = 0.03$; Figure 2.3). In other words, species with lower reproductive rates were more sensitive to wetland amount in a landscape than species with higher reproductive rates. In contrast, after controlling for the effect of reproductive rate, the effect of body length was not

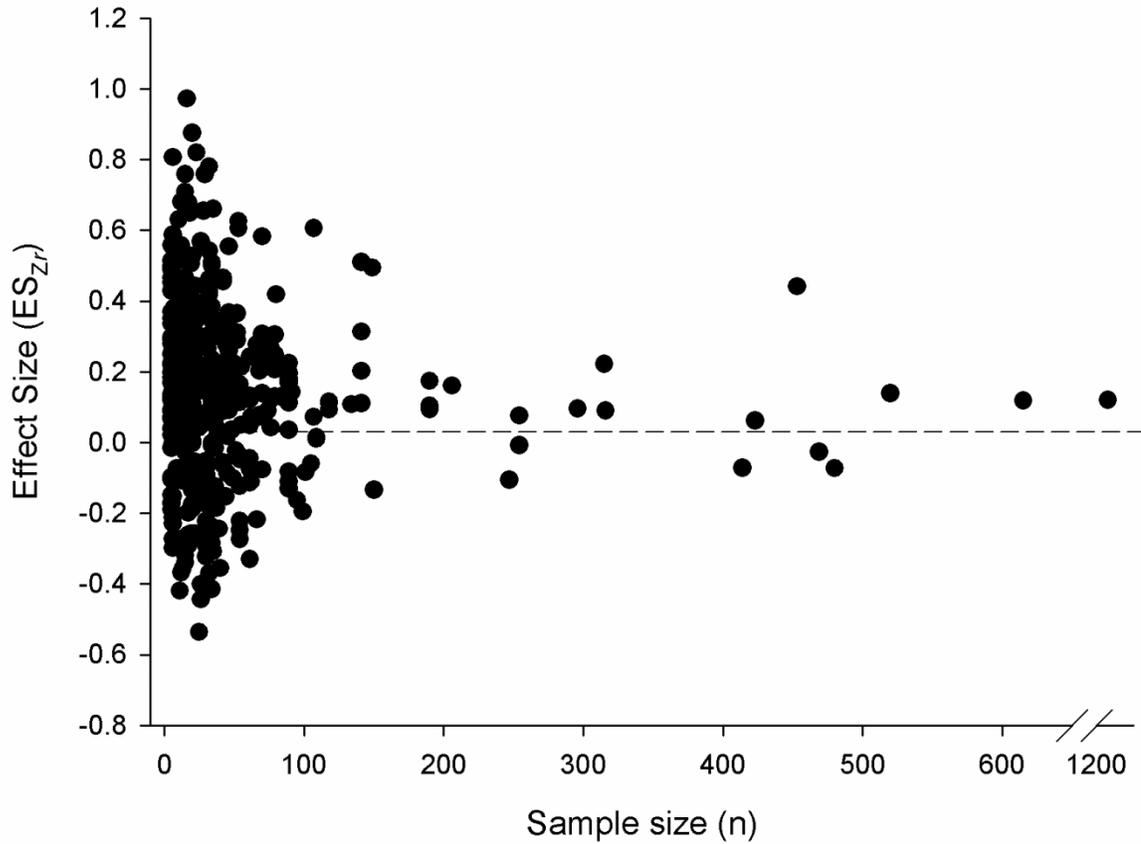


Figure 2.2: Relationship between z -transformed correlation coefficients (ES_{Zr}) and sample size (n) to assess publication bias. Dashed line is the summary mean-weighted effect size from random-effect meta-analysis across 426 effect sizes from 90 studies. There was no strong evidence of publication bias since effect sizes were symmetrically distributed around the summary effect and produced a funnel-shape with greater variation in studies at low sample sizes.

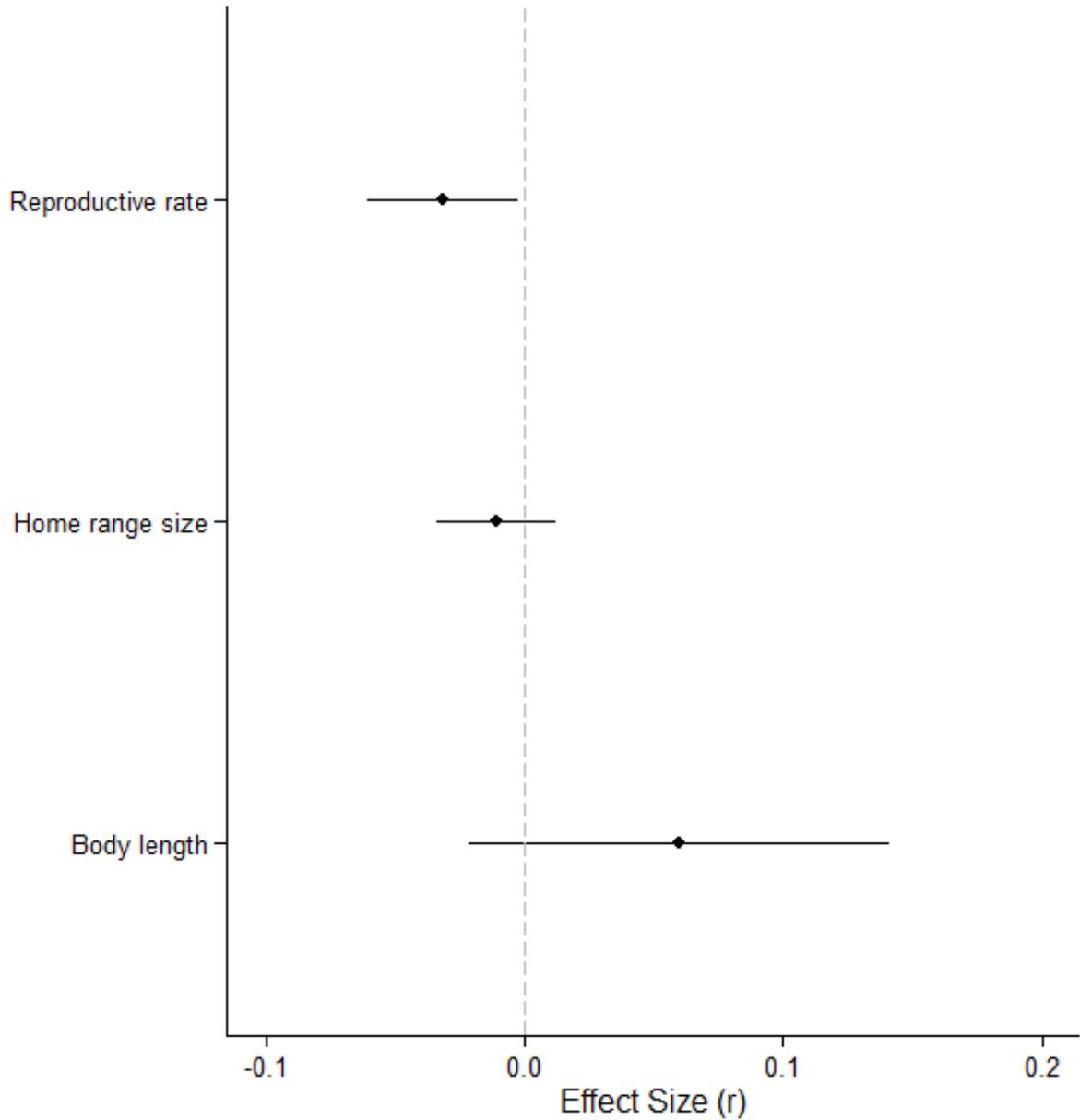


Figure 2.3: Effects of reproductive rate and mobility (indexed as home range size and body length) on population response of wetland vertebrates to wetland habitat amount in a landscape, including mammals, birds, reptiles and amphibians (n = 137 species). Points represent mean-weighted effect sizes (z-transformed correlation coefficients) from mixed-effects meta-regression and lines indicate 95% confidence intervals.

significant ($ES_{Zr} = 0.06$; 95% CI: -0.025, 0.137; $p = 0.15$; Figure 2.3).

Home range size did not explain any significant heterogeneity in effect sizes ($Q_M = 0.54$, $p = 0.46$, $n = 334$; Appendix C). The correlation between home range and reproductive rate was $r = -0.06$, $p = 0.26$, and between home range and body length was $r = 0.25$, $p < 0.0001$. Since home range did not explain significant heterogeneity and home range size information was missing for 92 of the total 426 effect sizes extracted across all taxa, I removed home range and re-analyzed the effect of body length and reproductive rate with a larger dataset ($n = 421$). The multiple meta-regression containing body length and reproductive rate explained a significant amount of heterogeneity in the effect sizes of the larger dataset ($Q_M = 25.38$, $p < 0.0001$). Consistent with the analysis with 334 effect sizes (above), reproductive rate was negatively related to the effect of wetland amount on population abundance ($ES_{Zr} = -0.03$; 95% CI: -0.061, -0.007; $p = 0.02$) and body length had no significant effect ($ES_{Zr} = 0.06$; 95% CI: -0.021, 0.134; $p = 0.16$). Refer to Appendix D for descriptive statistics of species traits across taxa.

The effect of the amount of wetland habitat in a landscape on animal abundance varied by taxonomic class ($Q_M = 25.17$, $p < 0.0001$; Appendix C). The weighted-mean effect size for mammals and birds was greater than that of reptiles and amphibians (Figure 2.4). The correlation between measures of mobility and reproductive rate differed across taxa, with negative correlations for mammals and birds, and positive correlations for reptiles and amphibians (Table 2.2). Since there was large heterogeneity of effect sizes within each taxon separately (birds: $Q = 168.96$, $p < 0.0001$, $n = 115$; reptiles: $Q = 35.32$, $p = 0.048$, $n = 24$; amphibians: $Q = 412.24$, $p < 0.0001$, $n = 189$), I

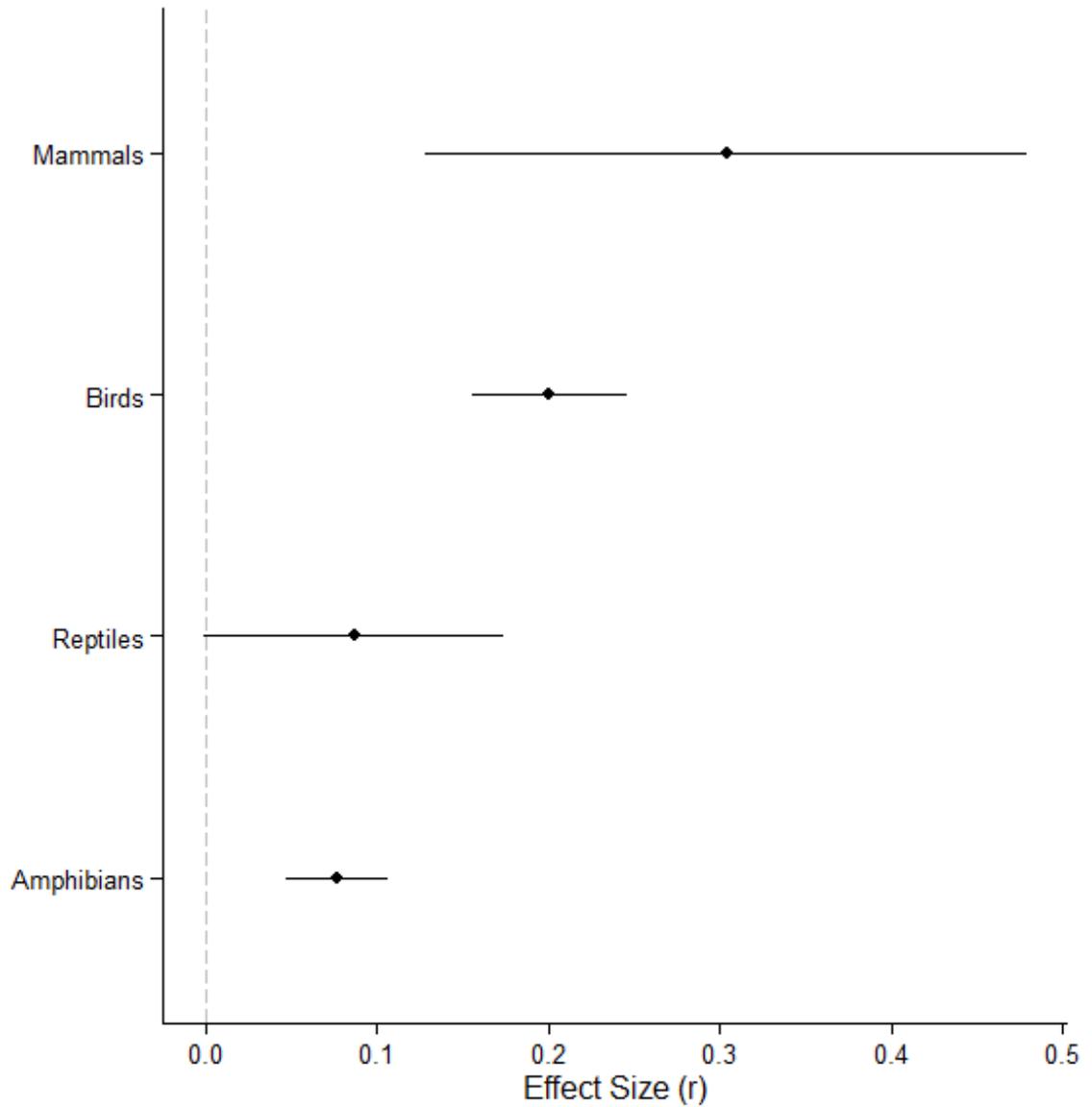


Figure 2.4: Population-level responses of wetland vertebrate classes to wetland habitat amount in a landscape. Points represent mean-weighted effect sizes (z -transformed correlation coefficients) from mixed-effects meta-regression and lines indicate 95% confidence intervals.

Table 2.2: Pearson correlation coefficients (above diagonals) and associated p-values (below diagonals and italicized) between species traits within each vertebrate class.

Taxa	# of effect sizes	Species Trait	log (home range)	log (body length)	log (reproductive rate)
Mammals	6	log (home range)		-0.75	0.80
		log (body length)	<i>0.06</i>		-0.75
		log (reproductive rate)	<i>0.08</i>	<i>0.09</i>	
Birds	115	log (home range)		0.40	-0.55
		log (body length)	<i><0.001</i>		-0.35
		log (reproductive rate)	<i><0.001</i>	<i><0.001</i>	
Reptiles	24	log (home range)		0.20	0.30
		log (body length)	<i>0.34</i>		0.55
		log (reproductive rate)	<i>0.15</i>	<i>0.01</i>	
Amphibians	189	log (home range)		-0.02	0.53
		log (body length)	<i>0.78</i>		0.07
		log (reproductive rate)	<i><0.001</i>	<i>0.35</i>	

tested for effects of mobility and reproductive rate within taxa. Refer to Appendix D for descriptive statistics of species traits for each taxonomic group.

For mammals, there were too few effect sizes ($n = 6$) to meaningfully test for effects of species traits. For birds, differences in study design did not any explain any significant heterogeneity in effect sizes (study type: $Q_M = 0.85$, $p = 0.36$, sampling effort: $Q_M = 0.12$, $p = 0.94$, sampled wetland area: $Q_M = 0.49$, $p = 0.45$; Appendix C). Bird effect sizes did not vary significantly by Order ($Q_M = 9.89$, $p = 0.20$; Appendix C). Reproductive rate and body length explained significant heterogeneity in the effects of wetland amount on population abundance of birds ($Q_M = 6.09$, $p = 0.014$; $Q_M = 3.86$, $p = 0.049$, respectively; Appendix C). A multiple meta-regression containing both body length and reproductive rate significantly explained heterogeneity ($Q_M = 7.75$, $p = 0.021$). Reproductive rate was negatively related to the effect of wetland amount on bird population abundance and the effect of reproductive rate was marginally significant ($ES_{Zr} = -0.25$; 95% CI: -0.506, -0.001; $p = 0.050$) while the effect of body length was non-significant ($ES_{Zr} = 0.12$; 95% CI: -0.056, 0.261; $p = 0.205$). I was able to test for the effect of body mass, which was moderately correlated with body length in my bird dataset ($r = 0.47$, $p < 0.001$). Body mass did not explain heterogeneity in effect sizes for birds ($Q_M = 0.315$, $p = 0.58$), nor did home range size ($Q_M = 0.052$, $p = 0.82$).

For reptiles (turtles and snakes), response to wetland amount did not vary significantly by Order ($Q_M = 0.66$, $p = 0.42$; Appendix C). Population-level effects of wetland amount on reptiles varied by study type ($Q_M = 6.10$, $p = 0.01$; Appendix C); amount-based studies (area-based buffers) had a lower mean-weighted effect size (0.05) than configuration-based (isolation or connectivity) studies (0.31). However, I could not

subset my data according to study type due to the small sample size ($n = 24$). In any case, in the non-subsetted data, neither mobility nor reproductive rate explained any significant heterogeneity in the effect of wetland amount (Appendix C). For amphibians (anurans and salamanders), population-level effects of wetland amount did not vary significantly by Order ($Q_M = 0.183$, $p = 0.668$), but responses varying by Family was marginally significant ($Q_M = 18.081$, $p = 0.054$; Appendix C). Therefore, I subsetted my data by Family and tested for effects of mobility and reproductive rate within the largest subset, which was the Family Ranidae ($n = 62$). Within the Ranidae subset, no study design moderator or species trait explained significant variation in the effect of wetland amount (Appendix C). In the non-subsetted data, no study design moderator or species trait explained significant variation in the effect of wetland amount (Appendix C).

Discussion

Contrary to the widely-held assumption in metapopulation and landscape ecology, my results suggest that dispersal ability is not a useful predictor of species sensitivity to habitat loss, at least for wetland vertebrates. When analyzed across all taxa, I found no evidence to support the prediction that animals with greater mobility were less sensitive to wetland habitat loss than species with lower mobility, when measured as home range size or body length. Analyses within each taxon also showed no effect of home range size or body length on species responses to wetland loss, as well as no effect of body mass for bird responses. Bowne and Bowers (2004) point out that despite the putative importance of mobility, there is very little evidence to validate the relationship between

movement and population persistence. Consistent with my results, studies that compared the relative influence of mobility to other species traits found that home range or movement distances were weakly related to species responses to habitat loss (Holland et al. 2005, Meyer et al. 2008, Thornton et al. 2011a). Moreover, studies that find strong effects of mobility are generally based on univariate models using indirect indices of mobility (Öckinger et al. 2009). I found that body length explained heterogeneity in species responses to wetland loss in univariate models across all taxa and within birds, but the effect of body length was no longer significant when reproductive rate was controlled for. Consistent with my results, there is generally little empirical support for a relationship between body size and sensitivity to habitat loss in vertebrates (Swihart et al. 2003, Meyer et al. 2008, Thornton et al. 2011ab, Vetter et al. 2011, Newbold et al. 2012).

A possible reason for the apparent lack of influence of mobility on species responses to habitat loss is that the effect of mobility varies. While several empirical studies have shown that more mobile species are less sensitive to habitat loss (de Vries et al 1996, Öckinger et al. 2009), other studies have found the opposite, that greater mobility decreases tolerance to habitat loss (Gibbs 1998, León-Cortés et al. 2003, van Houtan et al. 2007). If my meta-analysis included some species with mobility that positively influenced response to wetland amount and other species with mobility that negatively influenced response to wetland amount, it is possible that the two response types canceled each other out and resulted in no overall effect of mobility. Alternatively, no effect of mobility could have also resulted from a non-linear relationship with species responses to habitat loss. Thomas (2000) found that butterflies with intermediate mobility were more vulnerable to habitat loss than butterflies with either low or high

mobility. However, a scatterplot of species mobility and species responses to wetland loss was highly scattered in my dataset, indicating a very weak relationship rather than a non-linear one (Appendix E).

A second possible reason for the apparent lack of influence of mobility on species responses to habitat loss could be that mobility varies widely with landscape structure, such that the relative rankings of species' mobility changes as the landscape changes (Fahrig 2007). For example, translocation experiments of two forest bird species in three different landscape types (forested, timber harvested, agricultural) showed that the relative ability of each species to move in a landscape changed depending on landscape context (Gobeil and Villard 2002). Ovenbirds (*Seiurus aurocapilla*; forest specialist) had greater return rates than white-throated sparrows (*Zonotrichia albicollis*; forest generalist) in forested landscapes, whereas the opposite was found in harvested or agricultural landscapes. Several studies have shown that species' movement distances vary depending on landscape context (Bender and Fahrig 2005, Hadley and Betts 2009, Kennedy and Marra 2010) and that movement is a product of both species traits and landscape structure (Neuschulz et al. 2013). For example, home range sizes of northern saw-whet owls (*Aegolius acadicus*; Hinam and St. Clair 2008) and elk (*Cervus elaphus*; Anderson et al. 2005) increased by an order of magnitude as the amount of forest cover increased in a landscape. Similar to the results of my meta-analysis, Ferraz et al. (2007) found no effect of dispersal ability on patch occupancy responses of 55 tropical birds to forest patch isolation. They suggest as a possible explanation that species dispersal abilities change in disturbed landscapes, such that mobility estimated in continuous habitat is not useful to predict occupancy parameters in human-dominated landscapes.

Fahrig (2007) suggested that species mobility cannot be estimated independently of landscape structure and that to test dispersal ability the landscape context should match the location where movement data were collected. I was unable to test mobility using home range estimates that matched the location of each study included in my meta-analysis since home range estimates are generally very limited.

Lastly, it is possible that I did not find an effect of mobility because of high uncertainty in dispersal estimates for vertebrates (Bowne and Bowers 2004). I attempted to reduce this problem by using two measures of mobility, home range area and body size, that are known to be highly correlated with dispersal distance (Sutherland 2000, Bowman et al. 2002, Bowman 2003, Jenkins et al. 2007). However, the error associated with such estimates is still likely higher than the error associated with estimates of reproductive rate. For example, home range estimates for some species included in the meta-analysis varied by two orders of magnitude both within and between populations (e.g. Schooley and Branch 2006, Roe and Georges 2008, Blomquist and Hunter 2009, Cardador et al. 2009). If true, this means I was *a priori* more likely to find effects of reproductive rate than mobility on species sensitivity to habitat loss. Furthermore, although home range size is strongly correlated with dispersal distance in mammals and birds, I assumed this relationship for reptiles and amphibians (see Methods). If the relationship is weaker for reptiles and amphibians than for mammals and birds, body size is a more uncertain measure of mobility for reptiles and amphibians.

Body size is commonly used to index sensitivity to habitat loss in animal taxa (Swihart et al. 2003, Meyer et al. 2008, Thornton et al. 2011a, Vetter et al. 2011, Newbold et al. 2012). However, the exact inference one can make from a cross-species

effect of body size ambiguous because body size is simultaneously correlated to several life-history attributes (reviewed in Henle et al. 2004). For example, in addition to its positive correlation with dispersal, body size is positively correlated with area requirements and trophic level, and these relationships occur indirectly through a negative correlation with natural abundances and population fluctuations (Henle et al. 2004). I had expected that body size would be the best predictor of wetland vertebrate response to habitat loss because it is an indirect measure of several life-history mechanisms. However, I found that, in wetland vertebrates, the effect of body size was no longer significant once I had controlled for the effect of reproductive rate.

My results provide support for the hypothesis that animals with lower reproductive rates are more negatively affected by habitat loss than are animals with higher reproductive rates. When analyzed across all taxa, reproductive rate was the main explanatory variable for population-level responses to habitat loss, and the effect remained strong when controlling for body size (Figure 2.3; Appendix C). This suggests that reproductive rate affects species response to habitat loss, independent of its correlation with body size. Other studies have also found a greater effect of reproductive rate than movement-related traits. Simulation studies found that reproductive rate has a much larger effect on the amount of habitat required for population persistence than the per capita rate of emigration (Fahrig 2001) and dispersal ability (With and King 1999). An empirical test of the relative effects of reproductive rate and mobility at the landscape scale (Holland et al. 2005) found a strong negative association between reproductive rate and minimum habitat required for a group of dead wood boring beetles, whereas the effect of emigration rate was no longer significant once reproductive rate was controlled

for. The only other empirical test of the effect of reproductive rate at the landscape scale that I have found, Vance et al. (2003), reported a strong negative cross-species relationship between reproductive rate and the amount of forest in a landscape for forest birds. In line with my results, a recent meta-analysis of road and/or traffic effects across the same vertebrate taxa found that reproductive rate explained a larger amount of variation in population responses of mammals and amphibians to roads than mobility (indexed as home range size) and body size (Rytwinski and Fahrig 2012). Both types of landscape change – increased road density or habitat loss – both result in the loss of individuals. Therefore, the mechanism linking the landscape change to reproductive rate is the same: higher reproductive rates compensate for increased mortality and reduce local extinction risk (Purvis et al. 2000).

In contrast, several patch scale studies have found no effect of reproductive rate on species response to patch size and isolation. In a large meta-analysis of patch occupancy of 785 animal species across several taxa, Prugh et al. (2008) found no effect of fecundity on species responses to patch area and isolation. Similarly, no effect of reproductive rate was found on patch occupancy rates of 25 mid- and large-sized Neotropical mammals in sites within fragmented compared to continuous forest landscapes (Thornton et al. 2011a) or on the number of islands occupied by five lizard species (Wang et al. 2009). However, consistent with my results, a meta-analysis of patch area effects on butterfly species richness showed a negative effect of reproductive rate (Öckinger et al 2010). Patch scale studies may not find an effect of reproductive rate if patch size is not correlated to habitat amount in the landscape. A greater amount of habitat in a landscape represents a greater number of potential colonists available to a

patch via the mass effect that is the net flow of individuals from high abundance areas to low abundance areas (Shmida and Wilson 1985). For given amount of habitat in a landscape, species with higher reproductive rates will on average produce more colonists than species with lower reproductive rates. This high influx of individuals (immigration) from surrounding habitat would increase local population size with increasing habitat in the landscape (Fahrig 2003). This mechanism would only occur in patch-scale studies if the amount of habitat in the landscape is positively correlated with patch size (Fahrig 2003).

On the other hand, the relationship between reproductive rate and species response to habitat loss could occur through a correlation between reproductive rate and another unmeasured variable. For example, reproductive rate was found to be highly correlated with habitat and diet breadth in mammals (Thornton et al. 2011a), and niche breadth (composite of habitat and diet) was found to have a greater influence on patch occupancy rates of mammals and amphibians than body size (Swihart et al. 2003). Similarly, feeding guild (index of diet) was most strongly related to fragmentation sensitivity in a review of Neotropical vertebrate (including mammals, birds, reptiles and amphibians) responses at the patch-scale compared to body size (Vetter et al. 2011). However, in both Swihart et al. (2003) and Vetter et al. (2011), reproductive rate was not tested. Interestingly, a meta-analysis by Newbold et al. (2012) found that both generation length (surrogate for annual reproductive rate), habitat and diet breadth were in the top AIC models ($\Delta AIC < 2$) explaining population-level responses of pan-tropical birds to surrounding landuse intensity. This suggests that the mechanism behind the effect of reproductive rate on population response to habitat loss, i.e. greater reproductive output

and potential colonists, may act independently of any correlation with niche breadth, at least for birds. I am unable to test for an effect of niche breadth due to the lack of detailed habitat and diet information for most species included in my meta-analysis.

I found that wetland mammals and birds were more sensitive to wetland amount in a landscape than were wetland reptiles and amphibians (Figure 3). This difference was also found by Prugh et al. (2008) in a meta-analysis on patch area effects. These results may reflect the fact that many wetland reptile and amphibian species require more than one habitat type in a landscape to complete their life-cycle (e.g. foraging, nesting, hibernation) to sustain populations (i.e. landscape complementation) (Pope et al. 2000). In fact, the amount of forest in a landscape was found to be more important than wetland amount or connectivity for the occurrence of wetland reptiles (Attum et al. 2007, Chapter 3) and many species of wetland-breeding amphibians have a strong, positive association with the amount of forest in a landscape (Guerry and Hunter 2002, Houlahan and Findlay 2003, Van Buskirk 2005). If access to or quantity of the complementary habitat is limited, local population sizes will be low despite high wetland amount in a landscape. In fact, reptiles and amphibians were found to be more susceptible to negative effects of roads than mammals and birds (Rytwinski and Fahrig 2012). Taken together, these results suggest that wetland reptiles and amphibians are less limited by the amount of wetland habitat in a landscape than are wetland mammals and birds. Other factors, such as landscape complementation or road mortality, may have stronger effects than wetland loss on abundance and distribution of wetland reptiles and amphibians.

I present a comprehensive, worldwide review of wetland vertebrate responses to wetland habitat loss at a landscape scale (Appendix A); however some limitations need to

be considered. First, although I included study level moderators to control for potential bias at the study level, effect sizes obtained are likely influenced by study area attributes and the scale selected for analysis. Prugh et al. (2008) found that patch area and isolation effects varied depending on the predominant land cover in a study area. While I did not find an effect of study area type (natural, agricultural, rural or urban) on responses to wetland amount ($Q_M = 4.63$, $p = 0.20$; Appendix C), I could not assess whether other landscape variables (e.g. roads) confounded the effect of wetland loss. Moreover, if studies that conducted analysis at multiple scales are more likely to find the scale of effect for a given species (Jackson and Fahrig 2012), then studies that selected only one scale of analysis may systematically have lower effect sizes. However, I did not find an effect of the number of scales selected by a study on species response to wetland amount ($Q_M = 1.48$, $p = 0.22$; Appendix C). At the review-level, although I attempted to include unpublished studies such as theses (4 studies) and government reports (2 studies) in my review, published literature was the primary data source in my meta-analysis, representing 93% of studies. While I did not find evidence of publication bias, my review may be biased towards species from geographical areas with high publication rates (North America, Europe). Lastly, since the majority of studies were conducted on birds and amphibians, my mammal and reptile results are less solid.

Conclusions

My synthesis shows that wetland habitat loss at the landscape scale has an overall negative effect on the population abundance of wetland vertebrates across many

taxonomic groups and landscape contexts worldwide. My results support the hypothesis that species with lower reproductive rates are more negatively affected by landscape-scale habitat loss than are species with higher reproductive rates. Surprisingly, I found no evidence that mobility influences species response to habitat loss. This implies that immigration and colonization rate is more strongly related to reproduction, which determines the total number of potential colonists, than it is to a species' intrinsic mobility. From a conservation management perspective, my results suggest that priority should be placed on species with low reproductive rates. Also, my results suggest that conservation plans for declining wetland species should focus on actions aimed at increasing reproductive output, such as providing artificial nesting substrates or managing local wetland variables that increase reproductive success of target species (e.g. hydroperiod (Veysey et al. 2011); vegetation structure (Rehm and Baldassarre 2007, Skidds et al. 2007)).

Chapter 3: Effects of habitat loss, habitat configuration and matrix composition on declining wetland species

Abstract

Worldwide declines in wetland birds and turtles are attributed to landscape-scale habitat loss, habitat fragmentation and anthropogenic land use. However, due to multicollinearity, the relative importance of these factors is largely unknown. I evaluated the relative effects of wetland amount, wetland configuration (measured as the number of wetland patches), and matrix composition (measured as the amount of forest, cropland and road density) on the occurrence of eight declining wetland bird species and two threatened freshwater turtles across 66 to 70 landscapes. I selected landscapes to minimize correlations among the landscape-scale predictors and to represent the range of variation in each predictor available in the study region. For wetland birds, I found that the amount of wetland at a landscape-scale was more important than the other landscape variables, whereas surprisingly for turtles, the amount of forest in the surrounding landscape was more important than the other landscape variables. Wetland configuration independent of wetland amount was not an important predictor of any species. This is the first study to assess the relative, independent effects of the landscape-scale factors thought to contribute to wetland bird and turtle declines. My results confirm that wetland loss is the primary landscape-scale factor of wetland bird declines, but suggest that forest loss may play a greater role in freshwater turtle declines than previously realized; minimizing forest loss will have the most positive outcome for freshwater turtle

conservation. Therefore, effective conservation planning requires a multi-taxa approach to meet landscape-scale requirements of all declining wetland fauna.

Introduction

Worldwide declines of wetland birds and turtles are attributed to wetland loss and fragmentation at a landscape-scale, with anthropogenic land-uses such as roads and development also implicated (Millennium Ecosystem Assessment 2005). However, the relative contributions of these landscape-scale predictors remain unclear. Wetland birds and turtles have received considerably less attention than other taxa (e.g. amphibians) in landscape ecology, and the effects of landscape structure on their abundance and distribution are less well understood (Joyal et al. 2001, Semlitsch and Bodie 2003, Attum et al. 2008, Tozer et al. 2010).

Habitat loss generally has strong negative effects on species distribution and abundance relative to weaker and variable effects of habitat fragmentation (Fahrig 2003). In wetland ecosystems, the loss of wetland habitat has strong negative effects (e.g. Naugle et al. 2001, Tozer et al. 2010). Strong negative effects of wetland isolation (e.g. Joyal et al. 2001, Shriver et al. 2004 Attum et al. 2008, Smith and Chow-Fraser 2010) are also reported for wetland birds and turtles, and these are often reported as fragmentation effects. However, estimating the separate effects of wetland loss and wetland fragmentation is difficult because they are typically strongly correlated (Fahrig 2003). Therefore, the current understanding of the relative importance of habitat loss and fragmentation for wetland birds and turtles is limited.

In addition to the loss and fragmentation of habitat, the composition of the intervening space between habitat (or matrix composition) can also influence species abundance and distribution (Fahrig 2001, Prugh et al. 2008). The amount of forest cover, agriculture and roads surrounding wetlands have all been suggested to affect wetland birds and turtles. Forest cover is generally expected to be a positive matrix element (Alsfeld et al. 2010). For turtles, upland forest surrounding wetland is important for movement and refugia (i.e. short-term inactivity; Buhlmann and Gibbons 2001). Farmland is generally expected to have negative impacts due to increased dispersal mortality (Saumure et al. 2007), reduced wetland quality from nutrient and pollutant runoff (Sterrett et al. 2011), and wetland infilling from sedimentation (Naugle et al. 2001). Lastly, roads generally have negative effects on wildlife populations (Rytwinski and Fahrig 2012), either due to mortality or road avoidance behaviour. However, these matrix composition variables can also be correlated with wetland loss and/or fragmentation (e.g. landscapes with high agriculture often have high wetland loss) or with each other (e.g. landscapes with low forest cover often have low road density).

Correlations among landscape predictors (multi-collinearity) may confound inferences about the effects of wetland loss, wetland fragmentation and/or matrix composition. For example, Findlay and Houlahan (1997) were not able to determine whether forest cover or road density was the main driver of landscape effects on wetland biota, due to the high correlation between these two variables. High multi-collinearity reduces statistical power and causes the estimation of regression coefficients to be highly error-prone, leading to variability in the estimated direction and magnitude of effects (Eigenbrod et al. 2011). A possible example is the wide range of reported effects of

forest cover on wetland birds (e.g. positive, Alsfeld et al. 2010; negative, Budd and Kremenetz 2010; no effect, Findlay and Houlihan 1997).

It may not be possible to avoid multi-collinearity altogether in landscape ecology studies because the underlying processes causing landscape patterns are often linked. However, a comparative mensurative experimental approach which requires *a priori* selection of landscapes can minimize these correlations, allowing the estimation of separate effects (Fahrig 2003). I hypothesize that multi-collinearity among landscape-scale variables has not yet been adequately addressed in previous investigations of the effects of landscape structure on wetland birds and turtles. Therefore the relative importance of the major landscape variables causing declines in these species is largely unknown.

While the literature to date suggests that any or all of wetland loss, fragmentation and matrix composition could explain declines in wetland birds and turtles, in a management context it is important to know their relative effects. Landscape-scale variables represent competing landscape-scale management options that could be applied independently of one another. Therefore, estimating their relative effects would help prioritize future conservation management action. For example, if wetland biota respond negatively to wetland fragmentation (independent of wetland loss) then wetland policies should focus specifically on conserving and restoring large wetlands. Because multi-collinearity can confound inferences about species responses to landscape structure and misguide management recommendations, studies with landscape planning or species recovery applications must disentangle the estimated effects of landscape-scale predictors.

My objective was to determine the relative effects of wetland loss (measured as wetland amount, i.e. the proportion of wetland area within a landscape), wetland configuration (measured as the number of wetland patches in the landscape) and matrix composition (measured as the amounts of each of forest, agriculture and road density in the landscape) on the occurrence of declining wetland birds and turtles. To do this I used a comparative mensurative experimental approach, wherein I sampled a set of landscapes specifically selected for low multi-collinearity among these landscape variables and to represent the range of variation in each variable in the study area. Previous studies typically refer to fragmentation as both the loss and breaking apart of habitat. In this study, I assessed habitat fragmentation per se (Fahrig 2003), i.e. an aspect of the spatial configuration of wetlands independent of wetland loss. To avoid confusion over this distinction, I use the term "wetland configuration" rather than fragmentation. I estimated species presence of declining wetland birds and turtles in 66 to 70 landscapes (depending on the species group), varying in wetland amount and configuration, and in matrix composition.

Methods

Study Area

The study was conducted in the Thousand Islands ecosystem in southeastern Ontario, Canada, which is a ~2000 km² watershed that drains into the St. Lawrence River (Figure 3.1). The rural study area is characterized by 33% forest cover, 20% cropland, 14% pasture and field, 10% wetlands, 22% open water and 1% urban development. This

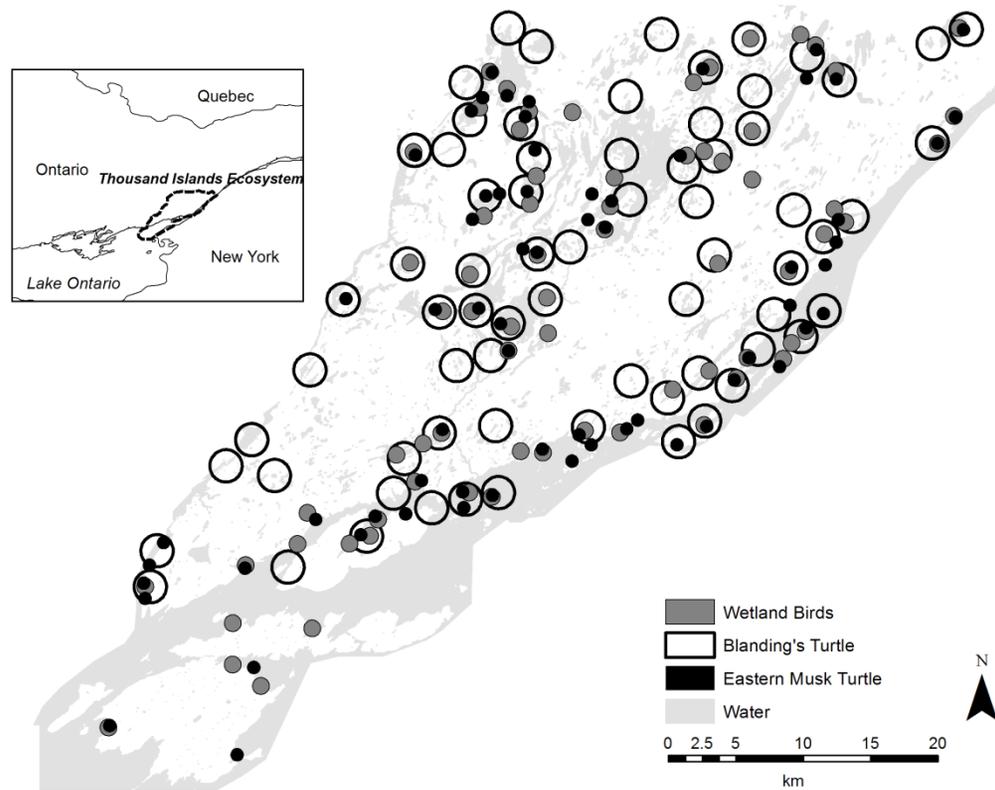


Figure 3.1: Distribution of landscapes sampled across the Thousand Islands ecosystem ($\sim 2000 \text{ km}^2$) study area in southeastern Ontario, Canada. Landscapes were defined as the area within a 625 m radius for wetland birds ($n = 70$ landscapes), a 1.2 km radius for Blanding's turtle (*Emydoidea blandingii*) ($n = 70$ landscapes), and a 500 m radius for eastern musk turtle (*Sternotherus odoratus*) ($n = 66$ landscapes). Reprinted with permission of Elsevier © 2013.

landscape composition is typical of rural northeastern North America (e.g. Samure et al. 2007); however there is less cropland in my study area, possibly due to localized reforestation. Forest is deciduous and mixed, and cropland is primarily corn, hay and soy. There are 5 wetland types (Ecological Land Vegetation classification for southern Ontario, OMNR 2009): 1) shallow open aquatic (water depth <2 m), dominated by floating aquatic vegetation (lily pads; *Nymphaea odorata* and *Nuphar variegata*), and submerged macrophytes (e.g. *Potamogeton* spp.), 2) emergent marsh, dominated by cattail (*Typha* spp.) , 3) shrub thicket swamp, primarily willow (*Salix* spp.) and alder (*Alnus* spp.), 4) deciduous swamp, dominated by maple (*Acer* spp.) and ash (*Fraxinus* spp.) and 5) mixed swamp, consisting of maple and white cedar (*Thuja occidentalis*).

Species Groups and Specific Wetland Habitats

I identified habitat for each species group, from the five wetland types in my study area, based on known habitat associations of the species; wetland types were used for habitat identification only.

Wetland Birds

I selected 8 wetland bird species that are declining in my region (Crewe et al. 2005): red-winged blackbird (*Agelaius phoeniceus*), swamp sparrow (*Melospiza georgiana*), marsh wren (*Cistothorus palustris*), American coot (*Fulica americana*), common Moorhen (*Gallinula chloropus*), sora (*Porzana carolina*), virginia rail (*Rallus limicola*) and least bittern (*Ixobrychus exilis*). Least bittern is designated threatened in Canada and is considered at-risk in 36 U.S. states (NatureServe 2012). Wetland types identified as habitat for wetland birds were emergent marsh or shrub thicket swamp (i.e.

wetland types 2 or 3) (Mowbray 1997, Eddleman et al.1988, Gibbs et al. 2009). I excluded wetland patches <0.4 ha (Gibbs et al. 2009).

Turtles

I selected two declining freshwater turtle species. Blanding's turtle (*Emydoidea blandingii*) is designated as threatened in Canada and considered at-risk in 14 of the 15 states in the U.S. within its range (NatureServe 2012). Eastern musk turtle (*Sternotherus odoratus*; hereafter 'musk turtle') is designated as threatened in Canada and considered at-risk in three U.S. states (NatureServe 2012). Wetland types identified as wetland habitat for Blanding's turtle were shallow open aquatic wetland adjacent to any other wetland type (i.e. wetland types 1 and 2 or 3 or 4 or 5) (Sajwaj and Lang 2000, Joyal et al. 2001). For musk turtle, shallow open aquatic wetland (i.e. wetland type 1) located on a lake or river network (Edmonds and Brooks 1996, Picard et al. 2011) was identified as wetland habitat.

Study Design and Landscape Selection

Here I define "landscape" as the spatial area within which the landscape variables were calculated (i.e. spatial scale). I based landscape size on movement distances and home range estimates from radio-telemetry studies for each species group (Jackson and Fahrig 2012; Appendix F). Landscapes were circles having a 625 m, 1.2 km and 500 m - radius for wetland birds, Blanding's turtle and musk turtle, respectively.

I characterized the landscape structure of the study area to assess collinearity among landscape variables prior to landscape selection. Twelve candidate landscape variables were selected to represent: 1) wetland amount, 2) wetland configuration (total

wetland edge, mean wetland nearest neighbor distance, number of wetland patches, wetland perimeter-area ratio, mean wetland patch size) and 3) matrix composition (agriculture, pasture and field, forest, open water, and road density). All landscape variables were quantified from the Ontario Land Cover dataset (OMNR 2003), except wetlands (OMNR 2009), and analyzed in Fragstats 3.3 and ArcGIS 9.3. I used a moving window analysis across the study area to measure each candidate variable such that the size of the window matched the landscape size selected for each species group (625 m, 1.2 km and 500 m; above). I then performed principal components analyses (varimax rotation) on a random subset of 1000 landscapes for each species group. The first 5 principal components explained 81-85% of the total variation in landscape structure in the study area. I selected one landscape variable from each orthogonal principal component based on the strongest factor loadings and ecological rationale for wetland species: 1) wetland amount (area (ha) of wetland habitat as identified for each species group within a landscape), 2) number of patches (the number of wetland habitat patches within a landscape) as a measure of wetland configuration, and three variables to measure matrix composition, 3) forest amount (area (ha) of forest cover in a landscape), 4) agriculture amount (area (ha) of cropland cover in a landscape); 5) road density (total length of all road types divided by the total area of the landscape (km/km^2)). Using only one variable for wetland amount and wetland configuration, versus three variables for matrix composition might imply that my study was biased in favor of finding matrix effects. However, I suggest the inclusion of three matrix variables was justified because the summed variation in landscape structure explained by the three matrix variables was

similar to the proportions of the variation explained by each of the other two variables (Appendix G).

I used a randomized stratified sampling design to select 100 non-overlapping landscapes for each species group and to minimize multi-collinearity among the five landscape variables. For landscape selection, I defined three strata for each variable, based on the proportion of area or density of that variable within a landscape (e.g. wetland amount: 0-30% “low”, 31-70% “medium” and 71-100% “high”). To the extent possible, an equal number of landscapes were selected per stratum. To avoid a correlation between wetland amount and configuration, I ensured that all possible combinations of the two variables were sampled; this required searching specifically for landscapes with unusual combinations, e.g. high wetland amount with high number of patches, or low wetland amount with low number of patches. I then conducted site visits of each candidate landscape to verify wetland type and obtain landowner permission for wetland access. This resulted in a final set of 70 landscapes for wetland birds, 70 landscapes for Blanding’s turtle and 66 landscapes for musk turtle. In each final set of landscapes, I sampled the full range in variation of each landscape variable, with the exception of the high stratum for agriculture amount (i.e. 71-100% cropland area in a landscape). The number of wetland patches in ranged from 1 to 9 patches in landscapes for wetland birds and 1 to 8 patches in landscapes for both turtle species. Road densities in landscapes ranged from 0 to 9 km/km² for wetland birds, 0 to 5.7 km/km² for Blanding’s turtle, and 0 to 8.7 km/km² for musk turtle. For all species groups, all pair-wise correlations between landscape variables in selected landscapes were $r < 0.46$ and variance inflation factors (VIF) < 3.2 (Table 3.1 and 3.2).

Table 3.1: Variance inflation factors (VIF) and pair-wise Pearson correlation coefficients of landscape-scale predictor variables in landscapes selected (n = 70) for wetland birds calculated at a spatial scale of 625 m. VIF were calculated for each wetland bird species; VIF values reported are for virginia rail (*Rallus limicola*). Landscape predictors are wetland amount (Wetland), wetland configuration (Number of Patches), forest cover (Forest), agriculture cover (Agriculture) and road density. Reprinted with permission of Elsevier © 2013.

	Wetland	Number of Patches	Forest	Agriculture
Wetland (VIF = 1.06)				
Number of Patches (VIF = 1.08)	-0.19			
Forest (VIF = 2.19)	-0.44	0.20		
Agriculture (VIF = 1.82)	-0.19	-0.16	-0.37	
Road Density (VIF = 1.48)	-0.38	0.08	-0.06	0.46

Table 3.2: Variance inflation factors (VIF) and pair-wise Pearson correlation coefficients of landscape-scale predictor variables in landscapes selected (n = 70) for Blanding’s turtle (*Emydoidea blandingii*) calculated at a spatial scale of 1.2 km (above the diagonal) and in landscapes selected (n = 66) for eastern musk turtle (*Sternotherus odoratus*) calculated at a spatial scale of 500 m (below the diagonal). Landscape predictors are wetland amount (Wetland), wetland configuration (Number of Patches), forest cover (Forest), agriculture cover (Agriculture) and road density. Reprinted with permission of Elsevier © 2013.

	Wetland (VIF = 1.36)	Number of Patches (VIF = 1.11)	Forest (VIF = 1.29)	Agriculture (VIF = 1.20)	Road Density (VIF = 1.27)
Wetland (VIF = 1.11)		-0.07	-0.27	-0.16	-0.32
Number of Patches (VIF = 1.01)	0.08		0.33	-0.12	0.10
Forest (VIF = 1.14)	-0.35	-0.12		-0.24	-0.03
Agriculture (VIF = 1.27)	-0.04	-0.04	-0.23		0.30
Road Density (VIF = 1.26)	-0.23	-0.06	-0.05	0.43	

Field Surveys

The aim of the field surveys was to determine whether a landscape was occupied by a focal wetland species. I surveyed wetlands within in each landscape, beginning at the center of the landscape, to determine presence/non-detection, given that I expected low abundance for rare species. For all species groups, I used time-limited surveys to standardized effort across landscapes; travel time between wetlands was excluded from search effort. I was not able to search the entire wetland area of each landscape, or to conduct repeat visits due to logistical constraints. A common challenge of landscape ecology studies is to obtain a sufficient sample of the landscape-scale predictors of interest, across the range of each predictor, while sampling at large spatial scales. To obtain sufficient statistical power to detect independent effects of several landscape variables, I needed to sample a large number of landscapes, here 66 to 70 landscapes. To achieve this, I spread the sampling over two years, 2009 and 2010, with approximately half of the landscapes surveyed in each year; each landscape for each species group was surveyed once.

A potential limitation of this sampling approach is that I am unable to estimate and statistically control for detection probability of each species. However, I designed my field methods to maximize detection probability and minimize potential sources of bias in detectability to the extent possible. For wetland birds, I used call-broadcast surveys which are demonstrated to significantly increase detection probabilities of secretive wetland birds; I selected call-broadcast frequencies tailored to each species. For Blanding's turtle, I used visual surveys when basking turtles are highly detectable, and for musk turtles, I used active short-range searches. These methods are the most effective,

given that I could not conduct trapping, which would have required multiple visits to multiple wetlands in each landscape, a logistical impossibility. In fact, preliminary tests of baited traps were completely unsuccessful for musk turtle. I also minimized variation in detectability across landscapes by restricting sampling to correspond with reported activity periods (e.g. breeding, spring basking) for each species group in my study region to limit seasonal variability. Overall, in the trade-off between the number of visits vs. the number of landscapes, I chose to maximize the number of landscapes surveyed, but I used survey protocols tailored to maximize detection probabilities of each species group.

To avoid spatio-temporal correlations in my results, I sampled landscapes following a randomized block design wherein the study region was divided into four spatial blocks (north, west, east, south), containing equal numbers of study landscapes. The sequence of landscape sampling was then determined by rotating through the four blocks and, within each, randomly selecting a landscape for sampling.

Wetland Bird Surveys

I sampled wetland birds using time-limited (1.5 h) auditory surveys with call-broadcasts. Call-broadcasts significantly increases detection rates for most rails (e.g. by 657% for virginia rail; Conway and Gibbs 2005) and five call-broadcasts improves detection of least bittern (by 80%; Bogner and Baldassarre 2002). My sampling protocol was a modified combination of the Marsh Monitoring Program (MMP 2009) and the National Least Bittern Survey (Jobin et al. 2009) protocols. In each landscape, I established 1 to 5 sample points in wetland habitat at least 50 m from the landscape edge and at least 250 m apart. The number of sample points increased with the amount of wetland in the landscape. Five sample points (totaling 1.5 h) was the maximum number

logistically possible during the sampling time from sunrise and 10h00. At each sample point, I conducted the following 18 minute survey: 1) 1 minute pre-survey silence, 2) 5 minute unlimited-radius auditory survey, 3) 5 minute least bittern call-broadcast (30 s on/30 s off), 4) 3 minute unlimited-radius auditory survey, and 5) 4 minute call-broadcast sequence of American coot, sora, common moorhen and virginia rail, where each species call was broadcasted once (30 s on/30 s off). I conducted most surveys from a canoe, with the exception of wetlands that were only accessible by foot. I conducted all surveys from late May to early July under conditions of no precipitation, temperature ≥ 13 °C and wind speed ≤ 20 km/h.

Turtle Surveys

I sampled Blanding's turtles using time-limited (maximum 4 h; Marchand and Litvaitis 2004) visual surveys (Attum et al. 2008) during mid-April and May, when typically >90% of individuals aerially bask (Sajwaj and Lang 2000) and visibility is maximized as seasonal vegetation growth has not yet begun (Attum et al. 2008). I used high-power binoculars from multiple vantage points that maximized visibility of the wetland shoreline and other potential basking sites (e.g. emergent rocks, logs, muskrat lodges, vegetation clumps and mats; Rowe and Moll 1991). Once I thoroughly searched the wetland in the center of the landscape, I moved to the nearest wetland, when there was >1 wetland patch in a landscape. Surveys concluded when a Blanding's turtle was detected or the time limit was reached. All surveys were conducted by the same observers between 8h00 and 17h30 when the air temperature was ≥ 15 °C (Rowe and Moll 1991) and there was no precipitation.

I sampled musk turtles by time-limited (maximum 8 h over two survey days; Marchand and Litvaitis 2004) active searches. Musk turtles typically bask at the surface of water under floating aquatic vegetation (Edmonds and Brooks 1996). Therefore, I manually searched under floating aquatic vegetation (primarily lily pads) and submerged macrophytes at the water surface within a 1 m x 1 m area directly in front of me, wearing polarized sunglasses to increase visibility. I searched by wading through the water within 5 m of the shoreline, where water depth was ≤ 1 m; search parameters were based on musk turtle activity patterns reported in Carrière (2007). This type of manual search yields higher detection rates relative to other methods (baited traps, dip nets and snorkel; Edmonds and Brooks 1996, Carrière 2007, Quesnelle et al., unpublished data). In addition, it minimizes non-detection error because, if a turtle was within 1 m of me, I was almost certain to detect it. Once I thoroughly searched the wetland in the center of the landscape, I moved to the nearest wetland, when there was >1 wetland patch in a landscape. Surveys concluded when a musk turtle was detected or the 8-hour time limit was reached. All surveys were conducted by the same observers, between 8h00 and 16h00 and from mid-July to mid-August (Carrière 2007) when there was no precipitation.

Statistical Analyses

All landscape variables were standardized to a mean of 0 and a standard deviation of 1. I modeled species that were recorded at 10-90% of landscapes. I used generalized linear models with a logit link function (logistic regression) within a discrete Bayesian model averaging (BMA) framework (Wintle et al. 2003) to determine the effects of landscape structure on wetland species occurrence. The BMA method calculates model-averaged regression coefficients based on Bayesian information criterion (BIC) values to

weight models and produces the posterior probability of a non-zero coefficient [Pr(i)] for each predictor (Wintle et al. 2003). I used the “bic.glm” function in the package ‘BMA’ (Raftery et al. 2012) in R (R Development Core Team, 2012).

I compared regression coefficients from BMA to assess the relative importance of landscape variables in predicting the probability of wetland species occurrence, as other methods for assessing relative importance (e.g. variance partitioning, summed Akaike weights) have been shown to be biased (Smith et al. 2009).

Results

Species were detected in the following proportion of landscapes: red-winged blackbird (1.0), swamp sparrow (0.9), marsh wren (0.5), virginia rail (0.5), least bittern (0.3), common moorhen (0.1), sora (0.1), and American coot (0.0), musk turtle (0.6), and Blanding’s turtle (0.4). Marsh wren, virginia rail, least bittern, Blanding’s turtle and musk turtle were suitable for analysis.

Wetland amount in a landscape was consistently more important than other landscape predictors for all wetland birds analyzed. For two of the three species, wetland amount was more important by an order of magnitude, based on model-averaged regression coefficients (Figure 3.2). There was strong evidence that increased wetland cover at the landscape-scale increased the probability of wetland bird occurrence

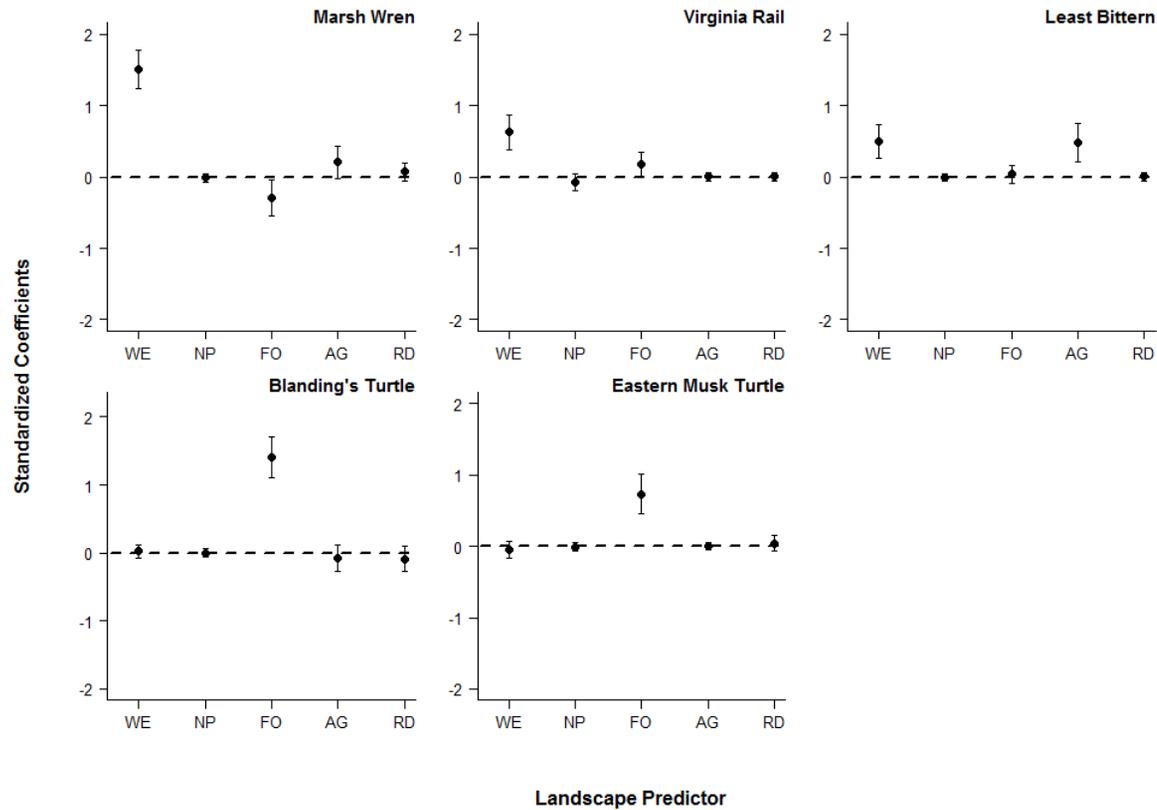


Figure 3.2: Bayesian model averaged coefficients from generalized linear models (logistic regression) of wetland species occurrence on landscape predictors: amount of wetland (WE), number of wetland patches (NP), amount of forest (FO), amount of agriculture (AG), and road density (RD), within sampled landscapes. Predictors were standardized (mean 0, SD 1) before analysis. Reprinted with permission of Elsevier © 2013.

(Figure 3.3). Wetland amount had the highest posterior probability of inclusion in models for wetland birds, where $\text{Pr}(i) > 0.68$ (Appendix H). There was no substantial evidence of effects of wetland configuration, forest cover, agriculture cover, or road density (all $\text{Pr}(i) \leq 0.58$) on occurrence of wetland birds (Appendix H).

For both Blanding's turtle and musk turtle the amount of forest in a landscape (a measure of matrix composition) was more important by an order of magnitude than any other predictor variable, based on model-averaged regression coefficients (Figure 3.2). There was strong evidence that forest cover at the landscape-scale increased the probability of turtle occurrence (Figure 3.3) as the posterior probability of a non-zero coefficient [$\text{Pr}(i)$] was 1.0 and 0.90 for Blanding's and musk turtle, respectively (Appendix H). There was no substantial evidence of effects of wetland amount, wetland configuration, agriculture cover, or road density (all $\text{Pr}(i) \leq 0.20$) on turtle occurrence (Figure 3.2; Appendix H).

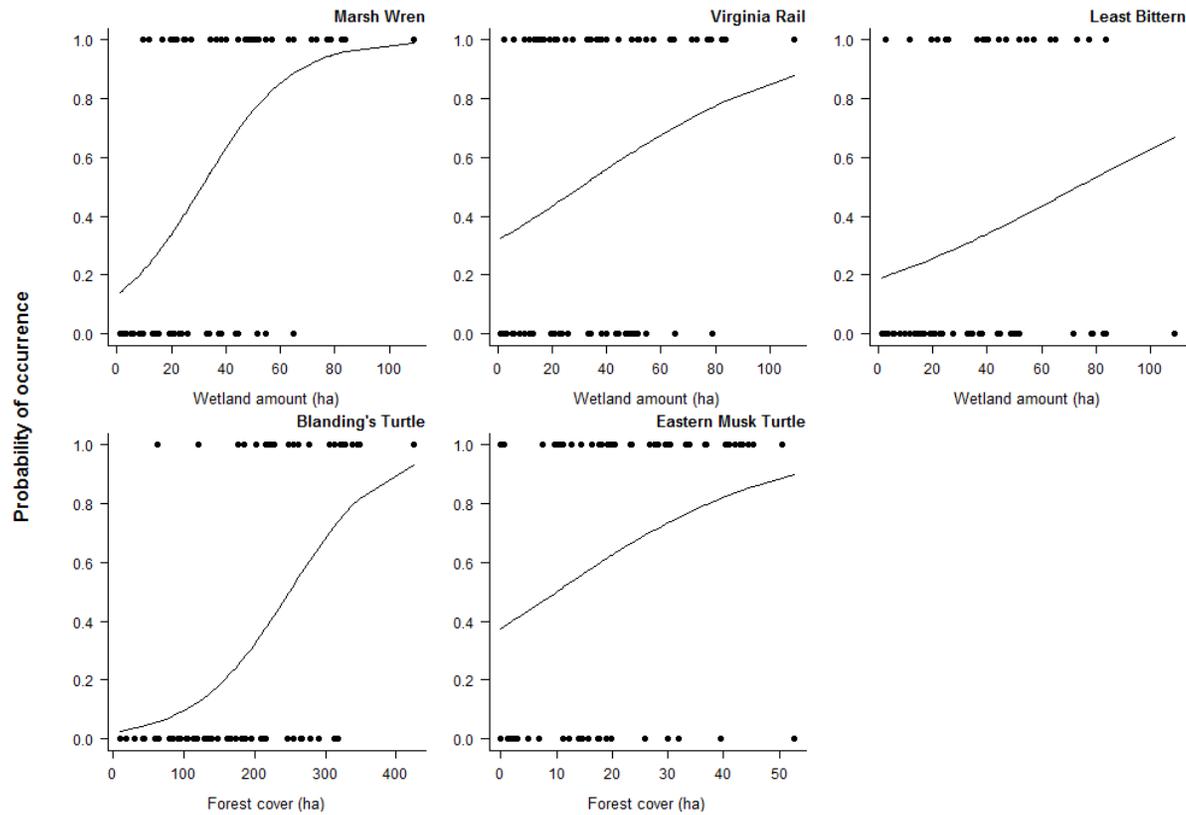


Figure 3.3: Logistic regression models of the landscape predictor with the strongest effect on the probability of occurrence for each wetland species determined by Bayesian model averaging (see Figure 3.2; Appendix H). Points are raw response data (presence/non-detection). Predictions were made holding other landscape variables at a constant mean value. Reprinted with permission of Elsevier © 2013.

Discussion

This is the first study to evaluate the independent, relative effects of habitat amount, habitat configuration, and matrix composition on wetland birds and turtles. My results support my a priori expectation that the amount of wetland in a landscape (i.e. wetland loss) is more important than other landscape-scale predictors of wetland bird distribution. A positive effect of wetland amount at a landscape-scale has been detected in previous studies for several wetland bird species and across several spatial scales (500-4000 m). This positive effect is most likely due to higher food and nesting site availability in landscapes with more wetland (Tozer et al. 2010).

I found that wetland configuration, when unconfounded with wetland amount (i.e. habitat fragmentation per se), did not have an important effect on any wetland species (bird or turtle). I therefore suggest that the negative effects of wetland isolation found in previous studies actually represent a strong positive effect of wetland amount at the landscape scale. In fact, wetland isolation should not be used as an index of wetland fragmentation since, in general, distances between habitat patches *decrease* with increasing fragmentation, when total habitat amount is held constant (Fahrig 2003).

I found that forest and agriculture cover were less important predictors of wetland bird distribution than was wetland amount; a similar result was found by Smith and Chow-Fraser (2010). Previous studies showing an effect of forest cover on wetland birds (e.g. Budd and Krementz 2010) did not control for the correlation between forest amount and wetland amount in the landscape, so these results may have been due to wetland amount rather than forest amount. On the other hand, forest amount in these studies

could have affected wetland birds through its negative correlation with agriculture; landscapes with > 50% cropland are thought to have a negative effect on wetland birds due to high nutrient loading and sedimentation (Naugle et al. 2001). Since landscapes with very high agriculture cover (>55%) were not available in my area, I may have underestimated the potential effect of cropland. On the other hand, the signs of the coefficients relating agriculture to wetland birds were positive (Figure 3.2), suggesting that cropland, at least at lower levels in the landscape (< 55%), may provide supplementary resources (Johnson and Dinsmore 1985).

I found essentially no effect of road density on wetland birds and turtles. This does not appear to be due to the ranges in road density values in my study, since my ranges in values (birds, 0-9 km/km²; turtles 0-5.7 km/km² and 0-8.7 km/km²) were comparable to those in studies that found significant effects of road density on wetland bird communities (e.g. 0-2.52 km/km², Whited et al. 2000; 0-8.8 km/km², Shriver et al. 2004) and turtle population structure and sex-ratios (e.g. 0.5-11.1 km/km², Marchand and Litvaitis 2004; 0.1-7.7 km/km², Steen and Gibbs 2004). It is possible that roads are affecting population abundances in my study but I did not detect these effects because I sampled for presence/absence, not abundance. If my presence-absence sampling was biased to detecting larger, older individuals, and if these individuals are less likely to be killed by roads than younger individuals in the population, then my occurrence data would be a poor indicator of the effects of roads on abundance. On the other hand, it should be noted that a significant negative effect of roads on turtle abundance (as opposed to population structure) has not yet been demonstrated in the literature, so it is possible that my results are actually consistent with previous findings, at least for turtles.

It is also likely that occurrence of at least some of the wetland species would be negatively affected by higher road densities than those in my study, such as those in urban areas (e.g. Smith and Chow-Fraser 2010).

My results did not support my a priori expectation that the amount of wetland in a landscape should be more important than other landscape-scale predictors of turtle distributions; instead, forest amount was the most important predictor. A positive effect of forest amount on turtles at a landscape scale has been detected in previous studies (Findlay and Houlihan 1997, Attum et al. 2008). For Blanding's turtles I hypothesize that landscapes with more forest provide better access to complementary resources thus indirectly supporting greater abundance (i.e. landscape complementation; Dunning et al. 1992). Blanding's turtles are semi-aquatic; individuals can spend up to 38% of the active season in upland forest (Joyal et al. 2001). Forests are used by Blanding's turtles for short-term basking prior to leaf emergence (Beaudry et al. 2009), nesting sites in exposed rocky outcrops or canopy breaks, and refugia for short-term inactivity (Joyal et al. 2001). Vernal pools (small ephemeral wetlands) in forests provide Blanding's turtles with seasonally abundant food resources and rehydration, mating sites and pre-nesting staging areas (Grgurovic and Sievart 2005, Beaudry et al. 2009). Vernal pools are typically not represented in landcover data because the forest overstory obscures their detection by remote sensing devices. Therefore, vernal pools could explain the relationship with forest cover I found. Finally, Blanding's turtles can migrate large distances (up to 2050 m) to different habitat types required for their life cycles, and they typically use multiple wetlands throughout a season (Joyal et al. 2001). Semi-aquatic turtles including Blanding's may preferentially use upland forests for movement relative to open habitats,

possibly due reduced thermal stress (Bowne 2008). Since turtle demography relies on high adult survivorship, factors that reduce adult mortality during movement (i.e. forest cover) could have a strong effect on the distribution of semi-aquatic turtles. Overall, my results suggest that the availability of forest in the landscape represents a more limiting factor than wetland availability on the distribution of Blanding's turtles. If true, it is possible that the apparent sensitivity of Blanding's turtle to aquatic habitat loss and fragmentation found by Grgurovic and Sievert (2005) may actually reflect sensitivity to declining forest amount, which is usually correlated with wetland loss and fragmentation.

In contrast to Blanding's turtles, musk turtles are highly aquatic and seldom leave the water, although gravid females will move overland to nest (mean 851 m), with nests located within 50 m from water (Steen et al. 2012). I suggest that the positive effect of forest cover on musk turtles may result from increasing accessibility to nesting habitats. Musk turtles exhibit relatively high evaporative water-loss (Ernst 1968), resulting in dispersal mortality (Buhlmann and Gibbons 2001); therefore, if accessibility to nesting sites is a limiting factor for this species, this could result in a strong effect of forest cover even though the species is primarily aquatic. Alternatively, since musk turtles are relatively sedentary, local wetland conditions, affected by forest cover (i.e. water quality, hydroperiod), may limit this species. Forest cover may buffer wetlands from fertilizer inputs and sedimentation; DeCatanzaro and Chow-Fraser (2010) observed that musk turtles were not found in wetlands of low water quality. On the other hand, I did not find an effect of agriculture on musk turtle. This is not say that such an effect would not occur at cropland covers higher than those in our study (>55%); however it does imply that the positive effect of forest cover I observed is not an indirect negative effect of

agricultural inputs. A related explanation is that more forested landscapes could maintain wetland hydrology and drainage patterns within a landscape (Richardson and McCarthy 1994), to support the occurrence of musk turtles. However, the negative correlation between wetland amount and forest amount at the landscape scale (Table 3.2) indicates this alternate explanation is unlikely. Therefore, I suggest that the most likely explanation for the positive effect of forest cover on musk turtle occurrence is the higher accessibility of nesting sites, which is presumably a limiting factor for their distribution.

I also considered that the relatively strong positive effect of forest cover on both turtle species could be due to a negative correlation between forest cover in the landscape and wetland riparian disturbance. Forested riparian loss can affect the abundance of turtles (Sterrett et al. 2011) by influencing wetland microclimate (water temperature), resource availability (food), and microhabitat (large woody debris). I calculated proportion forested riparian in each sampled landscape as the total length of wetland edge immediately adjacent to forest, divided by the total wetland edge length. I conducted post hoc analyses of the effect of proportion forested riparian on turtle occurrence by including it as a predictor in models with the landscape variables. Proportion forested riparian did not improve model fit for either turtle species (Appendix I), which suggests that forest cover beyond the wetland riparian zone likely explains the occurrence of both turtle species, as argued above.

It is surprising that wetland amount in the landscape had essentially no effect on the occurrence of the two turtle species. However, unlike wetland birds where feeding and breeding occur solely in wetlands, turtles require different habitats in their life cycle, which necessitates overland movement, increasing the importance of matrix composition

(e.g. forest cover). Indeed, local extirpation and shifting population structure of freshwater turtles has occurred despite protection of aquatic habitat in an area of intense anthropogenic development (Browne and Hecnar 2007), highlighting the importance of matrix composition. In fact, reptiles in general have relatively weak responses to patch area effects (Prugh et al. 2008).

I was unable to estimate the probability of detection of the wetland species surveyed because the study design necessitated a large sample size and logistical constraints prevented repeated visits. However, I suggest that my sampling methods maximized detectability of each species such that additional visits would only marginally increase the probability of detecting occupancy in a landscape. I also note that occurrence rates of marsh wren, virginia rail and least bittern in this study were within the range of detection reported in other landscape-scale studies using call-broadcast with multiple visits (Rehm and Baldassarre 2007, Tozer et al. 2010). Also, the occurrence rate of Blanding's (0.36) was similar to or higher than occurrence rates from trapping with repeated visual surveys (0.26; Joyal et al. 2001) or repeated visual surveys alone (0.10; Attum et al. 2008). Similarly for musk turtle, the sampling method selected (active surveys) yielded higher detection rates relative to baited traps and I maximized detectability by conducting searches over two days, if necessary.

Although my detection rates are high, it is possible that detectability covaried with landscape predictors, such that there were higher detection probabilities of wetland birds in landscapes with more wetland cover or of turtles in landscapes with more forest cover, which could cause spurious results. I evaluated this possibility by testing, post hoc, for correlations between factors that could affect detectability and landscape

predictors. These factors included Julian day, air temperature, cloud cover, and length of wetland shoreline. All of these were weakly correlated ($r < 0.2$) with landscape variables, indicating my results for wetland cover and forest cover are likely not spurious effects of detectability bias caused by these factors (Appendix J). Alternatively, perhaps an unknown local variable covaried with detectability and the landscape predictors, thus introducing a detection bias for which I am unable to test (Gu and Swihart 2004). While this is possible, such a bias would have to be quite strong to produce the large magnitude of effects of wetland amount and forest cover (Figure 3.2) I observed.

Conclusions

Given that wetland birds and turtles are undergoing some of the steepest wildlife population declines worldwide (Millennium Ecosystem Assessment 2005), this study has important implications for wetland conservation and landscape planning. My results confirm that wetland loss is the primary landscape-scale factor of wetland bird declines and suggest that forest loss plays a greater role in freshwater turtle declines than previously realized. Firstly, this suggests that minimizing wetland loss, irrespective of configuration, will be of most benefit for wetland bird conservation. Secondly, given the protection of core wetland habitat (Semlitsch and Bodie 2003), minimizing forest loss surrounding protected core wetlands will be of most benefit for freshwater turtle conservation. The large effect of forest cover on turtles provides the first (unconfounded) support for arguments that wetland buffers are insufficient and protected forest in wetland policy is imperative to conserve turtles, both as habitat (Buhlmann and Gibbons 2001) and travel corridors (Roe et al. 2007). It also suggests that increases in landscape

complementation (i.e. access to multiple resources) should take priority over reductions in wetland isolation (i.e. wetland amount) in landscape planning for turtle conservation. Moreover, my results suggest that wetland creation alone would have little impact on declining freshwater turtles; restoration of the matrix (i.e. reforestation) surrounding protected core wetlands would be more effective. For all declining wetland species, wetland policies and restoration plans based on wetland configuration (e.g. mean wetland size, nearest-neighbor distance) would have low returns unless they simultaneously increased total wetland amount or forest amount; alterations to configuration alone cannot offset the effects of overall loss of wetland or forest on birds and turtles, respectively. Therefore, my results demonstrate that understanding the relative importance of landscape-scale effects can inform priorities for habitat conservation and restoration in the context of multi-species conservation.

Despite that the birds and turtles I studied are all ‘wetland-dependent fauna’, the two taxa responded to different landscape factors. This is consistent with the general observation that there are low correlations between taxa in their responses to environmental change (Wolters et al. 2006). My results suggest that these low correlations could be due to differences in landscape-scale requirements. Wetland-nesting obligates that depend on specific wetland vegetation communities (e.g. least bittern, marsh wren) were more sensitive to landscape structure than were wetland-nesting generalists (e.g. red-winged blackbird), and species with life-cycles requiring more than one habitat type (turtles) were more strongly affected by landscape complementation, than wetland loss. Given these differences in responses to landscape structure, the practice of using one taxon (often birds) as a biodiversity model for

landscape planning is likely to fail for other groups. Even for a group of species identified by their primary habitat use (here ‘wetland-dependent fauna’), a multi-taxa approach should be a mandatory component of landscape conservation planning.

Chapter 4: Relative effects of landscape-scale wetland amount and landscape matrix quality on wetland vertebrates: a meta-analysis

Abstract

Conservation management of wetland-dependent species generally focuses on preserving or increasing wetland habitat. However, the quality of the landscape matrix (the intervening non-wetland portion of the landscape) has been shown to be more important than wetland availability for some wetland-dependent species. I used meta-analysis to compare the effects of wetland amount (measured as the area of wetland habitat in a landscape) and matrix quality (measured as the area of forest cover in the same landscape) on the population abundance of wetland-dependent vertebrates. I combined data across 63 studies worldwide and extracted 330 population responses for 155 species, at the spatial scale that best predicted the effects of wetland amount and forest amount for each response. In addition, to ensure my results were not biased by the scale selected, I assessed whether the relative effects of wetland and forest amount were scale-dependent. I found that the amount of wetland in a landscape had a larger effect than forest amount on the abundance of mammals and birds whereas, surprisingly, for amphibians the amount of forest in a landscape was more important than the amount of wetland. For reptiles, both wetland amount and forest amount showed only weak effects on abundance. These results were not scale-dependent, i.e. they were consistent across spatial scales. My results suggest that the population distribution of wetland-dependent amphibians is more strongly related to landscape matrix quality than wetland availability

in a landscape, likely due to their requirement for access to terrestrial resources. I conclude that conservation policies for wetland biodiversity that focus only on wetland habitat will be ineffective in conserving many of these species. In addition, population viability analyses based only on wetland amount may overestimate the capacity of a landscape to support populations of wetland-dependent species.

Introduction

Wetland-dependent species are undergoing some of the largest wildlife population declines worldwide, primarily due to habitat loss (Millennium Ecosystem Assessment 2005). Although it is widely recognized that many wetland-dependent species depend on more habitat types than just wetlands (Semlitsch and Bodie 2003), conservation management of these species generally focuses on preserving or increasing wetland habitat in a region. This focus is reinforced not only by the simple fact that we label these organisms "wetland-dependent," but also by the patchy nature of wetland habitat, which has led to the common assumption that many wetland-dependent species function as metapopulations (Gibbs 2000, Marsh and Trenham 2001, Vermaat et al. 2008). Thus, the most common policies used to protect wetland-dependent species are wetland regulations that preserve the wetlands themselves and, in some cases, small terrestrial buffers surrounding the wetlands (Haig et al. 1998, Semlitsch and Bodie 2003, Bauer et al. 2010). For example, worldwide establishment of wetland reserves (e.g. Ramsar Convention), international strategies to identify and protect key wetlands (e.g. North American Waterfowl Management Plan), and regional "no net loss" wetland policies (e.g.

US National Wetlands Mitigation Action Plan) primarily focus on preserving individual wetlands and groups of wetlands, or maintaining some total amount of wetland in a landscape, to conserve wetland biodiversity. Moreover, large-scale wetland restoration and creation aimed at increasing the wetland amount in a landscape is common practice for wetland-dependent species recovery plans (e.g. Gilbert et al. 2005) and to increase overall wetland biodiversity (Moreno-Mateos and Comín 2010).

However, for some wetland-dependent species, the quality of the surrounding landscape matrix (the intervening non-wetland portion of the landscape) can be more important than the amount of wetland in a landscape. For example, the amount of cropland in a landscape was found to have a larger effect than wetland amount on wetland breeding bird occurrence in the Prairie Pothole Region of South Dakota (Naugle et al. 1999) and agricultural landscapes of northeastern Spain (Cardador et al. 2011). Similarly, the amount of forest in a landscape was found to be relatively more important than the amount or spatial distribution of wetlands in a landscape for the abundance of several wetland breeding amphibians (Houlahan and Findlay 2003, Rubbo and Kiesecker 2005, Denoël and Ficetola 2008), as well as the occurrence of two freshwater turtle species and one watersnake species (Attum et al. 2007, Chapter 3). Moreover, roads and/or urban development have been found to be a better predictor of abundance and distribution of several amphibians (Vos and Chardon 1998, Pillsbury et al. 2008, Veysey et al. 2011) and freshwater turtles (Rizkalla and Swihart 2006) than the amount of wetland in a landscape. If other landscape variables are often more important than wetland amount for wetland-dependent species, then conservation strategies that focus

solely on preserving and/or increasing wetlands in a given landscape are likely to be ineffective at protecting these declining species.

In this study, I conducted a meta-analysis to determine whether, and for what species groups, the focus on wetland conservation is likely to fail to conserve populations of wetland-dependent species. I note that this issue has been raised in several papers, for several individual species or groups of wetland-dependent species (Semlitsch and Bodie 2003, Roe et al. 2006, Roe and Georges 2007), but has never been generally assessed across a range of wetland-dependent vertebrate taxa using quantitative methods. I compared the relative effects of wetland amount and matrix quality by combining the information in all available studies of wetland-dependent species using meta-analysis. I measured wetland amount as the area of wetland habitat in a landscape. I defined matrix quality as the amount of forest cover in the same landscape, where a landscape with more forest is assumed to represent a landscape with higher matrix quality. I had two reasons for making this assumption. First, forest amount in a landscape generally has a positive influence on wetland biodiversity and is typically negatively correlated with indicators of low matrix quality, such as road density (Findlay and Houlihan 1997). Second, the meta-analytic approach required me to independently estimate matrix quality for multiple studies across the globe, and forest is a land cover type that can be confidently measured from classified satellite imagery datasets. I quantitatively synthesized the results of 63 studies conducted across 5 continents that quantified the relationship between wetland-dependent animal abundance and wetland amount in a landscape. From these I obtained 330 population responses to wetland amount for 155 species including mammals, birds, amphibians and reptiles. I also obtained information on forest cover around each wetland

in each of these studies, and then compared the strength of the relationship between species abundance and wetland amount to the strength of the relationship between abundance and forest amount (Figure 4.1).

Methods

Study selection criteria

During my literature review and subsequent effect size extraction, I recorded the number of articles identified and the number of studies included and excluded based on my criteria according to the Preferred Reporting Items for Systematic Reviews and Meta-Analysis (PRISMA) Statement, and I also recorded the number of studies involved at each step of the meta-analysis (Figure 4.2). The studies included in this meta-analysis are a subset of the larger review of studies described in Chapter 2 that quantified the relationship between the amount of wetland habitat in a landscape and population abundance of at least one wetland-dependent species (Figure 4.2, steps 1-3). The current analysis includes all the studies in the Chapter 2 meta-analysis for which I could obtain the raw abundance data and forest cover in the surrounding landscapes, to estimate the relationship between abundance and forest cover (Figure 4.2, step 4). I again defined “wetland-dependent species” as any vertebrate (mammal, bird, reptile or amphibian) that uses wetlands as primary habitat for at least one part of its life cycle. All study inclusion criteria were the same as described in the Methods section of Chapter 2 (see ‘Study selection criteria’). The only additional criteria for Chapter 4 were as follows. I included

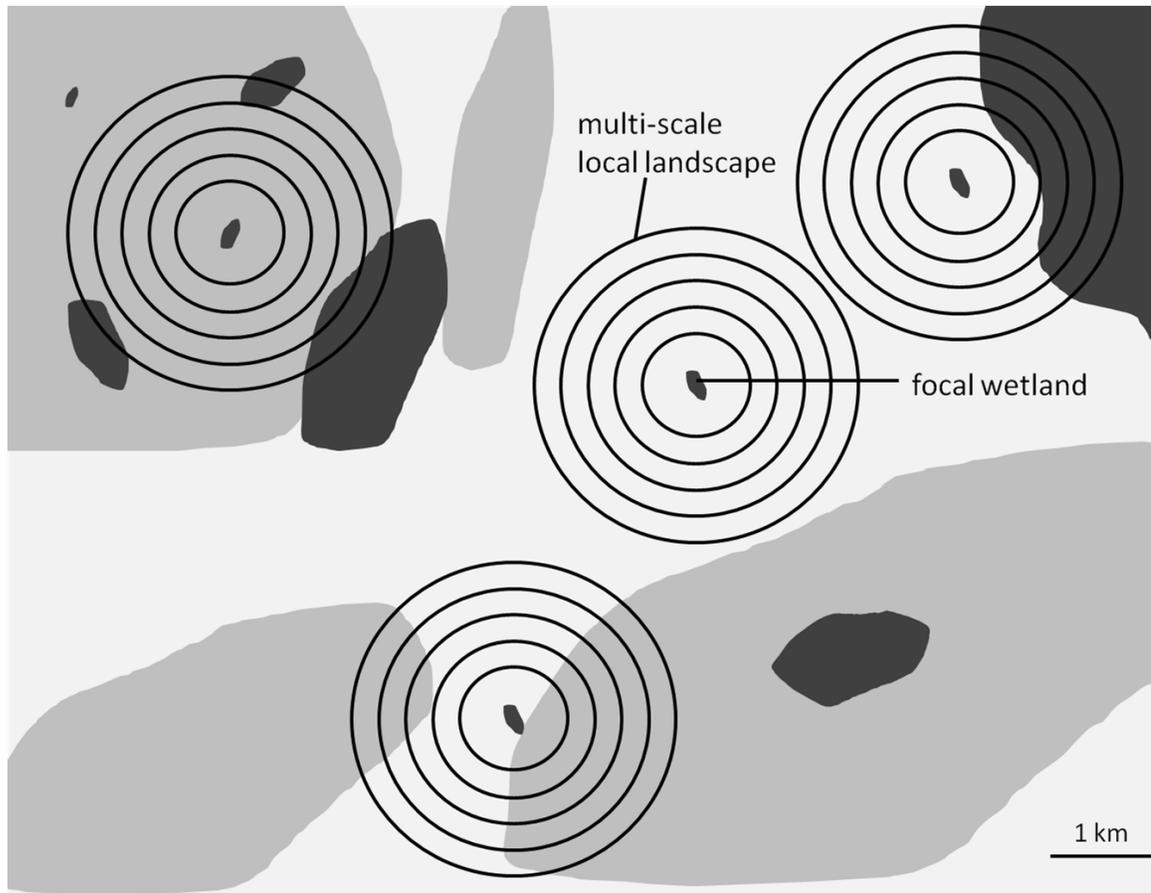


Figure 4.1: Example of the sampling design of a landscape scale study included in the meta-analysis. A species response (occurrence, abundance or density) is sampled at a set of focal wetlands within a study area. The study area is comprised (for simplicity) of three landcover types: wetland (dark grey), forest (light grey) and field (white). The amount of wetland area and amount of forest area are measured within nested local landscapes at multiple spatial scales surrounding each focal wetland. The scale of effect is the spatial scale where the strength of the relationship between species response and wetland amount (or forest amount) is the greatest.

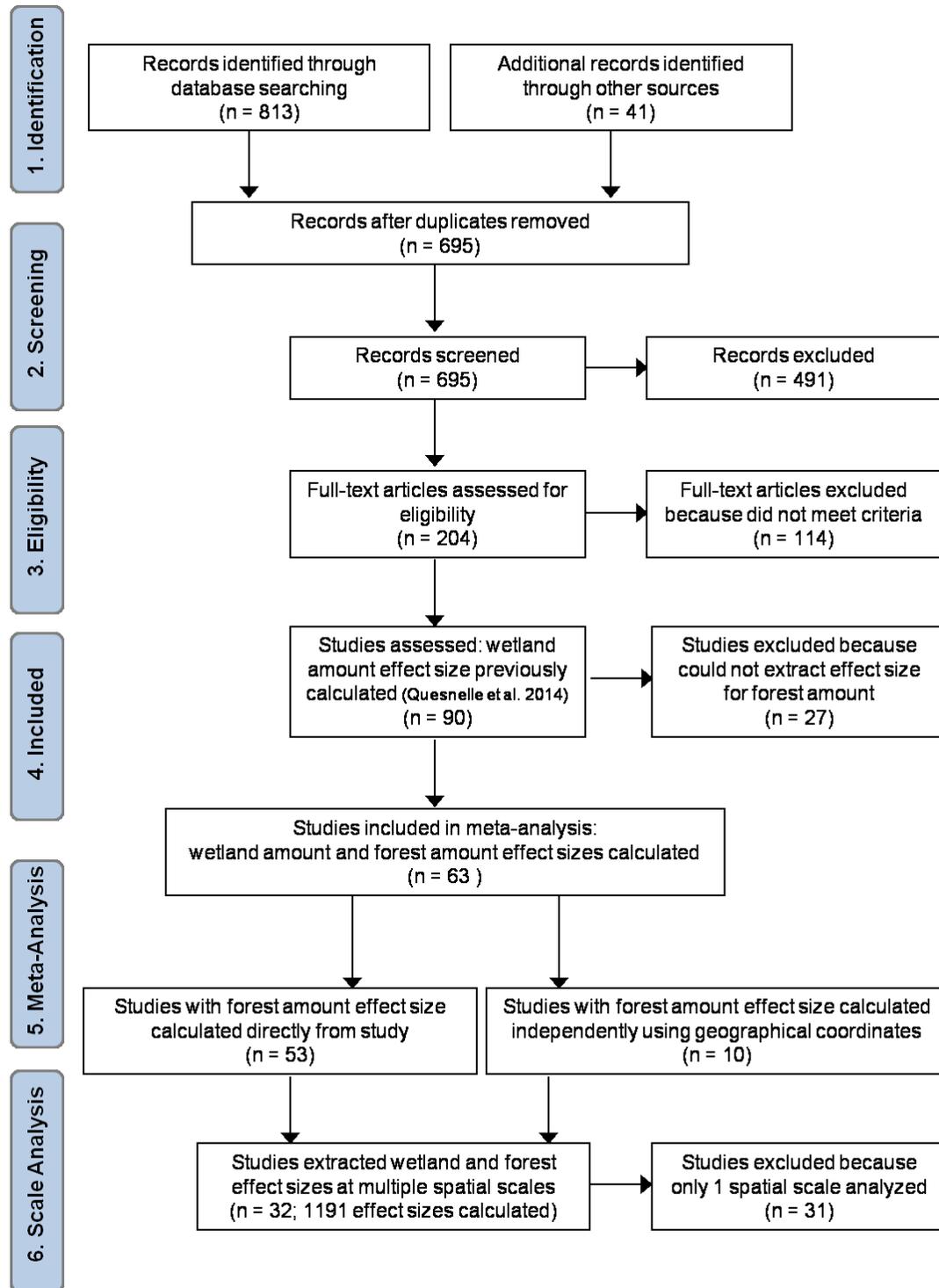


Figure 4.2: Modified PRISMA flow diagram indicating the number of studies considered at each step (identified in horizontal boxes on left) of the literature search and meta-analysis.

studies that measured forest amount (or for which I could obtain forest amount) as the percent of forest area in a landscape or the distance to the nearest forest (nearest-neighbour), depending on how wetland amount was measured (as area or distance). Wetland amount and forest amount data also had to be measured or measurable at the same spatial scale(s).

Effect size extraction

I had two effect sizes (ES) in my meta-analysis: Wetland ES, representing the quantitative relationship between population abundance of a given wetland species and the amount of wetland in a landscape, and Forest ES, representing the quantitative relationship between population abundance for the same wetland species and the amount of forest in a landscape. I extracted effect sizes from each study as described in the Methods section of Chapter 2 (see ‘Effect size extraction’) in addition with the following methodology.

When authors did not measure the forest amount in a landscape, or when forest was not measured at the same spatial scale(s) as wetland amount, I contacted authors to obtain spatial coordinates (with associated datum and projection information) to measure forest amount in each study landscape independently using classified satellite imagery datasets (Figure 4.2, step 5). For studies within the United States, I used the National Land Cover Data (Fry et al. 2011), a 30 m resolution classified landcover dataset based on Landsat Thematic Mapper (TM) satellite imagery from 2006. For studies outside of the United States, I used the Landsat Tree Cover, a 30 m resolution continuous tree cover dataset based on Landsat-5 TM and/or Landsat-t Enhanced Thematic Mapper Plus

(ETM+) from 2000 and 2005 (Sexton et al. 2013). The Tree Cover dataset estimates the percentage of horizontal ground in each pixel covered by woody vegetation greater than 5 m in height (i.e. tree cover) across the globe. I classified pixels with > 50% tree cover as forest. I then measured forest amount in each of the study landscapes at the same spatial scale(s) that the study measured wetland amount using ArcGIS 10.

When studies measured effects of wetland amount and forest amount at multiple spatial scales, I calculated an effect size for each variable at each scale. I then selected the largest estimate across scales measured for each of Wetland ES and Forest ES, on the assumption that this scale was closest to the scale at which each landscape variable best predicts the species' response (i.e. scale of effect, *sensu* Jackson and Fahrig 2012).

Study design moderators

I identified four study design moderators to test if differences in study design influenced the magnitude and direction of the effect sizes, and to statistically control for such effects in remaining analyses (Table 4.1). Three of the four study design moderators variables - study type, sampling effort, and wetland patch area - are as described in the Methods of Chapter 2 (see 'Study design moderators'); however there were additional considerations for the study type moderator as follows.

The Wetland ES or Forest ES extracted from each study might vary depending on how wetland or forest amount in a landscape was measured. I combined several measures of wetland amount and forest amount, including simple area-based measures and nearest-neighbour distances as well as more complex connectivity indexes based on the incidence function model, because these measures have been shown to be highly

Table 4.1: Study design moderator variables included in the meta-analysis.

Moderator Variable	Categories	Description
Study Type	Amount-based	Wetland amount was measured as percent wetland area in a landscape or buffer (area-based buffers)
	Configuration-based	Wetland amount was measured as the number of wetland patches in a landscape, or using wetland isolation (nearest-neighbour distances) or connectivity (incidence function model) metrics
Sampling Effort	Area-dependent	Sampling effort increased in proportion to the sampled wetland area
	Area-independent	Sampling effort was consistent across sampled wetlands
	Unknown	Relationship between sampling effort and sampled wetland area unknown
Patch Area	Included	Sampled wetland area was included in the calculation of wetland amount in the landscape
	Not included	Sampled wetland area was not included in the calculation of wetland amount in the landscape
Correlation	Continuous	Correlation between wetland amount and forest amount in a study

correlated and have similar performance in predicting ecological responses (Moilanen and Nieminen 2002, Bender et al. 2003, Tischendorf et al. 2003, Prugh 2009, Ranius et al. 2010, Thornton et al. 2011b, Martin and Fahrig 2012). These comparative studies also suggest that measures with more information about the amount of habitat in the landscape are better predictors, and therefore I expected a priori that studies using such measures would have larger effect sizes. I distinguished two study types, 1) amount-based studies, where wetland (or forest) amount was calculated as the percent wetland (or forest) area in a landscape or buffer surrounding the sampled wetland patch, or 2) configuration-based studies, where the configuration of wetland (or forest) was included in the calculation of the measure, such as the number of wetland (or forest) patches in a landscape, nearest-neighbour distances, wetland proximity or wetland connectivity. For all study types, I applied the convention that each effect size extracted from a study should represent the population response of a species to increasing wetland amount (or forest amount) in a landscape. However, for nearest-neighbour studies, a negative effect of increasing distance indicates that a species responded positively to closer wetlands (or forests), or equivalently, greater wetland (or forest) amount within the surrounding landscape. Therefore, I reversed the sign of the effect sizes extracted for nearest-neighbour studies to make them comparable to those extracted for all other studies representing the response to increasing wetland or forest amount (Gurevitch and Hedges 1993).

The fourth study design moderator considered that the effect of wetland amount (Wetland ES) and/or forest amount (Forest ES) may be confounded by the correlation between the two landscape variables. For example, if the amount of wetland and forest were positively correlated in a study, a positive Forest ES estimate may be statistical

artefact. Therefore, I recorded the correlation between wetland amount and forest amount in each study. If wetland amount and forest amount were measured at multiple spatial scales in a study, I calculated the correlation at their respective scales of effect. For example, if the scale of effect was 500 m for Wetland ES and 1000 m for Forest ES, I calculated the correlation between wetland amount measured at 500 m and forest amount measured at 1000 m.

Effect size calculations

I selected the Pearson correlation coefficient r between the population response and wetland or forest amount as my estimate of Wetland ES and Forest ES from each study. I used the same methods described in the Methods section of Chapter 2 (see ‘Effect size calculations’) to calculate and transform all effect sizes.

The next step was to obtain accurate and comparable sample sizes across studies. Meta-analysis weights each study by its inverse variance, based on the assumption that studies with greater precision will provide a more accurate estimate of the true effect. The variance of ES_{Zr} is approximated as $V_{Zr} = 1 / (n-3)$, where n is the total sample size of the study (Borenstein et al. 2009). I used the same process described in the Methods section of Chapter 2 (see ‘Effect size calculations’) to assess and obtain accurate sample sizes for each study included in Chapter 4.

After determining the adjusted sample size ($n_{adjusted}$) of each study, I calculated the inverse variance weight for each ES_{Zr} as $w = n_{adjusted} - 3$ (Lipsey and Wilson 2001). Studies with $w < 1$ were excluded from the meta-analysis. I also classified effect sizes by taxonomic group at the class level: mammal, bird, reptile and amphibian, and by order

within each taxonomic group. Refer to Appendix K for studies included in the meta-analysis and associated country, species with taxonomy classification, effect sizes, adjusted sample sizes and study-design categories.

Meta-Analyses

To assess the relative effects on wetland animal abundance of wetland amount and forest amount in a landscape, I used the estimates for each of Wetland ES_{Zr} and Forest ES_{Zr} calculated at their scale of effect (i.e. the largest ES_{Zr} calculated from the range of scales measured in a study, as described above). I conducted random-effects meta-analyses using the DerSimonian-Laird method to determine the summary weighted-mean effect size of the overall population response of wetland species to wetland amount at the landscape scale and the summary weighted-mean effect size of the overall population response of wetland species to forest amount at the landscape scale. Under the random-effects model, the weight assigned (w^*) to each effect size is the inverse of the sum of two variance components $w^* = 1/(w + T^2)$, where w (see above) is the unique sampling variance for each study (within-study error) and T^2 is the pooled variance of the true effects across all randomly selected studies (between-studies variance (Borenstein et al. 2009)). I also calculated the heterogeneity in true effects (Q statistic), which I compared against a chi-square distribution, to test whether the total variation in observed effect sizes (Q_T) was significantly greater than that expected from sampling error (Q_E).

I then tested whether moderator variables - study type, sampling effort, wetland patch area, and taxonomic class - could explain variation in the effect sizes (Q_M), i.e. $Q_T = Q_M + Q_E$, by performing univariate mixed-effects meta-analysis using restricted

maximum-likelihood (REML) estimation of heterogeneity. If a moderator variable explained significant heterogeneity in the effect sizes, I would then subset my data by that moderator variable to control for its influence on the relative effects of wetland amount and forest amount. To determine the relative effects of wetland and forest, I used a paired-sample t test to compare the mean z -transformed correlation coefficients between wetland animal abundance and each of these two landscape variables (i.e. mean Wetland ES_{Z_r} – mean Forest ES_{Z_r}). I used a paired-sample t test because Wetland ES_{Z_r} and Forest ES_{Z_r} were estimated for the same species within each study. I then performed a mixed-effects meta-regression to test whether the difference in effect size for wetland and forest for a study was confounded by the correlation between wetland amount and forest amount in the study.

To ensure the results were not biased by the spatial scale selected in each study, I assessed if the relative effects of wetland amount and forest amount were scale-dependent (i.e. changed with scale). I used each pair of Wetland ES_{Z_r} and Forest ES_{Z_r} estimated at every spatial scale measured for each species in a study (Figure 4.2, step 6). At each spatial scale, I calculated the absolute difference in effect size for wetland amount and forest amount (i.e. $|Forest\ ES_{Z_r} - Wetland\ ES_{Z_r}|$) to get the relative effect size. I then conducted linear mixed-effects regression with nested random effects using REML estimation. The response variable was the relative effect size and the predictor variable was $\log(\text{scale in m})$. The random variables were ‘study’ to control for variation among study areas, and ‘species’ nested within ‘study’ to account for studies that measured multiple species responses at multiple scales. I tested if the relative effect size changed significantly with scale using 95% highest posterior density (HPD) confidence intervals

and p-values generated from Markov chain Monte Carlo simulations ($n = 1000$ randomizations).

Lastly, I assessed publication bias by a rank correlation (Kendall's tau) test of the relationship between ES_{Z_r} and n along with visual inspection of a scatterplot between these two variables following (Jennions et al. 2013). All analyses were conducted in R 3.0.1, using the packages 'metafor', 'lme4' and 'languageR'.

Results

Although I found more than 200 studies that examined the effect of wetland amount in a landscape on population abundance of wetland-dependent vertebrates, only 63 of these met the inclusion criteria (Figure 4.2). These 63 studies, from 12 countries, generated 330 effect sizes across 155 species (including 7 species complexes). Studies were predominately from North America (46) and Europe (11), with remaining studies from Australia (3), South America (2), and Asia (1) (Appendix K). The summary weighted-mean effect size for wetland amount from a random-effects meta-analysis across all taxa was 0.10 (95% CI: 0.079, 0.129; $n = 330$), indicating an overall weak, positive effect of wetland amount in a landscape on wetland animal population abundance. The overall heterogeneity was $Q = 659.19$ ($p < 0.0001$), indicating highly significant variation in species responses to wetland amount. The summary weighted-mean effect size for forest amount from a random-effects meta-analysis across all taxa was 0.08 (95% CI: 0.048, 0.114; $n = 330$), indicating an overall weak, positive effect of forest amount in a landscape on wetland animal population abundance of the same

magnitude as the overall effect of wetland amount. The overall heterogeneity was $Q = 1158.13$ ($p < 0.0001$), indicating highly significant variation in species responses to forest amount. The correlation between wetland amount and forest amount across landscapes within a study was generally low, but varied widely across studies (mean $r = -0.02$, range $-0.84 - 0.80$; Appendix L). There was no strong evidence of publication bias, as there was a weak relationship between Wetland ES_{Zr} and sample size (Kendall's tau = -0.001 , $p = 0.99$), and between Forest ES_{Zr} and sample size (Kendall's tau = 0.023 , $p = 0.54$). Moreover, scatterplots between Wetland ES_{Zr} and sample size, as well as Forest ES_{Zr} and sample size, showed effect sizes were symmetrically distributed around the summary effect and produced a funnel-shape with greater variation in studies at low sample sizes as expected (Figure 4.3 and 4.4).

Mixed-effects meta-analysis across all taxa ($n = 330$) showed that none of the study design moderators - study type, sampling effort, or sampled wetland area - explained significant heterogeneity in the effects (study type: $Q_M = 0.66$, $p = 0.418$; sampling effort: $Q_M = 5.85$, $p = 0.055$; sampled wetland area: $Q_M = 2.83$, $p = 0.092$; Appendix M). Therefore I did not control for study design. The effect of wetland amount in a landscape on wetland animal abundance varied by taxonomic class ($Q_M = 30.57$, $p < 0.0001$; Appendix M). The weighted-mean effect size of wetland amount for mammals and birds was greater than that of amphibians and reptiles (Figure 4.5). Similarly, the effect of forest amount in a landscape on animal abundance varied by taxonomic class ($Q_M = 18.71$, $p = 0.0003$; Appendix M). However, the effect of taxa on the forest amount effect was opposite to the effect of taxa on the wetland amount effect;

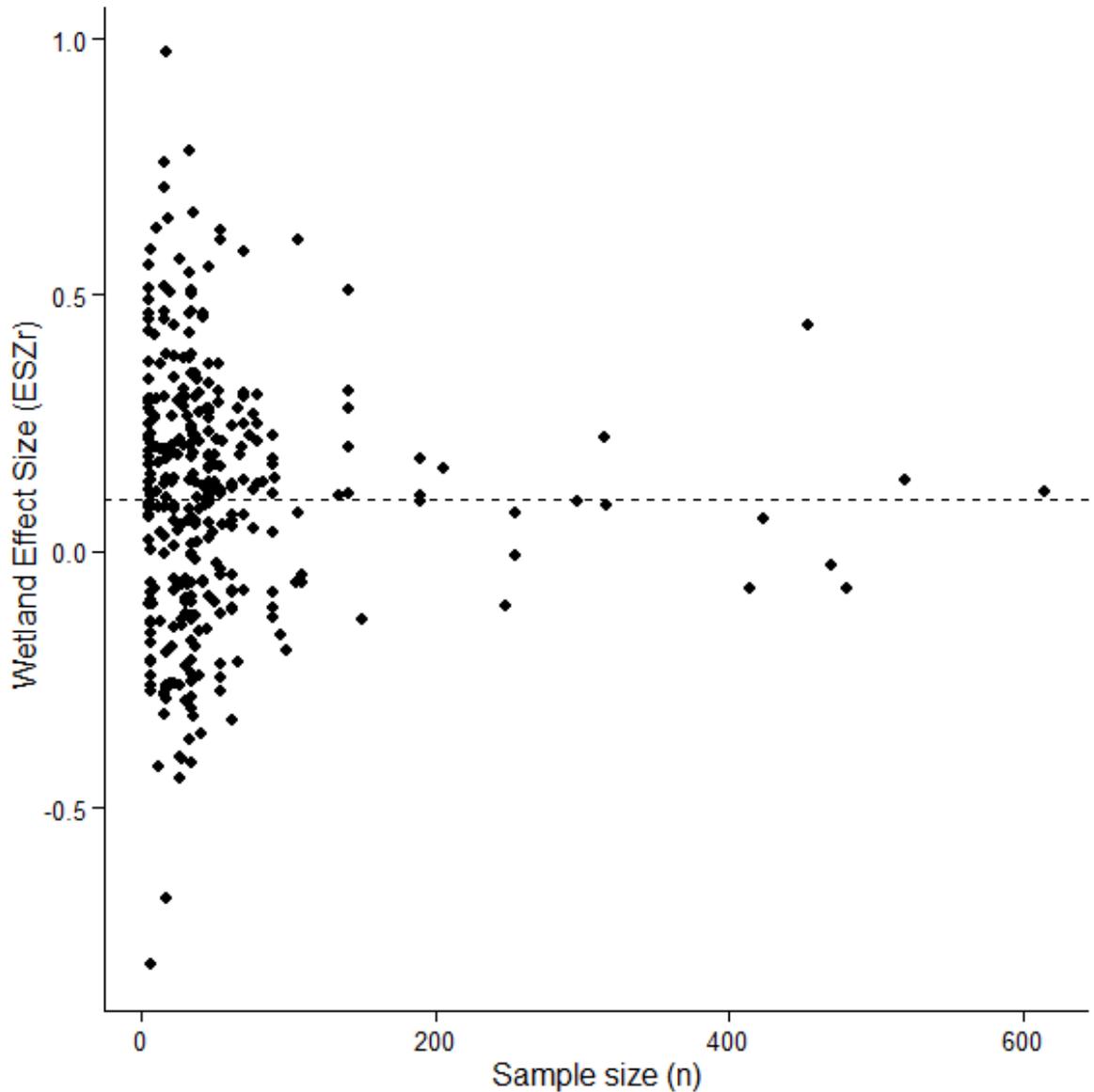


Figure 4.3: Relationship between z -transformed correlation coefficients (ES_{zr}) for wetland amount and sample size (n) to assess publication bias. Dashed line is the summary mean-weighted effect size from random-effect meta-analysis across 330 effect sizes from 63 studies. There was no strong evidence of publication bias since effect sizes were symmetrically distributed around the summary effect and produced a funnel-shape with greater variation in studies at low sample sizes.

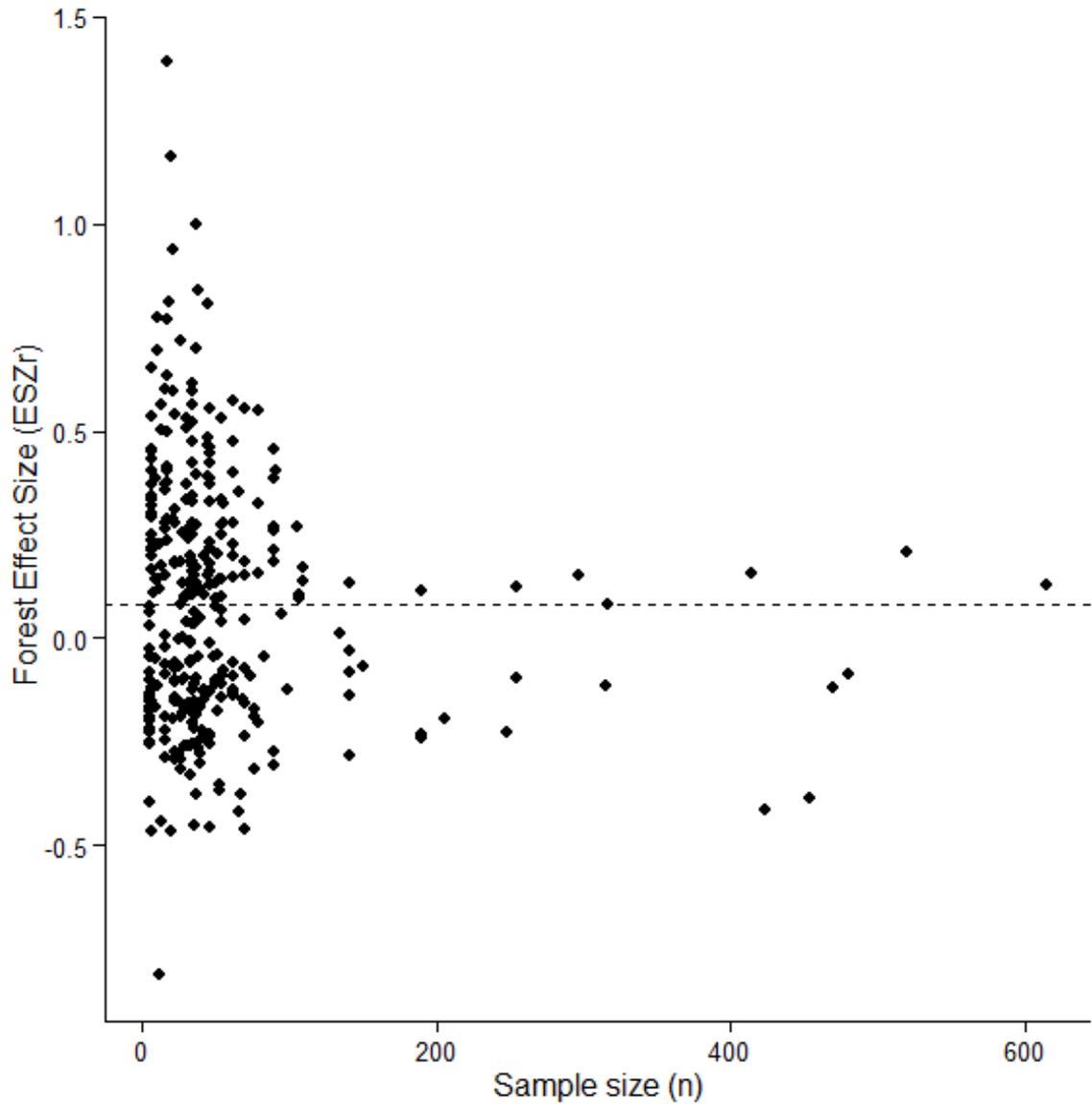


Figure 4.4: Relationship between z -transformed correlation coefficients (ES_{zr}) for forest amount and sample size (n) to assess publication bias. Dashed line is the summary mean-weighted effect size from random-effect meta-analysis across 330 effect sizes from 63 studies. There was no strong evidence of publication bias since effect sizes were symmetrically distributed around the summary effect, with the exception of two effect sizes at large sample sizes; otherwise, the scatterplot produced a funnel-shape with greater variation in studies at low sample sizes.

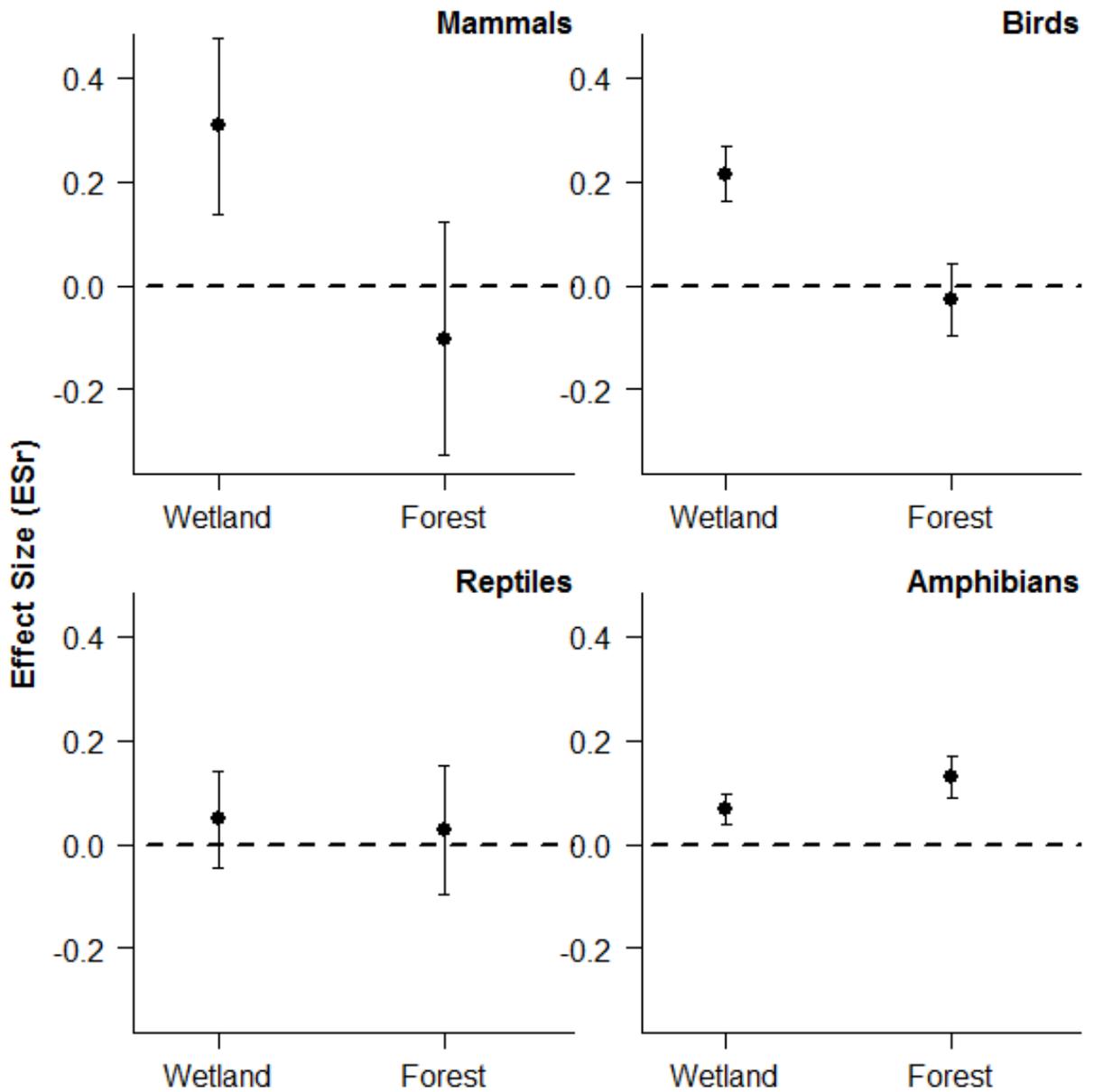


Figure 4.5: Population-level responses of wetland-dependent animal classes to wetland amount and forest amount in a landscape. Points represent mean-weighted effect sizes (z -transformed Pearson correlation coefficients) from mixed-effects meta-regressions and lines indicate 95% confidence intervals.

the weighted-mean effect size of forest amount for mammals and birds was lower than that of amphibians and reptiles (Figure 4.5). Therefore, I tested whether the mean effect size of wetland amount and forest amount were significantly different for each taxonomic class separately.

For mammals, the effects of wetland amount and forest amount were not significantly different ($t = 1.9754$, $df = 6$, $p\text{-value} = 0.096$); however the lack of a significant difference is likely due to the low sample size ($n = 7$). For birds, the effect of wetland amount was more important than the effect of forest amount ($t = 7.578$, $df = 97$, $p < 0.0001$). The correlation between wetland amount and forest amount in a landscape across studies for birds was low ($r = -0.14$, range $-0.56 - 0.80$; Appendix L). The effect size for wetland amount on birds was not related to the correlation between wetland amount and forest amount in a study ($Q_M = 0.06$, $p = 0.81$). Therefore, the larger effect of wetland amount than forest amount on birds was not a statistical artefact of the negative correlation between wetland amount and forest amount in a landscape.

For amphibians, the effect of forest amount was more important than the effect of wetland amount ($t = -4.129$, $df = 204$, $p < 0.0001$). The correlation between wetland amount and forest amount in a landscape across studies for amphibians was low (mean $r = 0.04$, range $-0.84 - 0.69$; Appendix L). The effect size of forest amount was not related to the correlation between wetland amount and forest amount in a study ($Q_M = 3.15$, $p = 0.08$). In other words, the larger, positive effect of forest amount than wetland amount on amphibians was not confounded by a correlation between forest amount and wetland amount. For reptiles, the effects of wetland amount and forest amount were not

significantly different ($t = -0.1681$, $df = 19$, $p = 0.868$), although the sample size was low ($n = 20$).

Because the effects of wetland amount and forest amount varied by taxonomic class, I tested if the relative effects were scale-dependent within each taxon separately. For mammals, there were too few effect sizes to meaningfully test whether the relative effects changed with spatial scale ($n = 7$). For birds, the relative effects of wetland amount and forest amount did not change with spatial scale (from 50 to 5000 m) (slope = 0.060, HPD95lower = -0.075, HPD95upper = 0.145, pMCMC = 0.544, $\Pr(>|t|) = 0.272$). The fitted model for birds showed that wetland amount was more important than forest cover at all spatial scales (Figure 4.6). For amphibians, the relative effects of wetland amount and forest amount did not change with spatial scale (from 100 to 6000 m) (slope = -0.006, HPD95lower = -0.025, HPD95upper = 0.009, pMCMC = 0.412, $\Pr(>|t|) = 0.4018$). The fitted model showed that forest amount was more important than wetland cover at all spatial scales (Figure 4.7). For reptiles, there were too few effect sizes ($n = 20$) to meaningfully test whether the relative effects changed with spatial scale.

Discussion

For wetland birds, my results support my a priori expectation that the amount of wetland in a landscape is more important than landscape matrix quality (Figure 4.5). The effect on bird population abundance of wetland amount was consistently strong and positive across spatial scales (Figure 4.6). This is not surprising because all breeding season activities (foraging, mating, nesting) are carried out at or near the edge of wetland

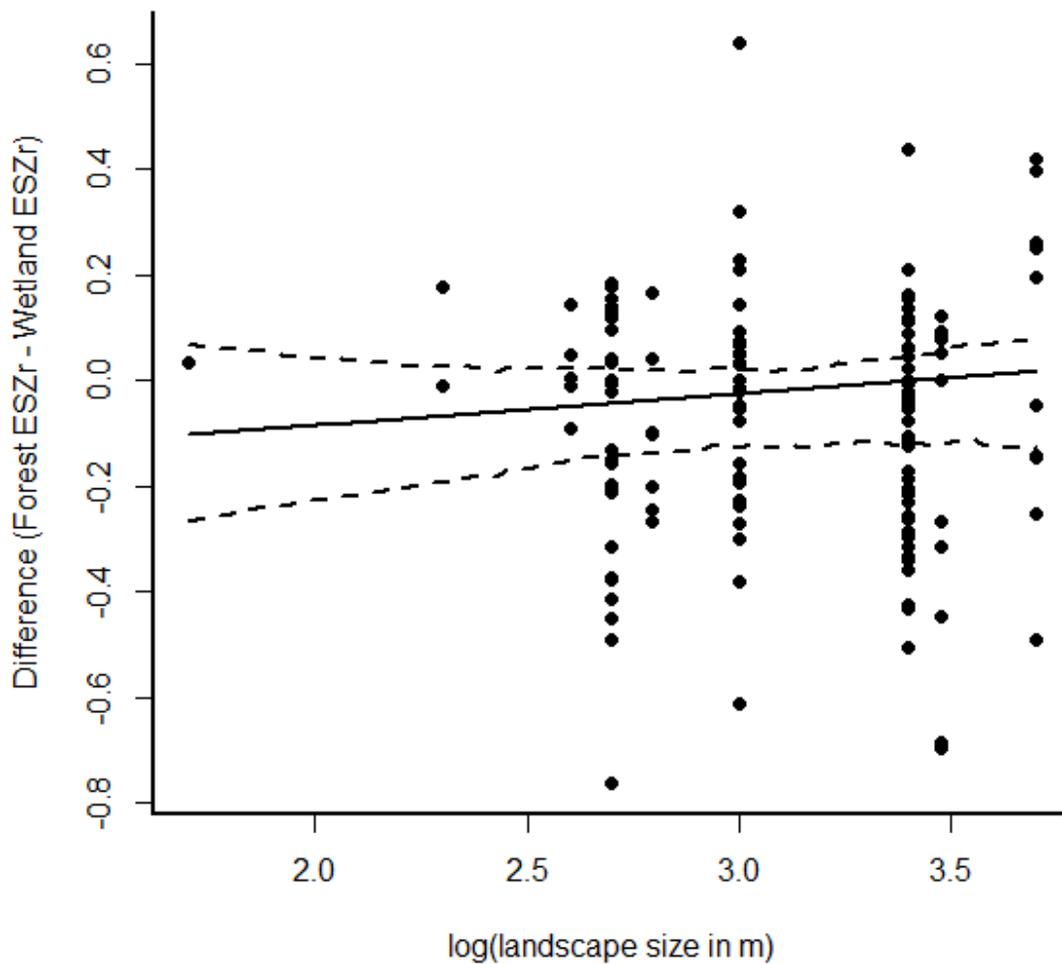


Figure 4.6: Difference in effects of wetland and forest amount ($|ESZ_r, \text{Forest} - ESZ_r, \text{Wetland}|$) on wetland bird population abundance at increasing spatial scales (log transformed). Solid line is the fitted relationship from a mixed-effects model and dashed lines are 95% highest posterior density confidence intervals obtained from Markov chain Monte Carlo simulations.

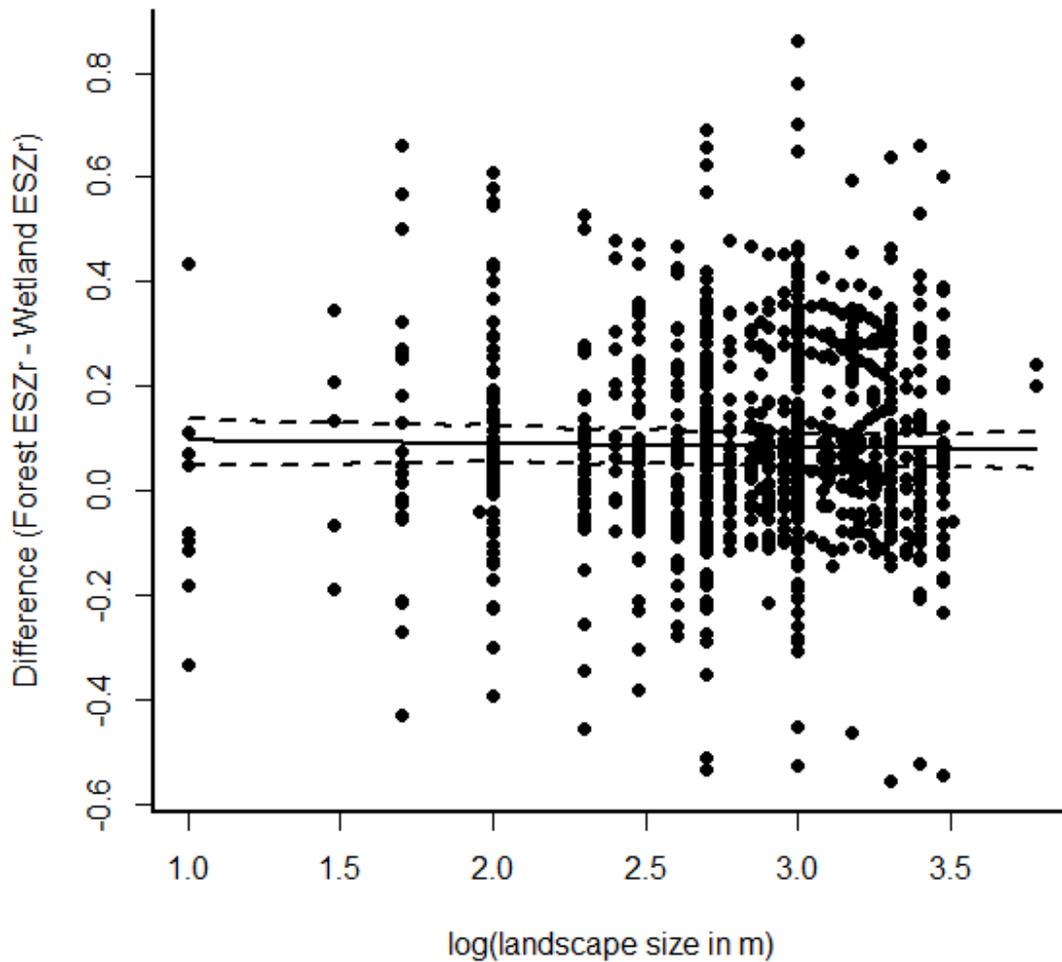


Figure 4.7: Difference in effects of wetland and forest amount ($|ESZ_r, \text{Forest} - ESZ_r, \text{Wetland}|$) on wetland-dependent amphibian population abundance at increasing spatial scales (log transformed). Solid line is the fitted relationship from a mixed-effects model and dashed lines are 95% highest posterior density confidence intervals obtained from Markov chain Monte Carlo simulations.

habitat. The positive effect of wetland amount is most likely due to higher food and nesting site availability in landscapes with more wetland (Tozer et al. 2010). The effect on bird population abundance of forest amount was relatively weak and confidence intervals overlapped zero (Figure 4.5). This suggests that the quality of the surrounding matrix is less important than the total area of wetland in a landscape for wetland birds, and is consistent with previous studies measuring the effects of forest, agriculture, and development in a landscape relative to the effect of wetland amount (Gibbs and Kinkel 1997, Shriver et al. 2004, Valente et al. 2011, Chapter 3). This explanation is plausible because birds do not interact directly with the surrounding matrix during movements between wetlands and therefore dispersal mortality is likely to be low regardless of matrix quality. In support of this notion, a recent meta-analysis found a very weak effect of road and/or traffic density on bird population abundance (Rytwinski and Fahrig 2012).

However, I acknowledge that my measure of landscape matrix quality (forest amount) was not perfect, because matrix quality can depend on the composition of the non-forest part of the matrix. For example, urbanization generally has stronger negative effects on abundance of wetland-dependent species than agriculture (Gagné and Fahrig 2007, Smith and Chow-Fraser 2010). Moreover, Ward et al. (2010) found negative effects of urban development on wetland bird abundance despite no wetland loss. These effects were attributed to changes in hydrology and vegetation structure of the wetland (Ward et al. 2010). Since such changes are one of the main mechanisms by which matrix quality influences local patch biodiversity (Driscoll et al. 2013), I may have underestimated the effect of matrix quality, e.g. urbanization, on wetland biodiversity. On the other hand, in a study designed to separate the independent effects of wetland

amount and urbanization, wetland amount in the surrounding landscape was found to have greater positive effect on wetland vegetation communities and water quality than the negative effect of urbanization (Patenaude et al. in review). Therefore, including urbanization in the measure of landscape matrix quality might increase the effect of matrix quality on birds, but it is unlikely to render matrix quality more important than wetland amount.

For wetland amphibians, my results did not support my a priori expectation that the amount of wetland would be more important than matrix quality; instead, forest amount was the more important predictor (Figure 4.5), at all spatial scales (Figure 4.7). In a qualitative review, Marsh and Trenham (2001) compared the relative effects of wetland amount (measured as distance to nearest pond) to effects of terrestrial habitat amount, measured as distance to forest, or as amount of pasture, shrubland and forest in a landscape. Consistent with my results, Marsh and Trenham (2001) posited that the amount of useable terrestrial habitat in a landscape (equivalent to high-quality landscape matrix in my terminology) was a better predictor of amphibian population dynamics and abundance than wetland amount. To the best of my knowledge, my quantitative review is first to test their hypothesis and support their prediction that processes occurring outside wetland habitat better explain the population distribution of amphibians than wetland availability in a landscape.

Forest amount had a larger effect than wetland amount on amphibian population abundance likely because forest is terrestrial post-breeding habitat for many wetland-breeding amphibians. For those species, forest provides complementary summer foraging, refugia (aestivation) and/or overwintering habitat required to complete their

life-cycle (i.e. juvenile and adult life stages). Given that at least one breeding wetland is available, a landscape with more forest provides more complementary habitat, thereby supporting greater amphibian abundances (i.e. landscape complementation; Dunning et al. 1992). If the availability of complementary habitat provided by forests is limited, such that juvenile and adult stages do not survive to the next breeding season, local population sizes at breeding wetlands will be low despite high wetland amount in the landscape. It is also possible that the density of emerging juveniles in upland forest is higher in the forest that is accessible from the wetland. This higher density might decrease survival, growth and reproductive development of individuals (Harper and Semlitsch 2007, Berven 2009), reducing the overall population growth rate even if there is ample wetland habitat available. This is contrary to the general assumption that wetland breeding habitat has a greater influence on population dynamics of amphibians than terrestrial habitat (reviewed by Marsh and Trenham 2001).

On the other hand, terrestrial habitat use would not explain a larger effect of forest amount than wetland amount on abundance of wetland-breeding amphibian species that do not require forest to complete their life-cycle. To test whether forest amount is important independent of habitat use, I classified each amphibian species included in the meta-analysis as a 'forest species' or 'non-forest species' using habitat accounts in the literature and field guides. I classified any species described as using forest during any part of its juvenile or adult stage as a 'forest species', so that any bias in testing the relative effects of wetland and forest amount was in the direction of wetland amount. I then compared the relative effects of wetland amount and forest amount for just the subset of 'non-forest species'. I found the overall mean effect of forest amount (0.083)

was still larger than the mean effect of wetland amount (0.054), but the difference was non-significant ($t = -0.612$, $df = 64$, $p = 0.543$). This suggests that the amount of forest in a landscape has a positive effect on amphibian abundance beyond providing terrestrial habitat.

A second possible reason for the greater effect on amphibian population abundance of forest amount than wetland amount is the importance of dispersal mortality in affecting population abundance. Experimental field studies have shown that many amphibians, including several ‘non-forest species’, orient towards and prefer to move through forest during emigration from the breeding pond, avoiding open areas such as fields, pastures and clearcuts (Rothermel and Semlitsch 2002, Patrick et al. 2006, Todd et al. 2009, Pittman and Semlitsch 2013). Moreover, these studies show that individuals move significantly greater distances in forest compared to open areas presumably due to higher mortality in open areas (Rothermel and Semlitsch 2002, Popescu and Hunter 2011). Amphibians experience significantly lower desiccation rates in forests than in fields (Rothermel and Semlitsch 2002) because canopy cover and leaf litter lowers temperature and increases humidity near the ground (Popescu and Hunter 2011). For example, wetland-breeding salamanders have significantly lower survival rates in clearcuts than in unharvested controls (Todd et al. 2014). An increase in dispersal mortality with decreasing forest cover the landscape could lead to lower local (e.g. pond) population sizes despite a high amount of wetland habitat.

In fact, demographic models for a range of wetland-breeding amphibians predict that factors that increase mortality during the post-breeding stage, such as loss of post-breeding terrestrial habitat or high dispersal mortality, affect amphibian population

dynamics more than factors that increase pre-metamorphic mortality, such as wetland loss (Biek et al. 2002, Vonesh and Cruz 2002, Salice et al. 2011). Since it is likely that reductions in the amount of surrounding forest in a landscape decrease survival rates of juveniles and adults, a larger effect on amphibian abundance of forest amount than wetland amount is therefore not surprising.

On the other hand, a strong positive effect of forest cover in a landscape could occur through a correlation between forest amount and another landscape variable. For example, the amount of forest cover in a landscape is generally negatively correlated with road density (Findlay and Houlihan 1997). Since road and/or traffic density has strong negative effects on amphibian population abundance (Rytwinski and Fahrig 2012), it is possible that the positive effect of forest amount is simply an artefact of the forest vs. road correlation (i.e. amphibians respond positively to landscapes with low road density). However, in previous studies on wetland biodiversity, when forest amount and road density are included in a model together (i.e. statistically controlled), both variables have near-significant effects (Findlay and Houlihan 1997, Houlihan and Findlay 2003), suggesting forest and roads each have independent effects. This was confirmed in a study by Eigenbrod et al. (2008) who selected sample landscapes such that forest amount and traffic density were not correlated. Eigenbrod et al. (2008) found that the independent effect of forest cover on wetland-breeding anuran populations was at least as large as the effect of traffic, but that the relative effects of these two variables varied with species. Therefore, I am confident that the strong, positive effect of forest amount I detected in my meta-analysis is not a statistical artefact of a negative effect of roads.

For wetland-dependent reptiles, my results suggest that populations of freshwater turtles and watersnakes are limited by neither the distribution of wetlands nor the amount of high quality matrix in a landscape (Figure 4.5), indicating that some other factor limits their distributions. I suggest that access to complementary resources (landscape complementation) is limiting reptile populations. Like wetland-breeding amphibians, wetland reptiles require different habitats to complete their life-cycle, which necessitates seasonal overland movements outside of wetlands. All freshwater turtles require terrestrial nesting habitat (Steen et al. 2012) and watersnakes (and some turtles) typically overwinter in terrestrial habitats (Roe et al. 2003). However, these complementary habitats are usually fine-scale features not captured in coarse-scale resolution (>30 m) landcover data such as forest cover. For example, Marchand and Litvaitis (2004) found a positive relationship between the area of suitable nesting habitat within 30 m of wetlands and painted turtle abundance for wetlands along gradient of forest amount.

It is also possible that road mortality may limit local reptile populations in landscapes with high road and/or traffic density despite high wetland amount, high quality matrix (i.e. forest) or availability of nesting sites or other complementary habitats. As road density increases in a landscape, it is unlikely that all complementary habitat occurs within a roadless area. In some landscapes, this could force all individuals of a local population to cross roads, resulting in very high mortality rates (e.g. Aresco 2005). Reptiles have delayed sexual maturity and low reproductive rates, which means their populations recover slowly from adult mortality events. In fact, nesting migrations increase female turtle vulnerability to road mortality, which is considered to be the most significant threat to freshwater turtle population persistence (Steen et al. 2012).

Similarly, road mortality is suggested to significantly contribute to watersnake declines (Roe et al. 2006), with females possibly more susceptible due to greater movements in search of parturition sites (Attum et al. 2007). In line with my results, a recent meta-analysis found that amphibians and reptiles were more susceptible to the negative effects of roads than mammals and birds (Rywinski and Fahrig 2012).

My review suggests that the distribution of wetland habitat in a landscape is not limiting to wetland-dependent amphibians or reptiles. This is surprising since metapopulation structure, based on the assumption that wetland patches contain local populations, is widely assumed to apply to the population dynamics of these animals (e.g. Werner et al. 2009, Cosentino et al. 2010, Heard et al. 2012). In contrast, a meta-analysis by Prugh et al. (2008) found that amphibians and reptiles in general have relatively weak responses to patch area effects compared to birds and mammals. As discussed above, unlike wetland-dependent birds and mammals, many amphibians and reptiles require different habitat types to complete their life cycles, which necessitates regular overland movements, increasing the importance of matrix quality (e.g. forest cover, roads). In fact, if the availability of complementary habitat near a given wetland is low, amphibians must move large distances to find them (Laan and Verboom 1990, Reh and Seitz 1990), and similarly turtles will need to move longer distances to reach nesting habitat (Baldwin et al. 2004) or overwintering sites (Harden et al. 2009). These longer movements increase the probability of dispersal mortality due to factors such as desiccation, predation or road mortality. My results imply that the population distribution of wetland-dependent amphibians and reptiles is more strongly related to landscape matrix quality

than to the availability of wetland in a landscape. Specifically, the quantity of and/or access to complementary terrestrial habitat is limiting.

Conclusions

Although the species I included in my meta-analysis are all “wetland-dependent”, my results show that this characterization can be misleading for conservation management, particularly for amphibians and reptiles. For wetland-dependent birds, and likely for mammals, my results suggest that increasing the amount of wetland in a landscape at any spatial scale will have a larger effect on local population abundance than improving matrix quality. In contrast, for wetland-dependent amphibians and reptiles, my results suggest that improving landscape matrix quality surrounding wetlands at any spatial scale will have a larger effect on local abundance than increasing wetland amount. For amphibians this means increasing the amount of complementary habitats (forest, meadows) adjacent to wetlands, for reptiles it means improving access to complementary resources (nesting or overwintering areas), and for both amphibians and reptiles it means limiting road development and preventing animals from attempting to cross roads, for example using barriers. Overall, my meta-analysis supports the wetland-based conservation approach for wetland-dependent birds (and likely mammals), but challenges the efficacy of this approach for wetland-dependent amphibians and reptiles. In fact, the first priority for conservation management of these species should be improving access to complementary terrestrial habitats. Management strategies that focus mainly on wetland habitat likely often overestimate the capacity of a landscape to support populations of

wetland-dependent amphibians and reptiles. In fact, I suggest that we should stop thinking of these species as wetland species since that designation is harmful to their persistence.

Chapter 5: General Discussion

The objective of my dissertation was to increase our knowledge of the effects of landscape structure on wetland-dependent vertebrate populations and the life-history traits that determine these responses. In doing this research, I also aimed to contribute to landscape ecology theory and provide conservation recommendations for wetland biodiversity in general.

In Chapter 2, I found that species with lower reproductive rates are more sensitive to habitat loss than species with higher reproductive rates. I used meta-analysis to show that the effect of reproductive rate is generalizable across taxa and different landscape types around the world. This result highlights the importance of reproductive rate in determining the amount of habitat required in a landscape for population persistence, which I believe is underappreciated in the landscape ecology literature. Although reproductive rate was theoretically found to have the largest effect on the amount of habitat required in a landscape for population persistence compared to that of mortality or movement (Fahrig 2001), to my knowledge there are only two empirical tests of this relationship (Vance et al. 2003, Holland et al. 2005). In fact, reproductive rate is generally not considered in reviews of species vulnerability to habitat loss and fragmentation (e.g. Ewers and Didham 2006). In contrast, dispersal ability is widely considered to strongly influence species tolerance to habitat loss. This is primarily based on the assumption that dispersal ability determines the probability of colonization. However, my results suggest that colonization rates are more related to reproductive rate because it determines the total number of potential colonists. Overall, my results bring further empirical evidence to the long-held belief in ecology and conservation biology

that species with low reproductive potential are more prone to extinction due to habitat loss than species with high reproductive potential.

Results of Chapter 2 suggest several other research questions. *Do other life-history traits influence species response to habitat loss?* As reviewed in the General Introduction, there were several traits that I initially hoped to include in my meta-analysis but were excluded due to insufficient data for wetland vertebrates. Of particular interest is the effect of habitat breadth; high habitat or resource specialization has been shown to increase vertebrate sensitivity to habitat loss (Swihart et al. 2006, Newbold et al. 2012). Moreover, testing for potential synergistic interactions between reproductive rate and habitat breadth may indicate highly vulnerable species groups, for example habitat specialists with low reproductive rates. *Also, do species with high reproductive rates dominate in landscapes with low habitat amount compared to high habitat amount landscapes?* Landscape structure is hypothesized to select functionally important traits, resulting in filtering of species communities or ‘species sorting’ (reviewed in Tschamke et al. 2012).

In Chapter 3, I found that habitat amount was more important than other landscape-scale factors for bird distributions, whereas the amount of forest in the surrounding landscape was more important for turtle distributions. This study is unique in that it compared the independent, relative effects of five major landscape-scale threats to wetland species by using a study design that minimized the correlations between landscape variables while maximizing the range in each variable. My results contribute to landscape ecology theory by corroborating that habitat fragmentation has a relatively weak effect on species distributions when measured independently of confounding

landscape variables, such as habitat amount and matrix composition, for less-studied taxa (i.e. wetland birds and turtles). Notably, a measure of matrix composition (forest cover) had a larger effect than habitat amount, which is not consistent with recent reviews in landscape ecology (e.g. Prevedello and Vieira 2010). This result prompted me to develop a second meta-analysis in Chapter 4 to investigate whether a similar pattern is found when synthesizing the results across a set of studies conducted under a variety of circumstances and whether this pattern was similar for other taxa.

In Chapter 4, I confirmed the results of Chapter 3 that response to landscape structure varied among taxa. I found that wetland amount is more important than other landscape-scale factors for wetland bird populations. This is likely also true for wetland mammals, given the large magnitude of the effect size; however, this needs to be confirmed by a larger sample size. In contrast, I found that matrix quality (indexed as forest cover) was more important than wetland amount for amphibian populations. Most surprisingly, the effect of wetland amount and forest cover both had very weak effects on reptile populations; however, this may be limited by the small sample size. These results contribute to ecological theory by confirming that wetland loss is the primary landscape-scale factor of wetland bird declines (and likely wetland mammals), but suggests that the amount of wetland in a landscape is not limiting to amphibians and reptiles. In fact, forest loss may have a greater role in amphibian declines than previously realized, even for species that do not require forest during their life-cycle. This has important implications for amphibian ecology as it provides (to the best of my knowledge) the first empirical test of Marsh and Trenham's (2001) prediction that "pond occupancy may be

more indicative of the spatial arrangement of terrestrial habitat than the arrangement of breeding ponds”.

My finding that the surrounding landscape matrix was more important than wetland amount for reptile (Chapter 3) and amphibian (Chapter 4) populations has important theoretical and conservation implications. First, my thesis provides further empirical evidence to support that “the matrix matters” (Watling et al. 2011) more than habitat loss for some species groups. The degree to which the surrounding matrix affects population persistence is considered an important research area in landscape ecology (Tscharnke et al. 2012). Second, my thesis suggests that these organisms are more strongly related to landscape matrix quality because the availability of and/or access to complementary terrestrial habitat (i.e. landscape complementation) is limiting to their populations. Lastly, my thesis challenges conservation policies that focus only on wetlands and/or wetland buffers to protect wetland-dependent amphibians and reptiles. In fact, the first priority for conservation management of these species should be improving access to complementary resources.

Several questions related to the results of Chapter 3 and 4 deserve attention in future studies. First, *what is the underlying process causing the large, positive effect of forest cover in the surrounding landscape on herptofauna populations?* Because both chapters are correlative studies, I cannot say for certain that the relationship between forest amount in a landscape and turtle occurrence (Chapter 2), as well as amphibian population abundance (Chapter 3) was caused by decreased dispersal mortality as I hypothesized. An alternative explanation is that wetlands in landscapes with less forest have altered hydrology (e.g. lower water levels) and decreased water quality (e.g.

increased sedimentation or contaminants) (Richardson and McCarthy 1994), which are shown to have a negative impact on abundance of amphibians (e.g. Houlihan and Findlay 2003) and reptiles (e.g. DeCatanzaro and Chow-Fraser 2010). Further investigation of the causal factor underlying the positive relationship between the amount of forest surrounding wetlands and herptofauna populations would inform conservation actions.

Second, *what are the relative effects of habitat loss (wetland amount and/or forest amount) and road and/or traffic density on wetland vertebrate populations?* In Chapter 4, I found that the effect on population abundance of both wetland and forest amount in a landscape was relatively weak for amphibians and reptiles compared to the large effect of wetland amount on mammals and birds (Figure 4.5). This result indicates that some other factor is limiting these organisms. I hypothesize that the population distribution of amphibians and reptiles are more strongly limited by road mortality than the amount of wetland or forest in a landscape. In support of this idea, a recent meta-analysis found that amphibians and reptiles were more susceptible to the negative effects of road and/or traffic density than mammals and birds (Rytwinski and Fahrig 2012). On the other hand, a more accurate measure of habitat amount for amphibians (e.g. area of wetland and forest in a landscape combined into a single predictor) could be more important than roads and/or traffic density. However, the combined effect size of wetland and forest amount for amphibians ($ES_{Zr} = 0.177$) from Chapter 4 is lower in magnitude than that of road and/or traffic density ($ES_{Zr} = -0.302$) reported in Rytwinski and Fahrig (2012), but note that this is purely conjectural. Repeating the methods in Chapter 4 to obtain information on road density in a landscape for each study to compare relative effects using meta-analysis would be interesting.

Third, *are the relative effects of landscape structure variables scale-dependent?* I show in Chapter 4 that the relative effects of wetland amount and forest amount on wetland-dependent vertebrates are not scale-dependent over more than one order of magnitude in landscape radius (e.g. 50 to 5000 m, for birds). This result is an important contribution to landscape ecology because, to my knowledge, only a few empirical studies have investigated whether relative effects of landscape variables change with scale (Krawchuk and Taylor 2003, Smith et al. 2011). These studies suggest that different landscape variables can affect the same ecological response most strongly at different spatial scales, such that their relative effects are scale-dependent. Smith et al. (2011) assessed the effects of landscape structure on the abundance of two forest birds in Ontario and found that the relative effects of habitat amount and fragmentation were scale-dependent; however, consistent with the results of my study, the relative effects of habitat amount and matrix quality (indexed as road density in a landscape) did not change with scale. Taken together, these results suggest that scale-dependency in the relative effects of habitat amount and matrix quality may generally not occur. This has an important conservation message – the ranking of landscape conservation options does not depend on the scale of influence of given managing organization (e.g. landowner, municipal or regional jurisdiction). For example, to increase local population of wetland birds, my results suggest that increasing habitat amount at any spatial scale, small or large, will have a larger effect than improvements to matrix quality. The generality of scale-independent relative effects warrants further research.

Lastly, *does increasing landscape heterogeneity increase biodiversity of wetland vertebrates?* It is now recognized in landscape ecology that many species and species

groups perceive landscapes in more complex ways than simply habitat-matrix, as I have used in my thesis, because resources are obtained from many different cover types (e.g. food, nesting, dispersal). Therefore, to better understand the effects of landscape structure on wetland biodiversity, I suggest that further research use a heterogeneous landscape view. Landscape heterogeneity describes the number and proportion of different cover types (compositional heterogeneity) and the spatial arrangement of cover types (configurational heterogeneity) (Fahrig et al. 2011). I predict that increased compositional heterogeneity of natural cover types (i.e. wetland and complementary habitat for herptofauna, such as forest and meadow) in a landscape should increase wetland biodiversity because a greater number of cover types should result in higher landscape complementation, but that this would reach a maximum at intermediate heterogeneity values based on the intermediate landscape heterogeneity hypothesis (Fahrig et al. 2011), which predicts that high landscape heterogeneity decreases biodiversity due to loss of habitat and decreases in patch size.

More generally, my thesis demonstrates the importance of estimating the relative effects of multiple threats to declining species, so that competing management options can be prioritized. Without the context of relative effects, I would not have detected the importance of matrix quality over wetland amount; instead, I would have added to and/or confirmed the list of threats to wetland biodiversity, which has little to contribute to conservation practice. Moreover, what is unique about my thesis is that I have used meta-analysis to provide generality in relative impacts of landscape-scale threats across different taxa as well as landscape types around the world. With limited conservation resources, it is impossible to address the multiple factors that are linked to population

declines. I argue, as have many others (e.g. Lindenmayer and Fischer 2007, Wilson et al. 2007, Smith et al. 2009, Hodgson et al. 2011, Crawford et al. 2014), that it is more effective to focus management on the threat with the largest impact on population persistence. Although my thesis focuses entirely on landscape-scale threats to wetland biodiversity, which is not a complete list, I consider one of the largest threats – habitat loss – but I suggest that this threat should be compared with other factors, such as road mortality, especially in the case of amphibian and reptiles.

A common finding throughout all chapters of my thesis is that different taxonomic groups respond to landscape structure differently. This is consistent with the general observation in ecology that there are low correlations between taxa in their responses to landscape change (Wolters et al. 2006). My results suggest that these low correlations could be due to differences in landscape-scale requirements. Given these differences in responses to landscape structure, the practice of using one taxon (often birds) as a biodiversity model for landscape planning is likely to fail for other groups (Cushman et al. 2010). My thesis highlights the need for a multi-taxa approach to landscape conservation planning, even for a group of species identified by their requirement for wetlands during their life-cycle (here “wetland-dependent species”). In fact, the very notion of classifying species by one habitat type (here wetlands) for conservation planning can be potentially harmful to their persistence, as I argue for amphibians and reptiles in Chapter 4. Although the results of Chapters 3 and 4 have identified the largest landscape factor of population distributions and corresponding conservation action for each taxonomic group, the challenge remains how to integrate these different landscape-scale requirements for multi-species conservation planning.

I believe my thesis can make some general conservation recommendations for wetland biodiversity that apply across vertebrate taxa. Results from Chapter 2 recommend that conservation priority should be placed on species with low reproductive rates. Moreover, conservation plans for declining wetland species should focus on managing habitat variables that increase reproductive output, rather than movement, of target species. For example, maintaining an equal proportion of open water: vegetation (i.e. hemi-marsh) for wetland birds such as least bittern, or the removal of predatory fish from farm ponds for amphibians. Results from Chapter 3 and 4 in general suggest that increasing wetland amount and/or landscape complementation at any management scale will have a larger effect on wetland biodiversity than management of wetland configuration (i.e. fragmentation per se) in a landscape. To maintain regional wetland biodiversity, I would recommend a mosaic of landscapes that vary in both the amount of wetland and in the amount of complementary habitats for amphibians and reptiles (e.g. forests, open habitats) surrounding wetlands. In other words, landscapes with varying levels of compositional landscape heterogeneity. Ideally, this would include landscapes with a high amount of wetland to maintain those species with very low reproductive rates, and landscapes with a high amount of different complementary habitats for amphibians and reptiles. In addition, there should be at least some landscapes with little to no road development, and all landscapes should have road mitigation structures to prevent animals from attempting to cross roads in key movement areas. Specifically, this would be complete fencing of the road with a solid barrier in conjunction with wildlife crossing structures (ecopassages).

Conclusion

Wetland protection alone does not protect wetland biodiversity. My results show that different wetland-dependent taxa respond to different aspects of landscape structure. My results imply that mammals and birds are more limited by the amount of wetland habitat in a landscape, whereas amphibians and reptiles are more limited by the surrounding landscape matrix quality, likely due to landscape complementation. I suggest that regional landscape heterogeneity could optimize conservation for wetland biodiversity. However, this will likely only be effective if some landscapes contain a high amount of wetland habitat for species with low reproductive rates, and if some landscapes contain no road development, so that road mortality does not decimate populations of amphibians and reptiles.

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Appendix A: Studies included in the meta-analysis and associated species, effect sizes (ES_r), adjusted sample sizes (n), and study design categories (study type, sampling effort, and patch area) (Chapter 2).

Study*	Country	Taxa ⁺	Species	ES _r	n	Study Type	Sampling Effort	Patch Area
1	Canada	m	<i>Castor canadensis</i>	0.28	24	Amount	Independent	no
2	USA	m	<i>Oryzomys palustris</i>	0.19	35	Amount	Dependent	yes
3	USA	m	<i>Microtus pennsylvanicus</i>	0.06	15	Amount	Dependent	yes
3	USA	m	<i>Sorex fumeus</i>	0.31	15	Amount	Dependent	yes
3	USA	m	<i>Synaptomys cooperi</i>	0.11	15	Amount	Dependent	yes
4	USA	m	<i>Sylvilagus palustris hefneri</i>	0.3	39	Amount	Dependent	yes
5	USA	m	<i>Neofiber alleni</i>	0.42	453	Configuration	Dependent	no
6	Japan	b	<i>Egretta intermedia</i>	0.14	32	Amount	Independent	yes
6	Japan	b	<i>Nycticorax nycticorax</i>	0.43	32	Amount	Independent	yes
6	Japan	b	<i>Ardea cinerea</i>	0.4	32	Amount	Independent	yes
7	USA	b	<i>Botaurus lentiginosus</i>	0.12	50	Amount	Dependent	yes
7	USA	b	<i>Ixobrychus exilis</i>	0.22	50	Amount	Dependent	yes
8	USA	b	<i>Botaurus lentiginosus</i>	0.14	83	Amount	Independent	yes
8	USA	b	<i>Podilymbus podiceps</i>	0.1	190	Amount	Independent	yes
8	USA	b	<i>Fulica americana</i>	0.17	190	Amount	Independent	yes
8	USA	b	<i>Porzana carolina</i>	0.09	190	Amount	Independent	yes
8	USA	b	<i>Ixobrychus exilis</i>	0.21	79	Amount	Independent	yes
9	Spain	b	<i>Circus aeruginosus</i>	0.15	35	Configuration	Unknown	no
10	Netherlands	b	<i>Acrocephalus scirpaceus</i>	0.09	316	Amount	Dependent	no

11	USA	b	<i>Ardea herodia</i>	0.64	29	Amount	Dependent	no
12	USA	b	<i>Ardea herodia</i>	0.28	29	Amount	Independent	no
13	USA	b	<i>Butorides virescens</i>	-0.17	5	Configuration	Dependent	no
13	USA	b	<i>Gallinago delicata</i>	0.12	5	Configuration	Dependent	no
13	USA	b	<i>Circus cyaneus</i>	0.04	5	Configuration	Dependent	no
13	USA	b	<i>Botaurus lentiginosus</i>	-0.15	5	Configuration	Dependent	no
13	USA	b	<i>Podilymbus podiceps</i>	-0.1	5	Configuration	Dependent	no
13	USA	b	<i>Rallus limicola</i>	-0.19	5	Configuration	Dependent	no
13	USA	b	<i>Porzana carolina</i>	-0.02	5	Configuration	Dependent	no
14	UK	b	<i>Botaurus stellaris</i>	0.15	44	Amount	Independent	yes
15	USA	b	<i>Cistothorus palustris</i>	0.09	11	Amount	Independent	yes
15	USA	b	<i>Agelaius phoeniceus</i>	0.17	11	Amount	Dependent	yes
15	USA	b	<i>Cistothorus platensis</i>	-0.4	11	Amount	Independent	yes
16	Brazil	b	<i>Plegadis chihi</i>	0.4	9	Amount	Dependent	yes
16	Brazil	b	<i>Platalea ajaja</i>	0.22	5	Amount	Dependent	yes
16	Brazil	b	<i>Theristicus caerulescens</i>	0.47	5	Amount	Dependent	yes
16	Brazil	b	<i>Nycticorax nycticorax</i>	0.06	32	Amount	Dependent	yes
16	Brazil	b	<i>Egretta thula</i>	0.25	9	Amount	Dependent	yes
16	Brazil	b	<i>Ardea cocoi</i>	0.53	6	Amount	Dependent	yes
16	Brazil	b	<i>Mycteria americana</i>	0.19	5	Amount	Dependent	yes
16	Brazil	b	<i>Syrigma sibilatrix</i>	0.07	5	Amount	Dependent	yes
16	Brazil	b	<i>Ciconia maguari</i>	0.42	5	Amount	Dependent	yes
16	Brazil	b	<i>Phimosus infuscatus</i>	0.51	5	Amount	Dependent	yes
16	Brazil	b	<i>Butorides striata</i>	0.25	5	Amount	Dependent	yes
16	Brazil	b	<i>Circus buffoni</i>	0.13	5	Amount	Dependent	yes

16	Brazil	b	<i>Ardea alba</i>	0.08	6	Amount	Dependent	yes
16	Brazil	b	<i>Chauna torquata</i>	0.17	5	Amount	Dependent	yes
16	Brazil	b	<i>Gallinago paraguaiiae</i>	0.32	5	Amount	Dependent	yes
16	Brazil	b	<i>Himantopus himantopus</i>	0.13	42	Amount	Dependent	yes
16	Brazil	b	<i>Pardirallus sanguinolentus</i>	0.18	5	Amount	Dependent	yes
16	Brazil	b	<i>Gallinula melanops</i>	-0.1	5	Amount	Dependent	yes
16	Brazil	b	<i>Rollandia rolland</i>	0.43	42	Amount	Dependent	yes
16	Brazil	b	<i>Jacana jacana</i>	0.43	42	Amount	Dependent	yes
16	Brazil	b	<i>Anas flavirostris</i>	0.28	5	Amount	Dependent	yes
16	Brazil	b	<i>Amazonetta brasiliensis</i>	0.43	5	Amount	Dependent	yes
16	Brazil	b	<i>Anas versicolor</i>	0.22	5	Amount	Dependent	yes
16	Brazil	b	<i>Dendrocygna viduata</i>	0.27	5	Amount	Dependent	yes
16	Brazil	b	<i>Callonetta leucophrys</i>	0.12	5	Amount	Dependent	yes
16	Brazil	b	<i>Rostrhamus sociabilis</i>	0.26	9	Amount	Dependent	yes
16	Brazil	b	<i>Dendrocygna bicolor</i>	0.22	5	Amount	Dependent	yes
16	Brazil	b	<i>Netta peposaca</i>	0.35	5	Amount	Dependent	yes
16	Brazil	b	<i>Aramus guarauna</i>	0.05	35	Amount	Dependent	yes
16	Brazil	b	<i>Fulica leucoptera</i>	0.02	5	Amount	Dependent	yes
16	Brazil	b	<i>Gallinula galeata</i>	0.58	35	Amount	Dependent	yes
16	Brazil	b	<i>Aramides ypecaha</i>	0.09	5	Amount	Dependent	yes
16	Brazil	b	<i>Podilymbus podiceps</i>	0.06	35	Amount	Dependent	yes
17	Canada	b	<i>Botaurus lentiginosus</i>	0.27	141	Amount	Dependent	yes
17	Canada	b	<i>Ixobrychus exilis</i>	0.47	141	Amount	Independent	yes
17	Canada	b	<i>Podilymbus podiceps</i>	0.3	141	Amount	Independent	yes
17	Canada	b	<i>Rallus limicola</i>	0.2	141	Amount	Independent	yes

17	Canada	b	<i>Porzana carolina</i>	0.11	141	Amount	Independent	yes
18	USA	b	<i>Ardea alba</i>	0.67	23	Amount	Independent	yes
18	USA	b	<i>Ardea herodias</i>	0.39	32	Amount	Independent	yes
19	Uganda	b	<i>Serinus koliensis</i>	0.35	17	Configuration	Dependent	no
19	Uganda	b	<i>Laniarius mufumbiri</i>	0.01	17	Configuration	Dependent	no
19	Uganda	b	<i>Chloropeta gracilirostris</i>	0.01	17	Configuration	Dependent	no
19	Uganda	b	<i>Acrocephalus rufescens</i>	0.36	17	Configuration	Dependent	no
19	Uganda	b	<i>Bradypterus carpalis</i>	0.28	17	Configuration	Dependent	no
19	Uganda	b	<i>Cisticola carruthersi</i>	0.18	17	Configuration	Dependent	no
20	USA	b	<i>Circus cyaneus</i>	-0.26	17	Amount	Independent	yes
21	Spain	b	<i>Circus aeruginosus</i>	0.36	7	Configuration	Dependent	no
21	Spain	b	<i>Acrocephalus arundinaceus</i>	-0.34	13	Configuration	Dependent	no
21	Spain	b	<i>Remiz pendulinus</i>	0.36	14	Configuration	Dependent	no
21	Spain	b	<i>Cettia cetti</i>	0.43	11	Configuration	Dependent	no
21	Spain	b	<i>Acrocephalus scirpaceus</i>	-0.03	15	Configuration	Dependent	no
21	Spain	b	<i>Rallus aquaticus</i>	-0.33	15	Configuration	Dependent	no
22	USA	b	<i>Limosa fedoa</i>	0.12	118	Amount	Dependent	yes
22	USA	b	<i>Phalaropus tricolor</i>	0.09	118	Amount	Dependent	yes
22	USA	b	<i>Recurvirostra americana</i>	0.11	118	Amount	Dependent	yes
22	USA	b	<i>Tringa semipalmata</i>	0.11	118	Amount	Dependent	yes
23	USA	b	<i>Ixobrychus exilis</i>	0.07	5	Amount	Dependent	no
23	USA	b	<i>Porphyrio martinica</i>	0.45	5	Amount	Dependent	no
23	USA	b	<i>Gallinula galeata</i>	0.4	5	Amount	Dependent	no
23	USA	b	<i>Rallus elegans</i>	0.29	5	Amount	Dependent	no
23	USA	b	<i>Dendrocygna bicolor</i>	0.09	5	Amount	Dependent	no

24	Canada	b	<i>Ixobrychus exilis</i>	0.24	70	Amount	Dependent	yes
24	Canada	b	<i>Melospiza georgiana</i>	0.24	70	Amount	Dependent	yes
24	Canada	b	<i>Setophaga petechia</i>	0.07	70	Amount	Dependent	yes
24	Canada	b	<i>Cistothorus palustris</i>	0.52	70	Amount	Dependent	yes
24	Canada	b	<i>Podilymbus podiceps</i>	0.3	70	Amount	Dependent	yes
24	Canada	b	<i>Rallus limicola</i>	0.29	70	Amount	Dependent	yes
24	Canada	b	<i>Geothlypis trichas</i>	0.14	70	Amount	Dependent	yes
25	USA	b	<i>Botaurus lentiginosus</i>	0.29	10	Amount	Independent	yes
25	USA	b	<i>Ixobrychus exilis</i>	0.03	15	Amount	Independent	yes
25	USA	b	<i>Porzana carolina</i>	0.18	16	Amount	Independent	yes
25	USA	b	<i>Rallus limicola</i>	0.14	16	Amount	Independent	yes
25	USA	b	<i>Podilymbus podiceps</i>	0.75	16	Amount	Independent	yes
26	Spain	b	<i>Himantopus himantopus</i>	0.24	21	Configuration	Dependent	no
26	Spain	b	<i>Tachybaptus ruficollis</i>	0.24	21	Configuration	Dependent	no
27	Thailand	b	<i>Limnodromus semipalmatus</i>	0.38	20	Amount	Independent	yes
27	Thailand	b	<i>Charadrius mongolus/C. leschenaultii</i>	0.29	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris canutus</i>	0.19	20	Amount	Independent	yes
27	Thailand	b	<i>Recurvirostra avosetta</i>	0.19	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris ferruginea</i>	0.16	20	Amount	Independent	yes
27	Thailand	b	<i>Actitis hypoleucos</i>	0.03	20	Amount	Independent	yes
27	Thailand	b	<i>Arenaria interpres</i>	0.19	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris ruficollis</i>	0.38	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris subminuta</i>	0.7	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris tenuirostris</i>	0.11	20	Amount	Independent	yes
27	Thailand	b	<i>Charadrius dubius</i>	0.36	20	Amount	Independent	yes

27	Thailand	b	<i>Eurynorhynchus pygmeus</i>	0.16	20	Amount	Independent	yes
27	Thailand	b	<i>Gallinago gallinago</i>	0	20	Amount	Independent	yes
27	Thailand	b	<i>Limicola falcinellus</i>	0.22	20	Amount	Independent	yes
27	Thailand	b	<i>Limosa lapponica</i>	0.16	20	Amount	Independent	yes
27	Thailand	b	<i>Limosa limosa</i>	-0.08	20	Amount	Independent	yes
27	Thailand	b	<i>Numenius arquata</i>	0.1	20	Amount	Independent	yes
27	Thailand	b	<i>Numenius madagascariensis</i>	0.19	20	Amount	Independent	yes
27	Thailand	b	<i>Numenius phaeopus</i>	-0.13	20	Amount	Independent	yes
27	Thailand	b	<i>Philomachus pugnax</i>	0.27	20	Amount	Independent	yes
27	Thailand	b	<i>Pluvialis fulva</i>	0.1	20	Amount	Independent	yes
27	Thailand	b	<i>Pluvialis squatarola</i>	-0.17	20	Amount	Independent	yes
27	Thailand	b	<i>Tringa erythropus</i>	0.48	20	Amount	Independent	yes
27	Thailand	b	<i>Tringa glareola</i>	-0.13	20	Amount	Independent	yes
27	Thailand	b	<i>Tringa nebularia</i>	0.19	20	Amount	Independent	yes
27	Thailand	b	<i>Tringa stagnatilis</i>	0.36	20	Amount	Independent	yes
27	Thailand	b	<i>Tringa totanus</i>	0.11	20	Amount	Independent	yes
27	Thailand	b	<i>Xenus cinereus</i>	0.01	20	Amount	Independent	yes
27	Thailand	b	<i>Charadrius alexandrinus</i>	0.17	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris alba</i>	0.25	20	Amount	Independent	yes
27	Thailand	b	<i>Calidris temminckii</i>	0.42	20	Amount	Independent	yes
28	USA	b	<i>Calidris alpina</i>	0.46	5	Amount	Dependent	yes
29	France	b	<i>Nycticorax nycticorax</i>	0.2	25	Amount	Dependent	yes
29	France	b	<i>Egretta garzetta</i>	-0.49	25	Amount	Dependent	yes
29	France	b	<i>Bubulcus ibis</i>	0.37	25	Amount	Dependent	yes
29	France	b	<i>Ardeola ralloides</i>	0.26	25	Amount	Dependent	yes

30	Canada	b	<i>Ixobrychus exilis</i>	0.64	15	Amount	Independent	no
30	Canada	b	<i>Melospiza georgiana</i>	0.47	15	Amount	Independent	no
30	Canada	b	<i>Cistothorus palustris</i>	0.61	15	Amount	Independent	no
30	Canada	b	<i>Agelaius phoeniceus</i>	0.44	15	Amount	Independent	no
30	Canada	b	<i>Rallus limicola</i>	0.2	15	Amount	Independent	no
30	Canada	b	<i>Geothlypis trichas</i>	-0.31	15	Amount	Independent	no
30	Canada	b	<i>Gallinula galeata</i>	0.42	15	Amount	Independent	no
30	Canada	b	<i>Porzana carolina</i>	0.13	15	Amount	Independent	no
31	USA	b	<i>Anas wyvilliana</i>	0.34	5	Amount	Dependent	yes
32	USA	b	<i>Ixobrychus exilis</i>	-0.08	62	Amount	Independent	yes
32	USA	b	<i>Porphyrio martinica</i>	0.01	109	Amount	Independent	yes
32	USA	b	<i>Gallinula galeata</i>	0.01	109	Amount	Independent	yes
33	USA	b	<i>Grus canadensis</i>	0.56	10	Amount	Dependent	yes
33	USA	b	<i>Chlidonias niger</i>	0.08	22	Amount	Dependent	yes
33	USA	b	<i>Geothlypis trichas</i>	0.65	32	Amount	Dependent	yes
33	USA	b	<i>Empidonax alnorum</i>	0.12	10	Amount	Dependent	yes
33	USA	b	<i>Melospiza georgiana</i>	0.49	32	Amount	Dependent	yes
33	USA	b	<i>Xanthocephalus xanthocephalus</i>	0.04	12	Amount	Dependent	yes
33	USA	b	<i>Gallinago delicata</i>	0.2	10	Amount	Dependent	yes
33	USA	b	<i>Setophaga petechia</i>	0.36	32	Amount	Dependent	yes
33	USA	b	<i>Cistothorus palustris</i>	0.42	22	Amount	Dependent	yes
33	USA	b	<i>Agelaius phoeniceus</i>	-0.35	32	Amount	Dependent	yes
33	USA	b	<i>Podilymbus podiceps</i>	0.09	22	Amount	Dependent	yes
33	USA	b	<i>Cistothorus platensis</i>	0.36	22	Amount	Dependent	yes
33	USA	b	<i>Rallus limicola</i>	-0.25	22	Amount	Dependent	yes

33	USA	b	<i>Fulica americana</i>	-0.05	22	Amount	Dependent	yes
33	USA	b	<i>Anas platyrhynchos</i>	0.08	32	Amount	Dependent	yes
33	USA	b	<i>Anas discors</i>	0.06	22	Amount	Dependent	yes
33	USA	b	<i>Porzana carolina</i>	0.2	32	Amount	Dependent	yes
34	USA	r	<i>Emydoidea blandingii</i>	0.09	7	Configuration	Dependent	no
34	USA	r	<i>Nerodia erythrogaster neglecta</i>	0.13	12	Configuration	Dependent	no
34	USA	r	<i>Nerodia sipedon sipedon</i>	0.21	21	Configuration	Dependent	no
34	USA	r	<i>Chrysemys picta marginata</i>	0.26	31	Configuration	Dependent	no
35	USA	r	<i>Chrysemys picta</i>	0.4	80	Configuration	Dependent	no
36	Canada	r	<i>Chrysemys picta marginata</i>	-0.19	20	Amount	Dependent	yes
37	USA	r	<i>Emydoidea blandingii</i>	0.5	6	Amount	Dependent	yes
37	USA	r	<i>Clemmys guttata</i>	0.43	10	Amount	Dependent	yes
38	USA	r	<i>Chrysemys picta</i>	0.22	35	Amount	Independent	no
39	USA	r	<i>Nerodia erythrogaster neglecta</i>	0.12	76	Amount	Independent	yes
39	USA	r	<i>Nerodia rhombifer rhombifer</i>	0.26	76	Amount	Independent	yes
39	USA	r	<i>Nerodia sipedon pleuralis</i>	0.04	76	Amount	Independent	yes
40	USA	r	<i>Glyptemys muhlenbergii</i>	0.09	43	Amount	Unknown	yes
24	Canada	r	<i>Sternotherus odoratus</i>	-0.21	66	Amount	Independent	yes
24	Canada	r	<i>Emydoidea blandingii</i>	-0.08	70	Amount	Independent	yes
41	USA	r	<i>Apalone spinifera spinifera</i>	-0.09	30	Amount	Independent	yes
41	USA	r	<i>Trachemys scripta elegans</i>	0.21	55	Amount	Independent	yes
41	USA	r	<i>Chelydra serpentina serpentina</i>	-0.13	150	Amount	Independent	yes
41	USA	r	<i>Chrysemys picta marginata</i>	0.16	206	Amount	Independent	yes
42	Australia	r	<i>Chelodina longicollis</i>	0.23	9	Configuration	Dependent	no
43	USA	r	<i>Chelydra serpentina</i>	0.04	25	Amount	Independent	no

43	USA	r	<i>Chrysemys picta</i>	0.18	25	Amount	Independent	no
44	USA	r	<i>Chelydra serpentina</i>	-0.06	42	Amount	Independent	no
44	USA	r	<i>Chrysemys picta</i>	-0.06	42	Amount	Independent	no
45	USA	a	<i>Ambystoma maculatum</i>	-0.23	56	Configuration	Dependent	no
45	USA	a	<i>Rana sylvatica</i>	-0.29	56	Configuration	Dependent	no
1	Canada	a	<i>Bufo boreas</i>	-0.07	9	Amount	Independent	no
1	Canada	a	<i>Pseudacris maculata</i>	-0.27	15	Amount	Independent	no
1	Canada	a	<i>Rana sylvatica</i>	-0.27	15	Amount	Independent	no
46	Switzerland	a	<i>Bufo bufo</i>	-0.3	34	Configuration	Dependent	yes
46	Switzerland	a	<i>Triturus cristatus</i>	0.11	38	Configuration	Dependent	yes
46	Switzerland	a	<i>Rana temporaria</i>	-0.15	44	Configuration	Dependent	yes
46	Switzerland	a	<i>Hyla arborea</i>	0.27	46	Configuration	Dependent	yes
46	Switzerland	a	<i>Ichthyosaura alpestris</i>	0.14	49	Configuration	Dependent	yes
46	Switzerland	a	<i>Rana esculenta</i>	0.28	52	Configuration	Dependent	yes
46	Switzerland	a	<i>Rana lessonae</i>	0.3	52	Configuration	Dependent	yes
46	Switzerland	a	<i>Lissotriton vulgaris</i>	0.27	66	Configuration	Dependent	yes
47	USA	a	<i>Hemidactylium scutatum</i>	-0.27	7	Configuration	Dependent	yes
48	USA	a	<i>Ambystoma tigrinum tigrinum</i>	0.19	89	Configuration	Dependent	no
49	USA	a	<i>Rana catesbeiana</i>	0.36	29	Amount	Dependent	yes
49	USA	a	<i>Rana palustris</i>	0.08	29	Amount	Dependent	yes
49	USA	a	<i>Pseudacris crucifer</i>	0.21	39	Amount	Dependent	yes
49	USA	a	<i>Rana clamitans</i>	0.15	39	Amount	Dependent	yes
49	USA	a	<i>Ambystoma maculatum</i>	0.11	45	Amount	Dependent	yes
49	USA	a	<i>Rana sylvatica</i>	0.02	45	Amount	Dependent	yes
49	USA	a	<i>Notophthalmus viridescens</i>	0.19	67	Amount	Dependent	yes

50	USA	a	<i>Rana draytonii</i>	0.37	22	Configuration	Dependent	no
50	USA	a	<i>Rana catesbeiana</i>	0.58	28	Configuration	Dependent	no
50	USA	a	<i>Pseudacris regilla</i>	-0.14	32	Configuration	Independent	no
51	Belgium	a	<i>Ichthyosaura alpestris</i>	-0.01	254	Configuration	Dependent	yes
51	Belgium	a	<i>Lissotriton vulgaris</i>	0.08	254	Configuration	Dependent	yes
52	France	a	<i>Lissotriton helveticus</i>	-0.08	101	Configuration	Dependent	yes
53	Canada	a	<i>Bufo americanus</i>	-0.12	36	Amount	Independent	no
53	Canada	a	<i>Hyla versicolor</i>	0.22	36	Amount	Independent	no
53	Canada	a	<i>Pseudacris crucifer</i>	0.33	36	Amount	Independent	no
53	Canada	a	<i>Rana clamitans</i>	0.13	36	Amount	Dependent	no
53	Canada	a	<i>Rana pipiens</i>	-0.16	36	Amount	Independent	no
53	Canada	a	<i>Rana sylvatica</i>	0.05	36	Amount	Independent	no
54	USA	a	<i>Rana luteiventris</i>	-0.16	95	Amount	Independent	no
54	USA	a	<i>Pseudacris regilla</i>	-0.19	99	Amount	Independent	no
54	USA	a	<i>Ambystoma macrodactylum</i>	-0.06	105	Amount	Independent	no
55	USA	a	<i>Ambystoma tigrinum mavortium</i>	-0.35	12	Amount	Dependent	yes
55	USA	a	<i>Bufo cognatus</i>	-0.07	12	Amount	Dependent	yes
55	USA	a	<i>Spea bombifrons</i>	0.59	12	Amount	Dependent	yes
55	USA	a	<i>Spea multiplicata</i>	0.51	12	Amount	Dependent	yes
56	USA	a	<i>Ambystoma laterale</i>	0.04	89	Amount	Dependent	yes
56	USA	a	<i>Ambystoma maculatum</i>	-0.08	89	Amount	Dependent	yes
56	USA	a	<i>Bufo americanus</i>	0.22	89	Amount	Independent	yes
56	USA	a	<i>Notophthalmus viridescens</i>	-0.11	89	Amount	Dependent	yes
56	USA	a	<i>Rana clamitans</i>	0.11	89	Amount	Independent	yes
56	USA	a	<i>Rana pipiens</i>	0.18	89	Amount	Independent	yes

56	USA	a	<i>Rana septentrionalis</i>	0.17	89	Amount	Independent	yes
56	USA	a	<i>Rana sylvatica</i>	-0.13	89	Amount	Independent	yes
57	Sweden	a	<i>Triturus cristatus</i>	0.11	134	Amount	Independent	no
58	Australia	a	<i>Limnodynastes dumerilii</i>	0.13	36	Amount	Independent	yes
58	Australia	a	<i>Limnodynastes peronii</i>	0.29	36	Amount	Independent	yes
58	Australia	a	<i>Litoria ewingii/verreauxii</i>	-0.02	36	Amount	Independent	yes
58	Australia	a	<i>Crinia signifera</i>	0.11	44	Amount	Independent	yes
58	Australia	a	<i>Paracrinia haswelli</i>	0.1	44	Amount	Independent	yes
59	Australia	a	<i>Crinia signifera</i>	0.15	30	Amount	Dependent	yes
59	Australia	a	<i>Limnodynastes dumerilii</i>	0.07	30	Amount	Dependent	yes
59	Australia	a	<i>Limnodynastes peronii</i>	-0.17	30	Amount	Dependent	yes
59	Australia	a	<i>Limnodynastes tasmaniensis</i>	0.18	30	Amount	Dependent	yes
59	Australia	a	<i>Litoria ewingii</i>	-0.31	30	Amount	Dependent	yes
59	Australia	a	<i>Litoria fallax</i>	-0.06	30	Amount	Dependent	yes
59	Australia	a	<i>Litoria raniformis</i>	-0.11	30	Amount	Dependent	yes
59	Australia	a	<i>Litoria verreauxii</i>	-0.09	30	Amount	Dependent	yes
60	Romania	a	<i>Bombina variegata</i>	0.11	54	Amount	Dependent	no
60	Romania	a	<i>Bufo bufo</i>	0.16	54	Amount	Dependent	no
60	Romania	a	<i>Hyla arborea</i>	-0.05	54	Amount	Dependent	no
60	Romania	a	<i>Lissotriton vulgaris</i>	-0.22	54	Amount	Dependent	no
60	Romania	a	<i>Pelobates fuscus</i>	-0.04	54	Amount	Dependent	no
60	Romania	a	<i>Rana dalmatina</i>	-0.12	54	Amount	Dependent	no
60	Romania	a	<i>Rana esculenta</i>	0.12	54	Amount	Dependent	no
60	Romania	a	<i>Rana temporaria</i>	-0.27	54	Amount	Dependent	no
60	Romania	a	<i>Triturus cristatus</i>	-0.24	54	Amount	Dependent	no

61	Australia	a	<i>Litoria raniformis</i>	0.54	107	Configuration	Dependent	no
62	Canada	a	<i>Bufo americanus</i>	0.17	34	Configuration	Dependent	no
62	Canada	a	<i>Hyla versicolor</i>	0.06	34	Configuration	Dependent	no
62	Canada	a	<i>Notophthalmus viridescens</i>	0.11	34	Configuration	Dependent	no
62	Canada	a	<i>Pseudacris crucifer</i>	0.11	34	Configuration	Dependent	no
62	Canada	a	<i>Pseudacris triseriata</i>	0.17	34	Configuration	Dependent	no
62	Canada	a	<i>Rana clamitans</i>	0.13	34	Configuration	Dependent	no
62	Canada	a	<i>Rana pipiens</i>	0.16	34	Configuration	Dependent	no
62	Canada	a	<i>Rana sylvatica</i>	0	34	Configuration	Dependent	no
63	USA	a	<i>Ambystoma maculatum</i>	0.07	62	Amount	Unknown	no
63	USA	a	<i>Rana sylvatica</i>	-0.11	62	Amount	Unknown	no
64	Canada	a	<i>Ambystoma laterale</i>	0.11	46	Amount	Dependent	no
64	Canada	a	<i>Ambystoma maculatum</i>	0.32	46	Amount	Dependent	no
64	Canada	a	<i>Bufo americanus</i>	0.09	46	Amount	Dependent	no
64	Canada	a	<i>Hyla versicolor</i>	0.35	46	Amount	Dependent	no
64	Canada	a	<i>Notophthalmus viridescens</i>	0.16	46	Amount	Dependent	no
64	Canada	a	<i>Pseudacris crucifer</i>	0.12	46	Amount	Dependent	no
64	Canada	a	<i>Pseudacris triseriata</i>	0.25	46	Amount	Dependent	no
64	Canada	a	<i>Rana catesbeiana</i>	0.16	46	Amount	Dependent	no
64	Canada	a	<i>Rana clamitans</i>	0.13	46	Amount	Dependent	no
64	Canada	a	<i>Rana pipiens</i>	0.19	46	Amount	Dependent	no
64	Canada	a	<i>Rana septentrionalis</i>	0.5	46	Amount	Dependent	no
64	Canada	a	<i>Rana sylvatica</i>	0.26	46	Amount	Dependent	no
65	Canada	a	<i>Ambystoma laterale/A. maculatum</i>	-0.28	34	Amount	Dependent	no
65	Canada	a	<i>Bufo americanus</i>	0.18	34	Amount	Independent	no

65	Canada	a	<i>Notophthalmus viridescens</i>	0.47	34	Amount	Dependent	no
65	Canada	a	<i>Pseudacris crucifer</i>	-0.23	34	Amount	Independent	no
65	Canada	a	<i>Rana catesbeiana</i>	-0.39	34	Amount	Dependent	no
65	Canada	a	<i>Rana clamitans</i>	0.23	34	Amount	Dependent	no
65	Canada	a	<i>Rana palustris</i>	0.33	34	Amount	Dependent	no
65	Canada	a	<i>Rana septentrionalis</i>	0.46	34	Amount	Dependent	no
65	Canada	a	<i>Rana sylvatica</i>	-0.25	34	Amount	Dependent	no
66	USA	a	<i>Rana pipiens</i>	-0.06	31	Amount	Independent	no
67	France	a	<i>Ichthyosaura alpestris</i>	0.13	79	Configuration	Independent	yes
67	France	a	<i>Lissotriton helveticus</i>	0.24	79	Configuration	Independent	yes
67	France	a	<i>Triturus cristatus</i>	0.3	79	Configuration	Independent	yes
68	USA	a	<i>Rana muscosa</i>	0.12	1231	Amount	Dependent	no
69	USA	a	<i>Rana palustris</i>	-0.15	22	Amount	Independent	yes
69	USA	a	<i>Rana pipiens</i>	0.19	22	Amount	Independent	yes
69	USA	a	<i>Bufo americanus</i>	-0.13	27	Amount	Independent	yes
69	USA	a	<i>Hyla versicolor</i>	-0.31	35	Amount	Independent	yes
69	USA	a	<i>Pseudacris crucifer</i>	-0.18	37	Amount	Independent	yes
69	USA	a	<i>Ambystoma tigrinum</i>	-0.24	39	Amount	Independent	yes
69	USA	a	<i>Pseudacris triseriata</i>	-0.16	39	Amount	Independent	yes
69	USA	a	<i>Rana clamitans</i>	0.08	39	Amount	Independent	yes
70	USA	a	<i>Hyla versicolor/H. chrysoscelis</i>	0.26	21	Amount	Independent	yes
70	USA	a	<i>Ambystoma tigrinum tigrinum</i>	-0.42	26	Amount	Independent	yes
70	USA	a	<i>Bufo americanus</i>	-0.25	26	Amount	Independent	yes
70	USA	a	<i>Bufo cognatus</i>	-0.38	26	Amount	Independent	yes
70	USA	a	<i>Pseudacris triseriata</i>	0.51	26	Amount	Independent	yes

70	USA	a	<i>Rana pipiens</i>	0.21	26	Amount	Independent	yes
70	USA	a	<i>Rana sylvatica</i>	0.29	26	Amount	Independent	yes
71	Panama	a	<i>Engystomops pustulosus</i>	0.51	7	Configuration	Unknown	no
72	Canada	a	<i>Rana clamitans</i>	0.13	21	Amount	Dependent	yes
73	USA	a	<i>Bufo fowleri</i>	-0.05	22	Amount	Independent	no
73	USA	a	<i>Hyla versicolor</i>	-0.08	22	Amount	Independent	no
73	USA	a	<i>Pseudacris crucifer</i>	0.33	22	Amount	Independent	no
73	USA	a	<i>Rana catesbeiana</i>	0.14	22	Amount	Independent	no
74	USA	a	<i>Rana virgatipes</i>	0.32	38	Amount	Independent	no
75	Switzerland	a	<i>Hyla arborea</i>	0.08	68	Amount	Independent	no
76	Switzerland	a	<i>Hyla arborea</i>	0.2	68	Configuration	Independent	no
77	USA	a	<i>Acris crepitans</i>	0.13	61	Amount	Independent	no
77	USA	a	<i>Bufo americanus</i>	-0.11	61	Amount	Independent	no
77	USA	a	<i>Hyla chrysoscelis</i>	0.05	61	Amount	Independent	no
77	USA	a	<i>Hyla versicolor</i>	0.12	61	Amount	Independent	no
77	USA	a	<i>Pseudacris triseriata</i>	0.06	61	Amount	Independent	no
77	USA	a	<i>Rana catesbeiana</i>	-0.07	61	Amount	Independent	no
77	USA	a	<i>Rana pipiens</i>	-0.05	61	Amount	Independent	no
78	Romania	a	<i>Bufo bufo</i>	-0.07	26	Amount	Dependent	yes
78	Romania	a	<i>Rana temporaria</i>	0.13	44	Amount	Dependent	yes
78	Romania	a	<i>Hyla arborea</i>	0.18	45	Amount	Dependent	yes
78	Romania	a	<i>Bombina variegata</i>	-0.02	51	Amount	Dependent	yes
78	Romania	a	<i>Rana dalmatina</i>	0.21	51	Amount	Dependent	yes
78	Romania	a	<i>Rana esculenta</i>	0.21	51	Amount	Dependent	yes
78	Romania	a	<i>Triturus cristatus</i>	0.35	52	Amount	Dependent	yes

78	Romania	a	<i>Lissotriton vulgaris</i>	0.54	53	Amount	Dependent	yes
78	Romania	a	<i>Salamandra salamandra</i>	0.56	53	Amount	Dependent	yes
79	Switzerland	a	<i>Bufo bufo</i>	-0.32	61	Configuration	Unknown	yes
79	Switzerland	a	<i>Bufo calamita</i>	0.24	61	Configuration	Unknown	yes
79	Switzerland	a	<i>Ichthyosaura alpestris</i>	0.14	91	Configuration	Unknown	yes
80	USA	a	<i>Bufo americanus/B. woodhouseii fowleri</i>	-0.06	27	Configuration	Dependent	no
80	USA	a	<i>Hyla versicolor</i>	-0.14	27	Configuration	Dependent	no
80	USA	a	<i>Rana catesbeiana</i>	0.05	27	Configuration	Dependent	no
80	USA	a	<i>Rana palustris</i>	-0.38	27	Configuration	Dependent	no
80	USA	a	<i>Ambystoma maculatum/A. jeffersonianum</i>	0.44	34	Configuration	Dependent	no
80	USA	a	<i>Notophthalmus viridescens</i>	-0.01	34	Configuration	Dependent	no
80	USA	a	<i>Pseudacris crucifer</i>	0	34	Configuration	Dependent	no
80	USA	a	<i>Rana clamitans</i>	-0.13	34	Configuration	Dependent	no
80	USA	a	<i>Rana sylvatica</i>	0.37	34	Configuration	Dependent	no
81	USA	a	<i>Rana pipiens</i>	0.22	73	Amount	Independent	no
82	USA	a	<i>Rana blairi/R. sphenoccephalus</i>	0.02	38	Configuration	Dependent	no
82	USA	a	<i>Bufo americanus</i>	-0.34	40	Configuration	Dependent	no
82	USA	a	<i>Rana catesbeiana</i>	0.27	44	Configuration	Dependent	no
82	USA	a	<i>Hyla versicolor/H. chrysoscelis</i>	0.23	45	Configuration	Dependent	no
82	USA	a	<i>Pseudacris crucifer</i>	0.02	45	Configuration	Dependent	no
82	USA	a	<i>Pseudacris maculata</i>	-0.09	45	Configuration	Dependent	no
82	USA	a	<i>Rana clamitans</i>	0.03	48	Configuration	Dependent	no
82	USA	a	<i>Acris crepitans</i>	0.12	49	Configuration	Dependent	no
82	USA	a	<i>Ambystoma texanum</i>	0.17	49	Configuration	Dependent	no
83	Brazil	a	<i>Dendropsophus elianae</i>	0.14	6	Configuration	Dependent	no

83	Brazil	a	<i>Dendropsophus minutus</i>	0.25	6	Configuration	Dependent	no
83	Brazil	a	<i>Dendropsophus nanus</i>	0.06	6	Configuration	Dependent	no
83	Brazil	a	<i>Dermatonotus muelleri</i>	0.24	6	Configuration	Dependent	no
83	Brazil	a	<i>Elachistocleis bicolor</i>	0.22	6	Configuration	Dependent	no
83	Brazil	a	<i>Eupemphix nattereri</i>	-0.15	6	Configuration	Dependent	no
83	Brazil	a	<i>Hypsiboas albopunctatus</i>	0	6	Configuration	Dependent	no
83	Brazil	a	<i>Hypsiboas raniceps</i>	-0.23	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus chaquensis</i>	0.1	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus fuscus</i>	0.27	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus labyrinthicus</i>	0.08	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus latrans</i>	-0.27	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus mystacinus</i>	0.67	6	Configuration	Dependent	no
83	Brazil	a	<i>Leptodactylus podicipinus</i>	0.18	6	Configuration	Dependent	no
83	Brazil	a	<i>Physalaemus centralis</i>	0.16	6	Configuration	Dependent	no
83	Brazil	a	<i>Physalaemus cuvieri</i>	0.21	6	Configuration	Dependent	no
83	Brazil	a	<i>Pseudis platensis</i>	0.14	6	Configuration	Dependent	no
83	Brazil	a	<i>Pseudopaludicola falcipes</i>	-0.29	6	Configuration	Dependent	no
83	Brazil	a	<i>Rhinella schneideri</i>	-0.11	6	Configuration	Dependent	no
83	Brazil	a	<i>Scinax fuscomarginatus</i>	-0.21	6	Configuration	Dependent	no
83	Brazil	a	<i>Scinax similis</i>	0.21	6	Configuration	Dependent	no
83	Brazil	a	<i>Scinax fuscovarius</i>	0.59	17	Configuration	Dependent	no
84	USA	a	<i>Rana palustris</i>	0.2	12	Amount	Dependent	no
84	USA	a	<i>Rana utricularia</i>	0.35	12	Amount	Dependent	no
84	USA	a	<i>Bufo americanus</i>	-0.26	17	Amount	Dependent	no
84	USA	a	<i>Hyla versicolor/chrysozelis</i>	0.37	17	Amount	Dependent	no

84	USA	a	<i>Notophthalmus viridescens</i>	0.14	17	Amount	Dependent	no
84	USA	a	<i>Pseudacris crucifer</i>	-0.28	17	Amount	Dependent	no
84	USA	a	<i>Rana catesbeiana</i>	-0.2	17	Amount	Dependent	no
84	USA	a	<i>Rana clamitans</i>	0.11	17	Amount	Dependent	no
84	USA	a	<i>Rana sylvatica</i>	0.57	18	Amount	Dependent	no
84	USA	a	<i>Acris crepitans</i>	0.47	19	Amount	Dependent	no
84	USA	a	<i>Bufo fowleri</i>	-0.25	19	Amount	Dependent	no
85	USA	a	<i>Ambystoma maculatum</i>	0.05	55	Amount	Dependent	yes
85	USA	a	<i>Rana sylvatica</i>	0.05	55	Amount	Dependent	yes
86	Australia	a	<i>Crinia parinsignifera</i>	-0.05	30	Amount	Independent	yes
86	Australia	a	<i>Crinia signifera</i>	-0.12	30	Amount	Independent	yes
86	Australia	a	<i>Limnodynastes dumerilii</i>	-0.22	30	Amount	Independent	yes
86	Australia	a	<i>Limnodynastes peronii</i>	-0.13	30	Amount	Independent	yes
86	Australia	a	<i>Limnodynastes tasmaniensis</i>	-0.28	30	Amount	Independent	yes
86	Australia	a	<i>Litoria ewingii/L. paraewingi</i>	0.29	30	Amount	Independent	yes
86	Australia	a	<i>Litoria peronii</i>	-0.1	30	Amount	Independent	yes
87	USA	a	<i>Ambystoma maculatum</i>	0.18	49	Amount	Dependent	no
87	USA	a	<i>Rana sylvatica</i>	-0.1	49	Amount	Dependent	no
88	Netherlands	a	<i>Rana arvalis</i>	0.09	74	Amount	Unknown	yes
89	Netherlands	a	<i>Hyla arborea</i>	0.46	149	Configuration	Unknown	yes
90	USA	a	<i>Pseudacris crucifer</i>	-0.11	13	Configuration	Dependent	no
90	USA	a	<i>Pseudacris triseriata</i>	-0.15	26	Configuration	Dependent	no
91	Switzerland	a	<i>Bufo bufo</i>	0.07	107	Amount	Unknown	yes
91	Switzerland	a	<i>Bufo calamita</i>	-0.1	247	Amount	Unknown	yes
91	Switzerland	a	<i>Rana temporaria</i>	0.1	296	Amount	Unknown	yes

91	Switzerland	a	<i>Hyla arborea</i>	0.22	315	Amount	Unknown	yes
91	Switzerland	a	<i>Ichthyosaura alpestris</i>	-0.07	414	Amount	Unknown	yes
91	Switzerland	a	<i>Rana ridibunda</i>	0.06	423	Amount	Unknown	yes
91	Switzerland	a	<i>Lissotriton helveticus</i>	-0.03	469	Amount	Unknown	yes
91	Switzerland	a	<i>Bombina variegata</i>	-0.07	480	Amount	Unknown	yes
91	Switzerland	a	<i>Rana dalmatina</i>	0.14	520	Amount	Unknown	yes
91	Switzerland	a	<i>Lissotriton vulgaris</i>	0.12	615	Amount	Unknown	yes

*Studies used in meta-analysis (Chapter 2) are cited below. Studies conducted in the same study area by the same researchers are identified as one study in the meta-analysis

[†]Taxa: m = mammal, b = bird, r = reptile, a = amphibian

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Appendix B: Species traits and reference information for the 220 species included in the meta-analysis (Chapter 2).

Study*	Taxa ⁺	Species	Order	Repro ^a	Source [§]	Home Range ^b (ha)	Source	Mass ^c (g)	Source	Length ^d (cm)	Source
1	m	<i>Castor canadensis</i>	Rodentia	3.0	Baker and Hill 2003	13	Baker and Hill 2003	23750	Baker and Hill 2003	120	Baker and Hill 2003
2	m	<i>Oryzomys palustris</i>	Rodentia	15.2	Jones et al. 2009; Linzey and NatureServe 2008	0.73	Eubanks et al. 2011	53.26	Jones et al. 2009	13.35	Jones et al. 2009
3	m	<i>Microtus pennsylvanicus</i>	Rodentia	33.8	Linzey and NatureServe 2008	0.17	Reich 1981	44.1	Reich 1981	16.7	Reich 1981
3	m	<i>Sorex fumeus</i>	Rodentia	8.5	Jones et al. 2009	0.12	Linzey 1983	35.7	Linzey 1983	12.4	Linzey 1983
3	m	<i>Synaptomys cooperi</i>	Soricomorpha	11.0	Owen 1984; Weinstein 1999	n/a		7.6	Owen 1984	11.1	Owen 1984
4	m	<i>Sylvilagus palustris hefneri</i>	Lagomorpha	6.5	Forys 1995	3.96	Forys and Humphrey 1996	1224.1	Forys 1995	33.9	Forys 1995
5	m	<i>Neofiber alleni</i>	Rodentia	11.5	Baker and Hill 2003	0.21	Schooley and Branch 2006	270.5	Baker and Hill 2003	33.3	Baker and Hill 2003
6	b	<i>Ardea cinerea</i>	Pelecaniformes	4.5	Martínez-Vilalta and Motis 1992	767.5	van Vessem et al. 1984	1546.5	Martínez-Vilalta and Motis 1992	94	Martínez-Vilalta and Motis 1992
6	b	<i>Egretta intermedia</i>	Pelecaniformes	4.0	Martínez-Vilalta and Motis 1992	28.26	Amano and Katayama 2009	400	Martínez-Vilalta and Motis 1992	64	Martínez-Vilalta and Motis 1992
6	b	<i>Nycticorax nycticorax</i>	Pelecaniformes	4.0	Martínez-Vilalta and Motis 1992	21.022	Wong et al. 1999	662.5	Martínez-Vilalta and Motis 1992	60.5	Martínez-Vilalta and Motis 1992
7	b	<i>Botaurus lentiginosus</i>	Ciconiiformes	4.1	Lowther et al. 2009	109.28	Lor 2007	435	Lowther et al. 2009	72.5	Lowther et al. 2009
7	b	<i>Ixobrychus exilis</i>	Ciconiiformes	4.5	Poole et al.	223.2	Griffin et al.	80	Poole et al.	32	Poole et al.

8	b	<i>Botaurus lentiginosus</i>	Ciconiiformes	4.1	2009 Lowther et al. 2009	109.28	2009 Lor 2007	435	2009 Lowther et al. 2009	72.5	2009 Lowther et al. 2009
8	b	<i>Fulica americana</i>	Gruiformes	8.5	Brisbin et al. 2002	0.15	Brisbin et al. 2002	637.5	Brisbin et al. 2002	37.5	Brisbin et al. 2002
8	b	<i>Ixobrychus exilis</i>	Ciconiiformes	3.8	Poole et al. 2009	223.2	Griffin et al. 2009	80	Poole et al. 2009	32	Poole et al. 2009
8	b	<i>Podilymbus podiceps</i>	Podicipediformes	7.0	Muller and Storer 1999	1.31	Muller and Storer 1999	410.5	Muller and Storer 1999	34.3	Muller and Storer 1999
8	b	<i>Porzana carolina</i>		10.3	Melvin and Gibbs 2012	0.19	Johnson and Dinsmore 1985	80	Melvin and Gibbs 2012	22.5	Melvin and Gibbs 2012
9	b	<i>Circus aeruginosus</i>	Falconiformes	4.5	Dijkstra and Zijlstra 1997; Orta 1994	2517.3	Cardador et al. 2009	603	Thiollay 1994	56.5	Thiollay 1994
10	b	<i>Acrocephalus scirpaceus</i>	Passeriformes	7.8	Catchpole 1974	0.033	Catchpole 1972; Chernetsov and Titov 2001	12.3	Dyrzcz 2006	13	Dyrzcz 2006
11	b	<i>Ardea herodias</i>	Ciconiiformes	4.1	Vennesland and Butler 2011	572.27	Custer and Galli 2002	2300	Vennesland and Butler 2011	117	Vennesland and Butler 2011
12	b	<i>Ardea herodias</i>	Ciconiiformes	4.1	Vennesland and Butler 2011	572.27	Custer and Galli 2002	2300	Vennesland and Butler 2011	117	Vennesland and Butler 2011
13	b	<i>Botaurus lentiginosus</i>	Ciconiiformes	4.6	Lor and Malecki. 2006	109.28	Lor 2007	435	Lowther et al. 2009	72.5	Lowther et al. 2009
13	b	<i>Butorides virescens</i>	Ciconiiformes	3.9	Davis and Kushlan 1994	n/a		241	Davis and Kushlan 1994	43.5	Davis and Kushlan 1994
13	b	<i>Circus cyaneus</i>	Falconiformes	4.4	Smith et al. 2011	260	Smith et al. 2011	458	Smith et al. 2011	47.5	Thiollay 1994
13	b	<i>Gallinago delicata</i>	Charadriiformes	3.9	Mueller 1999	235	Cline and Haig 2011	100	Mueller 1999	28	Mueller 1999
13	b	<i>Podilymbus podiceps</i>	Podicipediformes	6.8	Muller and Storer 1999	1.31	Muller and Storer 1999	410.5	Muller and Storer 1999	34.3	Muller and Storer 1999
13	b	<i>Porzana carolina</i>	Gruiformes	11.5	Lor and Malecki 2006	0.19	Johnson and Dinsmore 1985	80	Melvin and Gibbs 2012	22.5	Melvin and Gibbs 2012

13	b	<i>Rallus limicola</i>	Gruiformes	7.9	Lor and Malecki 2006	0.18	Johnson and Dinsmore 1985	87	Conway 1995	24.5	Conway 1995
14	b	<i>Botaurus stellaris</i>	Ciconiiformes	4.0	Gilbert et al. 2007	21.437	Gilbert et al. 2005	1403.5	Martínez-Vilalta and Motis 1992	72	Martínez-Vilalta and Motis 1992
15	b	<i>Agelaius phoeniceus</i>	Passeriformes	5.6	Yasukawa and Searcy 1995	0.16	Yasukawa and Searcy 1995	57.2	Yasukawa and Searcy 1995	22	Rosenthal 2004
15	b	<i>Cistothorus palustris</i>	Passeriformes	5.0	Kroodsma and Verner 1997	0.1	Leonard and Picman 1986; Verner 1965	11.5	Kroodsma and Verner 1997	122	Kroodsma and Verner 1997
15	b	<i>Cistothorus platensis</i>	Passeriformes	7.0	Herkert et al. 2001	0.178	Herkert et al. 2001	8.5	Herkert et al. 2001	11	Herkert et al. 2001
16	b	<i>Amazonetta brasiliensis</i>	Anseriformes	7.0	Carboneras 1992b	n/a		415	Carboneras 1992b	37.5	Carboneras 1992b
16	b	<i>Anas flavirostris</i>	Anseriformes	6.5	Carboneras 1992b	n/a		715	Carboneras 1992b	40	Carboneras 1992b
16	b	<i>Anas versicolor</i>	Anseriformes	8.0	Carboneras 1992b	n/a		407.5	Carboneras 1992b	40.5	Carboneras 1992b
16	b	<i>Aramides ypecaha</i>	Gruiformes	13.5	Taylor 1996	n/a		765	Taylor 1996	43	Taylor 1996
16	b	<i>Aramus guarana</i>	Gruiformes	11.0	Bryan 1996, 2002	2.105	Bryan 2002	1180	Bryan 1996	63.5	Bryan 1996
16	b	<i>Ardea alba</i>	Ciconiiformes	3.6	Pretelli et al. 2012	1865.6	Custer and Osborn 1978; Stolen et al. 2007	1100	Martínez-Vilalta and Motis 1992	92	Martínez-Vilalta and Motis 1992
16	b	<i>Ardea cocoi</i>	Ciconiiformes	3.0	Martínez-Vilalta and Motis 1992	n/a		n/a		111	Martínez-Vilalta and Motis 1992
16	b	<i>Butorides striata</i>	Ciconiiformes	3.5	Martínez-Vilalta and Motis 1992	n/a		192.5	Martínez-Vilalta and Motis 1992	40.5	Martínez-Vilalta and Motis 1992
16	b	<i>Callonetta leucophrys</i>	Anseriformes	9.0	Carboneras 1992b	n/a		275	Carboneras 1992b	36.5	Carboneras 1992b
16	b	<i>Chauna torquata</i>	Anseriformes	4.0	Carboneras 1992a	n/a		4400	Carboneras 1992a	89	Carboneras 1992a
16	b	<i>Ciconia maguari</i>	Ciconiiformes	3.1	González 1998	n/a		n/a		99.5	Elliot 1992
16	b	<i>Circus buffoni</i>	Falconiformes	3.5	Thiollay 1994	n/a		501	Thiollay 1994	52	Thiollay

16	b	<i>Dendrocygna bicolor</i>	Anseriformes	10.0	Carboneras 1992b	n/a	688	Carboneras 1992b	49	1994 Carboneras 1992b	
16	b	<i>Dendrocygna viduata</i>	Anseriformes	8.5	Carboneras 1992b	n/a	661	Carboneras 1992b	43	Carboneras 1992b	
16	b	<i>Egretta thula</i>	Ciconiiformes	2.9	Petry and Da Silva Fonseca 2005	1734.1	Stolen et al. 2007	370	Martínez-Vilalta and Motis 1992	57.8	Martínez-Vilalta and Motis 1992
16	b	<i>Fulica leucoptera</i>	Gruiformes	11.0	Taylor 1996	n/a	528.5	Taylor 1996	39	Taylor 1996	
16	b	<i>Gallinago paraguaiiae</i>	Charadriiformes	4.0	van Gils and Wiersma 1996	n/a	136	van Gils and Wiersma 1996	25.5	and Wiersma 1996	
16	b	<i>Gallinula galeata</i>	Gruiformes	11.4	McRae 2011, Taylor 1996	n/a	305	Taylor 1996	34	Taylor 1996	
16	b	<i>Gallinula melanops</i>	Gruiformes	6.0	Taylor 1996	n/a	154	Taylor 1996	26	Taylor 1996	
16	b	<i>Himantopus himantopus</i>	Charadriiformes	4.0	Pierce 1996	0.221	Robinson et al. 1999	185.5	Pierce 1996	37.5	Pierce 1996
16	b	<i>Jacana jacana</i>	Charadriiformes	6.4	Emlen et al. 1998	0.113	Emlen et al. 1998	124.5	Jenni 1996	23	Jenni 1996
16	b	<i>Mycteria americana</i>	Ciconiiformes	3.0	González 1999	7084.6	Bryan et al. 2012	2500	Elliot 1992	92.5	Elliot 1992
16	b	<i>Netta peposaca</i>	Anseriformes	10.0	Carboneras 1992b	n/a	1100	Carboneras 1992b	55.5	Carboneras 1992b	
16	b	<i>Nycticorax nycticorax</i>	Ciconiiformes	2.9	Petry and Da Silva Fonseca 2005	21.022	Wong et al. 1999	662.5	Martínez-Vilalta and Motis 1992	60.5	Martínez-Vilalta and Motis 1992
16	b	<i>Pardirallus sanguinolentus</i>	Gruiformes	5.0	Taylor 1996	n/a	212.3	Taylor 1996	34	Taylor 1996	
16	b	<i>Phimosus infuscatus</i>	Ciconiiformes	3.1	Petry and Da Silva Fonseca 2005	n/a	n/a		50	Matheu and del Hoyo 1992	
16	b	<i>Platalea ajaja</i>	Ciconiiformes	2.5	Matheu and del Hoyo 1992	13070	Dumas 2000	1400	Matheu and del Hoyo 1992	77.5	Matheu and del Hoyo 1992
16	b	<i>Plegadis chihi</i>	Ciconiiformes	2.3	Petry and Da Silva Fonseca	1589.6	Bray and Klebenow	612.5	Ryder and Manry 1994	56	Matheu and del Hoyo

16	b	<i>Podilymbus podiceps</i>	Podicipediformes	16.5	2005 Llimona and del Hoyo 1992	1.31	1988 Muller and Storer 1999	398.5	Llimona and del Hoyo 1992	34	1992 Llimona and del Hoyo 1992
16	b	<i>Rollandia rolland</i>	Podicipediformes	6.0	Burger 1974; Llimona and del Hoyo 1992	0.002	Burger 1974	n/a		30	Llimona and del Hoyo 1992
16	b	<i>Rostrhamus sociabilis</i>	Falconiformes	9.0	Angehr 1999; Thiollay 1994	1163.6	Beissinger and Snyder 1987	376.5	Thiollay 1994	42.5	Thiollay 1994
16	b	<i>Syrigma sibilatrix</i>	Ciconiiformes	3.0	Martínez-Vilalta and Motis 1992	n/a		n/a		55.5	Martínez-Vilalta and Motis 1992
16	b	<i>Theristicus caerulescens</i>	Ciconiiformes	2.5	Matheu and del Hoyo 1992	n/a		n/a		73.5	Matheu and del Hoyo 1992
17	b	<i>Botaurus lentiginosus</i>	Ciconiiformes	4	Lowther et al. 2009	109.28	Lor 2007	435	Lowther et al. 2009	72.5	Lowther et al. 2009
17	b	<i>Ixobrychus exilis</i>	Ciconiiformes	3.8	Poole et al. 2009	9.7	Bogner and Baldassarre 2002	80	Poole et al. 2009	32	Poole et al. 2009
17	b	<i>Podilymbus podiceps</i>	Podicipediformes	7.3	Muller and Storer 1999	1.31	Muller and Storer 1999	410.5	Muller and Storer 1999	34.3	Muller and Storer 1999
17	b	<i>Porzana carolina</i>	Gruiformes	12.2	Melvin and Gibbs 2012	0.19	Johnson and Dinsmore 1985	80	Melvin and Gibbs 2012	22.5	Melvin and Gibbs 2012
17	b	<i>Rallus limicola</i>	Gruiformes	8.6	Kaufmann 1989	0.18	Johnson and Dinsmore 1985	87	Conway 1995	24.5	Conway 1995
18	b	<i>Ardea alba</i>	Ciconiiformes	2.9	Pratt and Winkler 1985	1865.6	Osborn 1978, Stolen et al. 2007	1000	Mccrimmon et al. 2011	99	Mccrimmon et al. 2011
18	b	<i>Ardea herodias</i>	Ciconiiformes	3.2	Pratt and Winkler 1985	452.16	Butler 1991	2300	Vennesland and Butler 2011	117	Vennesland and Butler 2011
19	b	<i>Acrocephalus rufescens</i>	Passeriformes	2.5	Dyrcz 2006, Urban et al. 1997	0.063	Urban et al. 1997	22.8	Dyrcz 2006	17	Dyrcz 2006
19	b	<i>Bradypterus carpalis</i>	Passeriformes	2.5	Pearson 2006,	n/a		14	Pearson 2006	14	Pearson

					Urban et al. 1997					2006	
19	b	<i>Chloropeta gracilirostris</i>	Passeriformes	2.4	Jetz et al. 2008	n/a		14.4	Pearson 2006	13.5	Pearson 2006
19	b	<i>Cisticola carruthersi</i>	Passeriformes	3	Madge 2006, Urban et al. 1997	n/a		11.3	Madge 2006	12.5	Madge 2006
19	b	<i>Laniarius mufumbiri</i>	Passeriformes	2	Jetz et al. 2008	n/a		40	Fry 2009	18	Fry 2009
19	b	<i>Serinus koliensis</i>	Passeriformes	1.5	Collar et al. 2010, Fry and Keith 2004	n/a		13.5	Collar et al. 2010	10.8	Collar et al. 2010
20	b	<i>Circus cyaneus</i>	Falconiformes	4.4	Smith et al. 2011	260	Smith et al. 2011	458	Smith et al. 2011	47.5	Thiollay 1994
21	b	<i>Acrocephalus arundinaceus</i>	Passeriformes	5	Dyrcz 2006	0.405	Catchpole et al. 1985	27.2	Dyrcz 2006	19.5	Dyrcz 2006
21	b	<i>Acrocephalus scirpaceus</i>	Passeriformes	7.8	Catchpole 1974	0.033	Catchpole 1972, Chernetsov and Titov 2001	12.3	Dyrcz 2006	13	Dyrcz 2006
21	b	<i>Cettia cetti</i>	Passeriformes	6	Clement 2006	1.275	Clement 2006	13.3	Clement 2006	14	Clement 2006
21	b	<i>Circus aeruginosus</i>	Falconiformes	4.5	Dijkstra and Zijlstra 1997; Orta 1994	2517.3	Cardador et al. 2009	603	Thiollay 1994	56.5	Thiollay 1994
21	b	<i>Rallus aquaticus</i>	Gruiformes	17	Taylor 1992	0.159	Jenkins et al. 1995	115	Taylor 1992	26.5	Taylor 1992
21	b	<i>Remiz pendulinus</i>	Passeriformes	5	Madge 2008; Pogány et al. 2012	87.87	Mészáros et al. 2006	10.3	Madge 2008	10.5	Madge 2008
22	b	<i>Limosa fedoa</i>	Charadriiformes	4	Gratto-Trevor 2000	90	Gratto-Trevor 2000	369.5	Gratto-Trevor 2000	45	Gratto-Trevor 2000
22	b	<i>Phalaropus tricolor</i>	Charadriiformes	4	Colwell and Jehl 1994	78.5	Colwell and Jehl 1994	62.1	Colwell and Jehl 1994	23	Colwell and Jehl 1994
22	b	<i>Recurvirostra americana</i>	Charadriiformes	4	Robinson et al. 1997	1316	Demers et al. 2008	312.5	Robinson et al. 1997	45	Robinson et al. 1997
22	b	<i>Tringa semipalmata</i>	Charadriiformes	4	Lowther et al.	44.3	Ryan and	265	Lowther et al.	37	Lowther et

23	b	<i>Dendrocygna bicolor</i>	Anseriformes	12.4	2001 Pierluissi 2006	n/a	Renken 1987	765.5	2001 Hohman and Lee 2001	46.5	al. 2001 Hohman and Lee 2001
23	b	<i>Gallinula galeata</i>	Gruiformes	8.4	Pierluissi 2006	1.22	Matthews 1983	383	Bannor and Kiviat 2002	33.5	Bannor and Kiviat 2002
23	b	<i>Ixobrychus exilis</i>	Ciconiiformes	4.2	Pierluissi 2006	223.2	Griffin et al. 2009	80	Poole et al. 2009	32	Poole et al. 2009
23	b	<i>Porphyrio martinica</i>	Gruiformes	7.8	Pierluissi 2006	1.03	West and Hess 2002	248	West and Hess 2002	31.5	West and Hess 2002
23	b	<i>Rallus elegans</i>	Gruiformes	9.1	Pierluissi 2006	14.53	Pickens 2012	337.3	Poole et al. 2005	43	Poole et al. 2005
24	b	<i>Cistothorus palustris</i>	Passeriformes	5	Kroodsma and Verner 1997	0.1	Leonard and Picman 1986, Verner 1965	11.5	Kroodsma and Verner 1997	122	Kroodsma and Verner 1997
24	b	<i>Geothlypis trichas</i>	Passeriformes	8	Abroe et al. 2007	1.2	Guzy and Ritchison 1999	9.5	Guzy and Ritchison 1999	12	Guzy and Ritchison 1999
24	b	<i>Ixobrychus exilis</i>	Ciconiiformes	3.8	Poole et al. 2009	9.7	Bogner and Baldassarre 2002	80	Poole et al. 2009	32	Poole et al. 2009
24	b	<i>Melospiza georgiana</i>	Passeriformes	3.8	Mowbray 1997	0.17	Mowbray 1997	17.8	Mowbray 1997	13.5	Mowbray 1997
24	b	<i>Podilymbus podiceps</i>	Podicipediformes	6.4	Lor and Malecki 2006	1.31	Muller and Storer 1999	410.5	Muller and Storer 1999	34.3	Muller and Storer 1999
24	b	<i>Rallus limicola</i>	Gruiformes	7.9	Lor and Malecki 2006	0.18	Johnson and Dinsmore 1985	87	Conway 1995	24.5	Conway 1995
24	b	<i>Setophaga petechia</i>	Passeriformes	3.8	Lowther et al. 1999	0.284	Lowther et al. 1999	10	Lowther et al. 1999	12.5	Lowther et al. 1999
25	b	<i>Botaurus lentiginosus</i>	Ciconiiformes	4.6	Lor and Malecki 2006	109.28	Lor 2007	435	Lowther et al. 2009	72.5	Lowther et al. 2009
25	b	<i>Ixobrychus exilis</i>	Ciconiiformes	5.3	Bogner and Baldassarre 2002	9.7	Bogner and Baldassarre 2002	80	Poole et al. 2009	32	Poole et al. 2009
25	b	<i>Podilymbus podiceps</i>	Podicipediformes	6.4	Lor and Malecki 2006	1.31	Muller and Storer 1999	410.5	Muller and Storer 1999	34.3	Muller and Storer 1999
25	b	<i>Porzana carolina</i>	Gruiformes	11.5	Lor and	0.19	Johnson and	80	Melvin and	22.5	Melvin and

25	b	<i>Rallus limicola</i>	Gruiformes	7.9	Malecki 2006 Lor and Malecki 2006	0.18	Dinsmore 1985 Johnson and Dinsmore 1985	87	Gibbs 2012 Conway 1995	24.5	Gibbs 2012 Conway 1995
26	b	<i>Himantopus himantopus</i>	Charadriiformes	4	Cuervo 2005; José Javier Cuervo, personal communication	n/a		185.5	Pierce 1996	37.5	Pierce 1996
26	b	<i>Tachybaptus ruficollis</i>	Podicipediformes	8	Llimona and del Hoyo 1992	1.693	Gutiérrez and Figuerola 1997	183	Llimona and del Hoyo 1992	27	Llimona and del Hoyo 1992
27	b	<i>Actitis hypoleucos</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		58.5	van Gils and Wiersma 1996	20	van Gils and Wiersma 1996
27	b	<i>Arenaria interpres</i>	Charadriiformes	4	van Gils and Wiersma 1996	754.39	Rehfishch et al. 2003	137	van Gils and Wiersma 1996	23.5	van Gils and Wiersma 1996
27	b	<i>Calidris alba</i>	Charadriiformes	8	van Gils and Wiersma 1996	1962.5	Macwhirter et al. 2002	71.5	van Gils and Wiersma 1996	20.5	van Gils and Wiersma 1996
27	b	<i>Calidris canutus</i>	Charadriiformes	3.5	van Gils and Wiersma 1996	880	Leyrer et al. 2006	152.5	van Gils and Wiersma 1996	24	van Gils and Wiersma 1996
27	b	<i>Calidris ferruginea</i>	Charadriiformes	3.8	van Gils and Wiersma 1996	n/a		80.5	van Gils and Wiersma 1996	20.5	van Gils and Wiersma 1996
27	b	<i>Calidris ruficollis</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		34.5	van Gils and Wiersma 1996	14.5	van Gils and Wiersma 1996
27	b	<i>Calidris subminuta</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		28.5	van Gils and Wiersma 1996	14.5	van Gils and

27	b	<i>Calidris temminckii</i>	Charadriiformes	10	van Gils and Wiersma 1996	n/a		25.5	van Gils and Wiersma 1996	14	Wiersma 1996 van Gils and Wiersma 1996
27	b	<i>Calidris tenuirostris</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		181.5	van Gils and Wiersma 1996	27	van Gils and Wiersma 1996
27	b	<i>Charadrius alexandrinus</i>	Charadriiformes	6	Wiersma 1992	36	Brindock and Colwell 2011	44	Wiersma 1992	16.3	Wiersma 1992
27	b	<i>Charadrius dubius</i>	Charadriiformes	4	Wiersma 1992	n/a		39.5	Wiersma 1992	15.5	Wiersma 1992
27	b	<i>Charadrius mongolus/C. leschenaultii</i>	Charadriiformes	3	Wiersma 1992	n/a		81.3	Wiersma 1992	21.5	Wiersma 1992
27	b	<i>Eurynorhynchus pygmeus</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		31.8	van Gils and Wiersma 1996	15	van Gils and Wiersma 1996
27	b	<i>Gallinago gallinago</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		126.5	van Gils and Wiersma 1996	26	van Gils and Wiersma 1996
27	b	<i>Limicola falcinellus</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		48	van Gils and Wiersma 1996	17	van Gils and Wiersma 1996
27	b	<i>Limnodromus semipalmatus</i>	Charadriiformes	2	van Gils and Wiersma 1996	n/a		86	van Gils and Wiersma 1996	34.5	van Gils and Wiersma 1996
27	b	<i>Limosa lapponica</i>	Charadriiformes	4	van Gils and Wiersma 1996	1519.8	Rehfish et al. 2003	370.5	van Gils and Wiersma 1996	39	van Gils and Wiersma 1996

27	b	<i>Limosa limosa</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		336	van Gils and Wiersma 1996	40	van Gils and Wiersma 1996
27	b	<i>Numenius arquata</i>	Charadriiformes	4	van Gils and Wiersma 1996	78.5	Rehfishch et al. 2003	813.8	van Gils and Wiersma 1996	55	van Gils and Wiersma 1996
27	b	<i>Numenius madagascariensis</i>	Charadriiformes	4	van Gils and Wiersma 1996	393.88	Finn et al. 2007	925	van Gils and Wiersma 1996	59.5	van Gils and Wiersma 1996
27	b	<i>Numenius phaeopus</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		433.3	van Gils and Wiersma 1996	43	van Gils and Wiersma 1996
27	b	<i>Philomachus pugnax</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		156	van Gils and Wiersma 1996	25.8	van Gils and Wiersma 1996
27	b	<i>Pluvialis fulva</i>	Charadriiformes	4	Wiersma 1992	1962.5	Johnson and Connors 2010	146	Wiersma 1992	24.5	Wiersma 1992
27	b	<i>Pluvialis squatarola</i>	Charadriiformes	4	Wiersma 1992	314	Rehfishch et al. 1996	247	Wiersma 1992	29	Wiersma 1992
27	b	<i>Recurvirostra avosetta</i>	Charadriiformes	3.5	Pierce 1992	n/a		311	Pierce 1996	43.5	Pierce 1996
27	b	<i>Tringa erythropus</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		163.5	van Gils and Wiersma 1996	30.5	van Gils and Wiersma 1996
27	b	<i>Tringa glareola</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		66	van Gils and Wiersma 1996	21	van Gils and Wiersma 1996
27	b	<i>Tringa nebularia</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		207.5	van Gils and Wiersma 1996	32.5	van Gils and Wiersma 1996

27	b	<i>Tringa stagnatilis</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		81.5	van Gils and Wiersma 1996	24	van Gils and Wiersma 1996
27	b	<i>Tringa totanus</i>	Charadriiformes	4	van Gils and Wiersma 1996	1808.6	Rehfish et al. 2003	120	van Gils and Wiersma 1996	28	van Gils and Wiersma 1996
27	b	<i>Xenus cinereus</i>	Charadriiformes	4	van Gils and Wiersma 1996	n/a		88	van Gils and Wiersma 1996	23.5	van Gils and Wiersma 1996
28	b	<i>Calidris alpina</i>	Charadriiformes	4	Warnock and Gill 1996	35600	Taft et al. 2008	56	Warnock and Gill 1996	19	Warnock and Gill 1996
29	b	<i>Ardeola ralloides</i>	Ciconiiformes	4.5	Hafner et al. 2001	1962.5	Hafner et al. 2001	300	Martínez-Vilalta and Motis 1992	44.5	Martínez-Vilalta and Motis 1992
29	b	<i>Bubulcus ibis</i>	Ciconiiformes	4.3	Prosper and Hafner 1996	122.66	Telfair 2006	365	Martínez-Vilalta and Motis 1992	51	Martínez-Vilalta and Motis 1992
29	b	<i>Egretta garzetta</i>	Ciconiiformes	4.1	Bennetts et al. 2000	314	Hafner and Britton 1983	459	Martínez-Vilalta and Motis 1992	60	Martínez-Vilalta and Motis 1992
29	b	<i>Nycticorax nycticorax</i>	Ciconiiformes	4	Martínez-Vilalta and Motis 1992	21.022	Wong et al. 1999	662.5	Martínez-Vilalta and Motis 1992	60.5	Martínez-Vilalta and Motis 1992
30	b	<i>Agelaius phoeniceus</i>	Passeriformes	5.6	Yasukawa and Searcy 1995	0.15	Yasukawa and Searcy 1995	57.2	Yasukawa and Searcy 1995	22	Rosenthal 2004
30	b	<i>Cistothorus palustris</i>	Passeriformes	5	Kroodsma and Verner 1997	0.1	Leonard and Picman 1986, Verner 1965	11.5	Kroodsma and Verner 1997	122	Kroodsma and Verner 1997
30	b	<i>Gallinula galeata</i>	Gruiformes	8	Brackney et al. 1982	1.22	Matthews 1983	383	Bannor and Kiviat 2002	33.5	Bannor and Kiviat 2002
30	b	<i>Geothlypis trichas</i>	Passeriformes	8	Abroe et al. 2007	1.2	Guzy and Ritchison 1999	9.5	Guzy and Ritchison 1999	12	Guzy and Ritchison 1999

30	b	<i>Ixobrychus exilis</i>	Ciconiiformes	3.8	Poole et al. 2009	9.7	Bogner and Baldassarre 2002	80	Poole et al. 2009	32	Poole et al. 2009
30	b	<i>Melospiza georgiana</i>	Passeriformes	3.8	Mowbray 1997	0.17	Mowbray 1997	17.8	Mowbray 1997	13.5	Mowbray 1997
30	b	<i>Porzana carolina</i>	Gruiformes	11.5	Lor and Malecki 2006	0.19	Johnson and Dinsmore 1985	80	Melvin and Gibbs 2012	22.5	Melvin and Gibbs 2012
30	b	<i>Rallus limicola</i>	Gruiformes	7.9	Lor and Malecki 2006	0.18	Johnson and Dinsmore 1985	87	Conway 1995	24.5	Conway 1995
31	b	<i>Anas wyvilliana</i>	Anseriformes	8.3	Engilis et al. 2002	n/a		532	Engilis et al. 2002	31.5	Carboneras 1992b
32	b	<i>Gallinula galeata</i>	Gruiformes	6.7	Helm et al. 1987	1.22	Matthews 1983	383	Bannor and Kiviat 2002	33.5	Bannor and Kiviat 2002
32	b	<i>Ixobrychus exilis</i>	Ciconiiformes	3.8	Poole et al. 2009	223.2	Griffin et al. 2009	80	Poole et al. 2009	32	Poole et al. 2009
32	b	<i>Porphyrio martinica</i>	Gruiformes	5.2	West and Hess 2002	1.03	West and Hess 2002	248	West and Hess 2002	31.5	West and Hess 2002
33	b	<i>Agelaius phoeniceus</i>	Passeriformes	5.6	Yasukawa and Searcy 1995	0.16	Yasukawa and Searcy 1995	57.2	Yasukawa and Searcy 1995	22	Rosenthal 2004
33	b	<i>Anas discors</i>	Anseriformes	10.1	Rohwer et al. 2002	0.69	Stewart and Titman 1980	431	Rohwer et al. 2002	38.5	Mingo 2008
33	b	<i>Anas platyrhynchos</i>	Anseriformes	8.7	Drilling et al. 2002	225	Gilmer et al. 1975	1150	Drilling et al. 2002	57.5	Drilling et al. 2002
33	b	<i>Chlidonias niger</i>	Charadriiformes	2.8	Maxson et al. 2007	452.16	Chapman Mosher 1986	55	Heath et al. 2009	24.5	Heath et al. 2009
33	b	<i>Cistothorus palustris</i>	Passeriformes	5	Kroodsma and Verner 1997	0.1	Leonard and Picman 1986, Verner 1965	11.5	Kroodsma and Verner 1997	122	Kroodsma and Verner 1997
33	b	<i>Cistothorus platensis</i>	Passeriformes	7	Herkert et al. 2001	0.178	Herkert et al. 2001	8.5	Herkert et al. 2001	11	Herkert et al. 2001
33	b	<i>Empidonax alnorum</i>	Passeriformes	3.7	Lowther 1999	2.15	Lowther 1999	13	Lowther 1999	15	Lowther 1999
33	b	<i>Fulica americana</i>	Gruiformes	8.5	Brisbin et al. 2002	0.15	Brisbin et al. 2002	637.5	Brisbin et al. 2002	37.5	Brisbin et al. 2002
33	b	<i>Gallinago delicata</i>	Charadriiformes	3.9	Mueller 1999	235	Cline and Haig 2011	100	Mueller 1999	28	Mueller 1999
33	b	<i>Geothlypis trichas</i>	Passeriformes	3.5	Abroe et al.	0.38	Guzy and	9.5	Guzy and	12	Guzy and

					2007		Ritchison 1999		Ritchison 1999		Ritchison 1999
33	b	<i>Grus canadensis</i>	Gruiformes	2	Tacha et al. 1992	85.1	Tacha et al. 1992	3545	Tacha et al. 1992	120	Tacha et al. 1992
33	b	<i>Melospiza georgiana</i>	Passeriformes	3.8	Mowbray 1997	0.17	Mowbray 1997	17.8	Mowbray 1997	13.5	Mowbray 1997
33	b	<i>Podilymbus podiceps</i>	Podicipediformes								
33	b	<i>Porzana carolina</i>	Gruiformes	10.2	Melvin and Gibbs 2012	0.19	Johnson and Dinsmore 1985	80	Melvin and Gibbs 2012	22.5	Melvin and Gibbs 2012
33	b	<i>Rallus limicola</i>	Gruiformes	7.5	Kaufmann 1989	0.18	Johnson and Dinsmore 1985	87	Conway 1995	24.5	Conway 1995
33	b	<i>Setophaga petechia</i>	Passeriformes	4.1	Lowther et al. 1999	0.284	Lowther et al. 1999	10	Lowther et al. 1999	12.5	Lowther et al. 1999
33	b	<i>Xanthocephalus xanthocephalus</i>	Passeriformes	3.8	Twedt and Crawford 1995	78.5	Twedt and Crawford 1995	80	Twedt and Crawford 1995	240	Twedt and Crawford 1995
24	r	<i>Emydoidea blandingii</i>	Testudine	8	MacCulloch and Weller 1988	12	Millar and Blouin-Demers 2011	1601	Millar 2010	22.4	Millar 2010
24	r	<i>Sternotherus odoratus</i>	Testudine	4.7	COSEWIC 2002	6.2	Carrière 2007	173.5	Edmonds 1999	10.1	Edmonds 1999
34	r	<i>Chrysemys picta marginata</i>	Testudine	12.4	Rowe et al. 2003	1.2	Rowe 2003	354.4	Rowe 2003	14.4	Rowe 2003
34	r	<i>Emydoidea blandingii</i>	Testudine	10.2	Congdon and van Loben Sels 1993	48.99	Congdon et al. 2011	1200	Congdon and van Loben Sels 1993	19.5	Congdon and van Loben Sels 1993
34	r	<i>Nerodia erythrogaster neglecta</i>	Squamata	17.7	Gibbons and Dorcas 2004	15.8	Roe et al. 2004	401	Gibbons and Dorcas 2004	99	Conant and Collins 1991
34	r	<i>Nerodia sipedon sipedon</i>	Squamata	27.2	Ernst and Ernst 2003	4	Roe et al. 2004	165	Roe et al. 2004	83.9	Conant and Collins 1991
35	r	<i>Chrysemys picta</i>	Testudine	20	Christiansen and Moll 1973	1.2	Rowe 2003	354.4	Rowe 2003	14.4	Rowe 2003
36	r	<i>Chrysemys picta marginata</i>	Testudine	14.4	Whillans and Crossman	1.2	Rowe 2003	500	Brooks et al 2003	15	MacCulloch 2002

					1977; Ernst and Lovich 2009						
37	r	<i>Clemmys guttata</i>	Testudine	5.3	Joyal 1996	7.6	Joyal et al. 2001	183	Litzgus and Mousseau 2006	11.2	Litzgus and Brooks 1998
37	r	<i>Emydoidea blandingii</i>	Testudine	8.5	Joyal et al. 2000	36.3	Joyal et al. 2001	1342.5	Joyal et al. 2000	20.9	Joyal et al. 2000
38	r	<i>Chrysemys picta</i>	Testudine	14.4	Whillans and Crossman 1977; Ernst and Lovich 2009	1.2	Rowe 2003	392.5	Steen and Gibbs 2004	13.3	Gibbs et al. 2007
39	r	<i>Nerodia erythrogaster neglecta</i>	Squamata	17.7	Gibbons and Dorcas 2004	26.5	Gibbons and Dorcas 2004	401	Gibbons and Dorcas 2004	99	Conant and Collins 1991
39	r	<i>Nerodia rhombifer rhombifer</i>	Squamata	23.1	Plummer 1992	3.5	Gibbons and Dorcas 2004	553.4	Plummer 1992	99	Conant and Collins 1991
39	r	<i>Nerodia sipedon pleuralis</i>	Squamata	27.2	Ernst and Ernst 2003	2.82	Roth and Greene 2006	n/a		79	Conant and Collins 1991
40	r	<i>Glyptemys muhlenbergii</i>	Testudine	3.5	Whitlock 2002	0.55	Whitlock 2002	114.7	Whitlock 2002	8.3	Gibbs et al. 2007
41	r	<i>Apalone spinifera spinifera</i>	Testudine	35.8	Ernst and Lovich 2009; Robinson and Murphy 1978	1740.5	Galois et al. 2002	n/a		25.6	Gibbs et al. 2007
41	r	<i>Chelydra serpentina serpentina</i>	Testudine	35.7	Iverson et al. 1997	11.13	Paisley et al. 2009	5600	Paisley et al. 2009	29.9	Paisley et al. 2009
41	r	<i>Chrysemys picta marginata</i>	Testudine	12.4	Rowe et al. 2003	32.45	Tran et al. 2007	354.4	Rowe 2003	14.4	Rowe 2003
41	r	<i>Trachemys scripta elegans</i>	Testudine	36.6	Tucker et al. 1998; Tucker 2001	327.6	Jaeger and Cobb 2012	1719.7	Tucker et al. 1998; Tucker 2001	19.3	Gibbons and Lovich 1990
42	r	<i>Chelodina longicollis</i>	Testudine	8.4	Kennett and Georges 1990;	11.5	Roe and Georges 2008	602	Jarrett 2011	25	Jarrett 2011

43	r	<i>Chelydra serpentina</i>	Testudine	30.9	Arthur Georges, personal communication	5.98	Pettit et al. 1995	6005	Steen and Gibbs 2004	28.3	Gibbs et al. 2007
43	r	<i>Chrysemys picta</i>	Testudine	14.4	Whillans and Crossman 1977	1.2	Rowe 2003	392.5	Steen and Gibbs 2004	13.3	Gibbs et al. 2007
44	r	<i>Chelydra serpentina</i>	Testudine	30.9	Petokas and Alexander 1980	5.98	Pettit et al. 1995	6005	Steen and Gibbs 2004	28.3	Gibbs et al. 2007
44	r	<i>Chrysemys picta</i>	Testudine	14.4	Whillans and Crossman 1977; Ernst and Lovich 2009	1.2	Rowe 2003	392.5	Steen and Gibbs 2004	13.3	Gibbs et al. 2007
1	a	<i>Bufo boreas</i>	Anura	12000	Samollow 1980, Vonesh and Cruz 2002	41.775	Browne and Paszkowski 2010	n/a		9	Russell et al. 2000
1	a	<i>Pseudacris maculata</i>	Anura	325	Whiting 2010	7.292	Spencer 1964	n/a		3	Russell et al. 2000
1	a	<i>Rana sylvatica</i>	Anura	876	Corn and Livo 1989	2.92	Rittenhouse and Semlitsch 2009	n/a		4	Russell et al. 2000
45	a	<i>Ambystoma maculatum</i>	Caudata	268.8	Karraker 2007	1.093	Semlitsch and Bodie 2003	n/a		20	Petranka 2010
45	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.9089	Baldwin et al. 2006			5.25	Gibbs et al. 2007
46	a	<i>Bufo bufo</i>	Anura	3100	Cooper et al. 2008	54.73	Sinsch 1988	n/a		11.5	Cooper et al. 2008
46	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a		4.3	Cooper et al. 2008
46	a	<i>Ichthyosaura alpestris</i>	Caudata	150	Griffiths 1996	3.05	Kovar et al. 2009	n/a		10	Spellerberg 2002
46	a	<i>Lissotriton vulgaris</i>	Caudata	180	Kuzmin 2013c	2.8	Kovar et al.	n/a		9	Spellerberg

46	a	<i>Rana esculenta</i>	Anura	11000	Cooper et al. 2008	1.766	2009 Holenweg Peter 2001	n/a	9	2002 Cooper et al. 2008	
46	a	<i>Rana lessonae</i>	Anura	2420	Kuzmin and Cavagnaro 2013	1.539	Holenweg Peter 2001	n/a	6.8	Cooper et al. 2008	
46	a	<i>Rana temporaria</i>	Anura	2585	Cooper et al 2008	9.397	Kovar et al. 2009	n/a	10	Cooper et al. 2008	
46	a	<i>Triturus cristatus</i>	Caudata	200	Griffiths 1996	20.82	Jehle and Arntzen 2000, Sinsch 2007	7.4	Jehle and Arntzen 2000	13	Spellerberg 2002
47	a	<i>Hemidactylum scutatum</i>	Caudata	47	Petranka 2010	3.54	Richmond 1999	n/a	7.5	Petranka 2010	
48	a	<i>Ambystoma tigrinum tigrinum</i>	Caudata	421	Petranka 2010	0.287	Madison and Farrand 1998	30.3	Madison and Farrand 1998	34	Petranka 2010
49	a	<i>Ambystoma maculatum</i>	Caudata	268.8	Karraker 2007	1.093	Semlitsch and Bodie 2003	n/a	20	Petranka 2010	
49	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a	8.9	Petranka 2010	
49	a	<i>Pseudacris crucifer</i>	Anura	900	Gibbs et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007	
49	a	<i>Rana catesbeiana</i>	Anura	16000	Gibbs et al. 2007	12.94	Semlitsch and Bodie 2003	n/a	12	Conant and Collins 1991; Gibbs et al. 2007	
49	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	3.83	Lamoureux et al 2002	n/a	7.3	Gibbs et al. 2007	
49	a	<i>Rana palustris</i>	Anura	2500	Gibbs et al. 2007	n/a		n/a	6	Gibbs et al. 2007	
49	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007	
50	a	<i>Pseudacris regilla</i>	Anura	325	Whiting 2010	7.292	Spencer 1964	n/a	2.8	Conant and Collins	

50	a	<i>Rana catesbeiana</i>	Anura	16000	Gibbs et al. 2007	12.94	Semlitsch and Bodie 2003	n/a		12	1991 Conant and Collins 1991; Gibbs et al. 2007
50	a	<i>Rana draytonii</i>	Anura	2000	Fellers 2013	94.985	Bulger et al. 2003	131	Bulger et al. 2003	10.5	Bulger et al. 2003
51	a	<i>Ichthyosaura alpestris</i>	Caudata	150	Griffiths 1996	3.05	Kovar et al. 2009	n/a		10	Spellerberg 2002
51	a	<i>Lissotriton vulgaris</i>	Caudata	180	Kuzmin 2013c	2.8	Kovar et al. 2009	n/a		9	Spellerberg 2002
52	a	<i>Lissotriton helveticus</i>	Caudata	365	van der Meijden and Cavagnaro 2013	1.766	Diego-Rasilla and Luengo 2007	n/a		6.5	Spellerberg 2002
53	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a		8.5	Fisher et al. 2007
53	a	<i>Hyla versicolor</i>	Anura	2000	Gibbs et al. 2007	9.075	Johnson et al 2007	n/a		4	Conant and Collins 1991; Gibbs et al. 2007
53	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a		2.5	Conant and Collins 1991; Gibbs et al. 2007
53	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a		8.5	Fisher et al. 2007
53	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007; Gilbert et al. 1994	135.95	Blomquist and Hunter 2009	n/a		7	Fisher et al. 2007
53	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a		5.3	Conant and Collins 1991; Gibbs et al. 2007
54	a	<i>Ambystoma macrodactylum</i>	Caudata	121	Howard and Wallace 1985	0.02	Pilliod and Fronzuto 2013	n/a		13.5	Petranka 2010
54	a	<i>Pseudacris regilla</i>	Anura	575	Rorabaugh and	3.97	Rorabaugh and	n/a		3.7	Rorabaugh

					Lannoo 2013		Lannoo 2013				and Lannoo 2013
54	a	<i>Rana luteiventris</i>	Anura	600	Reaser and Pilliod 2013	7.3	Pilliod et al. 2002	n/a		7.3	Russell et al. 2000
55	a	<i>Ambystoma tigrinum mavortium</i>	Caudata	2385	Rose and Armentrout 1976	0.923	Steen et al. 2006	34.3	Gray 2002	34	Petranka 2010
55	a	<i>Bufo cognatus</i>	Anura	11074	Krupa 1994	28.26	Graves and Krupa 2013	54	Gray 2002	6.9	Conant and Collins 1991
55	a	<i>Spea bombifrons</i>	Anura	1600	Woodward 1987	25.95	Landreth and Christensen 1971	11.2	Gray 2002	4.5	Conant and Collins 1991
55	a	<i>Spea multiplicata</i>	Anura	1062	Pfenning and Woodward 2005, Woodward 1987	n/a		8.7	Gray 2002	4.5	Conant and Collins 1991
56	a	<i>Ambystoma laterale</i>	Caudata	206	Petranka 2010	1.469	Faccio 2003	n/a		10.3	Petranka 2010
56	a	<i>Ambystoma maculatum</i>	Caudata	268.8	Karraker 2007	1.093	Semlitsch and Bodie 2003	n/a		20	Petranka 2010
56	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
56	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a		8.9	Petranka 2010
56	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	3.83	Lamoureux et al 2002	n/a		7.3	Gibbs et al. 2007
56	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007; Gilbert et al. 1994	135.95	Blomquist and Hunter 2009	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
56	a	<i>Rana septentrionalis</i>	Anura	1250	Shirose and Brooks 1995	n/a		n/a		6.3	Gibbs et al. 2007
56	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a		5.3	Conant and Collins

											1991; Gibbs et al. 2007
57	a	<i>Triturus cristatus</i>	Caudata	200	Griffiths 1996	20.82	Jehle and Arntzen 2000, Sinsch 2007	7.4	Jehle and Arntzen 2000	13	Spellerberg 2002
58	a	<i>Crinia signifera</i>	Anura	125	Hero et al. 2005	0.502	Westgate et al. 2012	n/a		2.5	Hero et al. 2005
58	a	<i>Limnodynastes dumerilii</i>	Anura	3900	Hero 2013	1.96	Carthew et al. 2009	n/a		6.9	Tyler and Knight 2009
58	a	<i>Limnodynastes peronii</i>	Anura	857	Hero et al. 2005	0.159	Martin Westgate, personal communication	n/a		5.9	Tyler and Knight 2009
58	a	<i>Litoria ewingii/verreauxii</i>	Anura	675	Hero and Bishop 2013, Hero et al. 2013, Turner 2004	n/a		n/a		3.4	Tyler and Knight 2009
58	a	<i>Paracrinia haswelli</i>	Anura	188.3	Anstis 2007	0.502	Westgate et al. 2012	n/a		3.3	Tyler and Knight 2009
59	a	<i>Crinia signifera</i>	Anura	125	Hero et al. 2005	0.502	Westgate et al. 2012	n/a		2.5	Hero et al. 2005
59	a	<i>Limnodynastes dumerilii</i>	Anura	3900	Hero 2013	1.96	Carthew et al. 2009	n/a		6.9	Tyler and Knight 2009
59	a	<i>Limnodynastes peronii</i>	Anura	857	Hero et al. 2005	0.159	Martin Westgate, personal communication	n/a		5.9	Tyler and Knight 2009
59	a	<i>Limnodynastes tasmaniensis</i>	Anura	1000	Wilson et al. 2012	n/a		n/a		3.8	Tyler and Knight 2009
59	a	<i>Litoria ewingii</i>	Anura	600	Hero and Bishop 2013	n/a		n/a		3.5	Tyler and Knight 2009

59	a	<i>Litoria fallax</i>	Anura	250	Turner 2004	n/a		n/a	2.6	Tyler and Knight 2009	
59	a	<i>Litoria raniformis</i>	Anura	3387	Heard et al. 2012a	3.14	Heard et al. 2012b	n/a	7.1	Tyler and Knight 2009	
59	a	<i>Litoria verreauxii</i>	Anura	750	Hero et al. 2013, Turner 2004	n/a		n/a	3.3	Tyler and Knight 2009	
60	a	<i>Bombina variegata</i>	Anura	72.5	Kuzmin 2013a	1.766	Beshkov and Jameson 1980, Hartel 2008	n/a	4.1	Spellerberg 2002	
60	a	<i>Bufo bufo</i>	Anura	3100	Cooper et al. 2008	54.73	Sinsch 1988	n/a	11.5	Cooper et al. 2008	
60	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008	
60	a	<i>Lissotriton vulgaris</i>	Caudata	180	Kuzmin 2013c	2.8	Kovar et al. 2009	n/a	9	Spellerberg 2002	
60	a	<i>Pelobates fuscus</i>	Anura	1740	Kuzmin and Andreone 2013	4.906	Hels 2002	n/a	7.3	Spellerberg 2002	
60	a	<i>Rana dalmatina</i>	Anura	910	Ponsero and Joly 1998	0.93	Ponsero and Joly 1998	n/a	7.5	Cooper et al. 2008	
60	a	<i>Rana esculenta</i>	Anura	11000	Cooper et al. 2008	1.766	Holenweg Peter 2001	n/a	9	Cooper et al. 2008	
60	a	<i>Rana temporaria</i>	Anura	2585	Cooper et al 2008	9.397	Kovar et al. 2009	n/a	10	Cooper et al. 2008	
60	a	<i>Triturus cristatus</i>	Caudata	200	Griffiths 1996	20.82	Jehle and Arntzen 2000, Sinsch 2007	7.4	Jehle and Arntzen 2000	13	Spellerberg 2002
61	a	<i>Litoria raniformis</i>	Anura	3387	Heard et al. 2012a	3.14	Heard et al. 2012b	n/a	7.1	Tyler and Knight 2009	
62	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a	7	Conant and Collins 1991; Gibbs et al. 2007	
62	a	<i>Hyla versicolor</i>	Anura	2000	Gibbs et al.	9.075	Johnson et al	n/a	4	Conant and	

					2007		2007			Collins 1991; Gibbs et al. 2007
62	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a	8.9	Petranka 2010
62	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
62	a	<i>Pseudacris triseriata</i>	Anura	1000	Gibbs et al. 2007, Cooper et al. 2008	0.785	Kramer 1973	n/a	3	Conant and Collins 1991; Gibbs et al. 2007
62	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a	8.5	Fisher et al. 2007
62	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007; Gilbert et al. 1994	135.95	Blomquist and Hunter 2009	n/a	7	Fisher et al. 2007
62	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
63	a	<i>Ambystoma maculatum</i>	Caudata	224	Petranka 2010	1.093	Semlitsch and Bodie 2003	n/a	20	Petranka 2010
63	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
64	a	<i>Ambystoma laterale</i>	Caudata	206	Petranka 2010	1.469	Faccio 2003	n/a	10.3	Petranka 2010
64	a	<i>Ambystoma maculatum</i>	Caudata	268.8	Karraker 2007	1.093	Semlitsch and Bodie 2003	n/a	20	Petranka 2010
64	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a	8.5	Fisher et al. 2007
64	a	<i>Hyla versicolor</i>	Anura	2000	Gibbs et al. 2007	9.075	Johnson et al 2007	n/a	4	Conant and Collins 1991; Gibbs

64	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a	8.9	et al. 2007 Petranka 2010
64	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
64	a	<i>Pseudacris triseriata</i>	Anura	1000	Gibbs et al. 2007, Cooper et al. 2008	0.785	Kramer 1973	n/a	3	Conant and Collins 1991; Gibbs et al. 2007
64	a	<i>Rana catesbeiana</i>	Anura	13000	Howard 1978	12.94	Semlitsch and Bodie 2003	n/a	12	Conant and Collins 1991; Gibbs et al. 2007
64	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a	8.5	Fisher et al. 2007
64	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007; Gilbert et al. 1994	135.95	Blomquist and Hunter 2009	n/a	7	Fisher et al. 2007
64	a	<i>Rana septentrionalis</i>	Anura	1250	Shirose and Brooks 1995	n/a		n/a	6.5	Fisher et al. 2007
64	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
65	a	<i>Ambystoma laterale/A. maculatum</i>	Caudata	259.4	Karraker 2007, Petranka 2010	1.28	Faccio 2003, Semlitsch and Bodie 2003	n/a	15.2	Petranka 2010
65	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a	8.5	Fisher et al. 2007
65	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a	8.9	Petranka 2010
65	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007

65	a	<i>Rana catesbeiana</i>	Anura	13000	Howard 1978	12.94	Semlitsch and Bodie 2003	n/a		12	Conant and Collins 1991; Gibbs et al. 2007
65	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a		8.5	Fisher et al. 2007
65	a	<i>Rana palustris</i>	Anura	2500	Gibbs et al. 2007	n/a		n/a		6.5	Fisher et al. 2007
65	a	<i>Rana septentrionalis</i>	Anura	1250	Shirose and Brooks 1995	n/a		n/a		6.5	Fisher et al. 2007
65	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a		5.3	Conant and Collins 1991; Gibbs et al. 2007
66	a	<i>Rana pipiens</i>	Anura	3045	Corn and Livo 1989	135.95	Blomquist and Hunter 2009	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
67	a	<i>Ichthyosaura alpestris</i>	Caudata	150	Griffiths 1996	3.05	Kovar et al. 2009	n/a		10	Spellerberg 2002
67	a	<i>Lissotriton helveticus</i>	Caudata	365	van der Meijden and Cavagnaro 2013	1.766	Diego-Rasilla and Luengo 2007	n/a		6.5	Spellerberg 2002
67	a	<i>Triturus cristatus</i>	Caudata	200	Griffiths 1996	20.82	Jehle and Arntzen 2000, Sinsch 2007	7.4	Jehle and Arntzen 2000	13	Spellerberg 2002
68	a	<i>Rana muscosa</i>	Anura	233	Vredenburg et al. 2013	0.53	Matthews and Pope 1999	26.9	Matthews and Miaud 2007	6	Matthews and Miaud 2007
69	a	<i>Ambystoma tigrinum</i>	Caudata	421	Petranka 2010	0.287	Madison and Farrand 1998	30.3	Madison and Farrand 1998	34	Petranka 2010
69	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2008	34.1	Forester et al. 2006	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
69	a	<i>Hyla versicolor</i>	Anura	2000	Gibbs et al.	9.075	Johnson et al	n/a		4	Conant and

					2007		2007				Collins 1991; Gibbs et al. 2007
69	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a		2.5	Conant and Collins 1991; Gibbs et al. 2007
69	a	<i>Pseudacris triseriata</i>	Anura	1000	Gibbs et al. 2007, Cooper et al. 2008	0.785	Kramer 1973	n/a		3	Conant and Collins 1991; Gibbs et al. 2007
69	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a		8.5	Fisher et al. 2007
69	a	<i>Rana palustris</i>	Anura	2500	Gibbs et al. 2007	n/a		n/a		6	
69	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007	135.95	Blomquist and Hunter 2009	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
70	a	<i>Ambystoma tigrinum tigrinum</i>	Caudata	421	Petranka 2010	0.287	Madison and Farrand 1998	30.3	Madison and Farrand 1998	34	Petranka 2010
70	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2008	34.1	Forester et al. 2006	n/a		7	Conant and Collins 1991; Gibbs et al. 2007
70	a	<i>Bufo cognatus</i>	Anura	11075	Krupa 1995	28.26	Graves and Krupa 2013	54	Gray 2002	6.9	Conant and Collins 1991
70	a	<i>Hyla versicolor/H. chrysoscelis</i>	Anura	1683	Ritke et al. 1990, Jacqueline Doyle, personal communication	9.075	Johnson et al 2007	n/a		4	Conant and Collins 1991
70	a	<i>Pseudacris triseriata</i>	Anura	1000	Gibbs et al. 2007, Cooper et al. 2008	0.785	Kramer 1973	n/a		3	Conant and Collins 1991; Gibbs

70	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007	135.95	Blomquist and Hunter 2009	n/a	7	et al. 2007 Conant and Collins 1991; Gibbs et al. 2007
70	a	<i>Rana sylvatica</i>	Anura	650	Berven 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
71	a	<i>Engystomops pustulosus</i>	Anura	234.2	Ryan 1983	2.005	Marsh et al. 1999	n/a	2.8	Marsh et al. 1999
72	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	1.47	Oldham 1967	n/a	8.5	Fisher et al. 2007
73	a	<i>Bufo fowleri</i>	Anura	3700	Green 2012	0.64	Boenke 2011	n/a	6.3	Conant and Collins 1991; Gibbs et al. 2007
73	a	<i>Hyla versicolor</i>	Anura	1305	Jacqueline Doyle, personal communication	9.075	Johnson et al 2007	n/a	4	Conant and Collins 1991; Gibbs et al. 2007
73	a	<i>Pseudacris crucifer</i>	Anura	900	Gibbs et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
73	a	<i>Rana catesbeiana</i>	Anura	16000	Gibbs et al. 2007	12.94	Semlitsch and Bodie 2003	n/a	12	Conant and Collins 1991; Gibbs et al. 2007
74	a	<i>Rana virgatipes</i>	Anura	400	Mitchell 2012	0.816	Given 1988	n/a	5.4	Conant and Collins 1991
75	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008
76	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008
77	a	<i>Acris crepitans</i>	Anura	248	Cooper et al.	0.132	Gray 1983,	n/a	2.5	Conant and

					2008			Lemckert 2004			Collins 1991
77	a	<i>Bufo americanus</i>	Anura	6000	Cooper et al. 2008	34.1	Forester et al. 2006	n/a	7	Conant and Collins 1991	
77	a	<i>Hyla chrysoscelis</i>	Anura	2060	Ritke et al. 1990	n/a		9.4	Ritke et al. 1990	5	Ritke et al. 1990
77	a	<i>Hyla versicolor</i>	Anura	1305	Jacqueline Doyle, personal communication	9.075	Johnson et al 2007	n/a	4	Conant and Collins 1991	
77	a	<i>Pseudacris triseriata</i>	Anura	1000	Cooper et al. 2008	0.785	Kramer 1973	n/a	3	Conant and Collins 1991	
77	a	<i>Rana catesbeiana</i>	Anura	13000	Howard 1978	12.94	Semlitsch and Bodie 2003	n/a	12	Conant and Collins 1991; Gibbs et al. 2007	
77	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007	135.95	Blomquist and Hunter 2009	n/a	7	Conant and Collins 1991	
78	a	<i>Bombina variegata</i>	Anura	72.5	Kuzmin 2013a	1.766	Beshkov and Jameson 1980, Hartel 2008	n/a	4.1	Spellerberg 2002	
78	a	<i>Bufo bufo</i>	Anura	3100	Cooper et al. 2008	54.73	Sinsch 1988	n/a	11.5	Cooper et al. 2008	
78	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008	
78	a	<i>Lissotriton vulgaris</i>	Caudata	180	Kuzmin 2013c	2.8	Kovar et al. 2009	n/a	9	Spellerberg 2002	
78	a	<i>Rana dalmatina</i>	Anura	910	Ponsero and Joly 1998	0.93	Ponsero and Joly 1998	n/a	7.5	Cooper et al. 2008	
78	a	<i>Rana esculenta</i>	Anura	11000	Cooper et al. 2008	1.766	Holenweg Peter 2001	n/a	9	Cooper et al. 2008	
78	a	<i>Rana temporaria</i>	Anura	2585	Cooper et al 2008	9.397	Kovar et al. 2009	n/a	10	Cooper et al. 2008	
78	a	<i>Salamandra</i>	Caudata	40	Rebelo and	0.13	Schulte et al.	n/a	20	Griffiths	

<i>salamandra</i>				Leclair 2003; Warburg et al. 1979	2007	1996, Spellerberg 2002					
78	a	<i>Triturus cristatus</i>	Caudata	200	Griffiths 1996	20.82	Jehle and Arntzen 2000, Sinsch 2007	7.4	Jehle and Arntzen 2000	13	Spellerberg 2002
79	a	<i>Bufo bufo</i>	Anura	3100	Cooper et al. 2008	54.73	Sinsch 1988	n/a		11.5	Cooper et al. 2008
79	a	<i>Bufo calamita</i>	Anura	3500	Cooper et al. 2008	14.655	Miaud and Sanuy 2005	n/a		8	Cooper et al. 2008
79	a	<i>Ichthyosaura alpestris</i>	Caudata	150	Griffiths 1996	3.05	Kovar et al. 2009	n/a		10	Spellerberg 2002
80	a	<i>Ambystoma maculatum/A. jeffersonianum</i>	Caudata	230.4	Petranka 2010	1.899	Faccio 2003, Semlitsch and Bodie 2003	n/a		17.2	Conant and Collins 1991, Petranka 2010
80	a	<i>Bufo americanus/B. woodhouseii fowleri</i>	Anura	5850	Gibbs et al. 2007, Green 2013	17.37	Boenke 2011, Forester et al. 2006	n/a		6.6	Conant and Collins 1991; Gibbs et al. 2007
80	a	<i>Hyla versicolor</i>	Anura	1305	Jacqueline Doyle, personal communication	9.075	Johnson et al 2007	n/a		4	Conant and Collins 1991; Gibbs et al. 2007
80	a	<i>Notophthalmus viridescens</i>	Caudata	304	Petranka 2010	0.045	Healy 1975	n/a		8.9	Petranka 2010
80	a	<i>Pseudacris crucifer</i>	Anura	900	Gibbs et al. 2007	5.89	Delzell 1958	n/a		2.5	Conant and Collins 1991; Gibbs et al. 2007
80	a	<i>Rana catesbeiana</i>	Anura	16000	Gibbs et al. 2007	12.94	Semlitsch and Bodie 2003	n/a		12	Conant and Collins 1991; Gibbs et al. 2007
80	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	3.83	Lamoureux et al 2002	n/a		7.3	Gibbs et al. 2007

80	a	<i>Rana palustris</i>	Anura	2500	Gibbs et al. 2007	n/a	n/a	6	Gibbs et al. 2007
80	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	5.3	Conant and Collins 1991; Gibbs et al. 2007
81	a	<i>Rana pipiens</i>	Anura	3000	Gibbs et al. 2007	135.95	Blomquist and Hunter 2009	7	Conant and Collins 1991; Gibbs et al. 2007
82	a	<i>Acris crepitans</i>	Anura	300	Gibbs et al. 2007	0.132	Gray 1983, Lemckert 2004	2.5	Conant and Collins 1991; Gibbs et al. 2007
82	a	<i>Ambystoma texanum</i>	Caudata	625	Trauth 2012	0.21	Semlitsch and Bodie 2003	15	Petranka 2010
82	a	<i>Bufo americanus</i>	Anura	6000	Cooper et al. 2008	34.1	Forester et al. 2006	7	Conant and Collins 1991
82	a	<i>Hyla versicolor/H. chrysoscelis</i>	Anura	1683	Ritke et al. 1990, Jacqueline Doyle, personal communication	9.075	Johnson et al 2007	4	Conant and Collins 1991
82	a	<i>Pseudacris crucifer</i>	Anura	900	Gibbs et al. 2007	5.89	Delzell 1958	2.5	Conant and Collins 1991; Gibbs et al. 2007
82	a	<i>Pseudacris maculata</i>	Anura	325	Whiting 2010	7.292	Spencer 1964	2.8	Conant and Collins 1991
82	a	<i>Rana blairi/R. sphenoccephalus</i>	Anura	4625	Crawford et al. 2013, Gibbs et al. 2007	706.5	Meade 2008	6.8	Conant and Collins 1991
82	a	<i>Rana catesbeiana</i>	Anura	13000	Howard 1978	12.94	Semlitsch and Bodie 2003	12	Conant and Collins

82	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	3.83	Lamoureux et al 2002	n/a	6.5	1991; Gibbs et al. 2007 Conant and Collins 1991
83	a	<i>Dendropsophus elianeae</i>	Anura	n/a		n/a		n/a	n/a	
83	a	<i>Dendropsophus minutus</i>	Anura	400	Lima et al. 2006	n/a		n/a	2.3	Lima et al. 2006
83	a	<i>Dendropsophus nanus</i>	Anura	242	Prado and Haddad 2005	n/a		0.6	2	Prado and Haddad 2005, Prado and Haddad 2005
83	a	<i>Dermatonotus muelleri</i>	Anura	6000	Fausto Nomura, personal communication	n/a		n/a	4.5	Provete 2013a
83	a	<i>Elachistocleis bicolor</i>	Anura	478	Prado and Haddad 2005	n/a		1.4	2.6	Prado and Haddad 2005
83	a	<i>Eupemphix nattereri</i>	Anura	2516	Giaretta and Facure 2006	n/a		n/a	4.2	Giaretta and Facure 2006
83	a	<i>Hypsiboas albopunctatus</i>	Anura	899	Guimarães et al. 2011	n/a		n/a	5.3	Brasileiro et al. 2005, Guimarães et al. 2011
83	a	<i>Hypsiboas raniceps</i>	Anura	1991	Prado and Haddad 2005	n/a		12.3	6	Prado and Haddad 2005
83	a	<i>Leptodactylus chaquensis</i>	Anura	4936	Prado and Haddad 2005	n/a		35.3	7.1	Prado and Haddad 2005
83	a	<i>Leptodactylus fuscus</i>	Anura	214	Prado and Haddad 2005	n/a		8.6	4.4	Prado and Haddad 2005

83	a	<i>Leptodactylus labyrinthicus</i>	Anura	2101	Silva et al. 2005	n/a	n/a	13.2	Silva et al. 2005	
83	a	<i>Leptodactylus latrans</i>	Anura	n/a		n/a	n/a	10	Dietl et al. 2013	
83	a	<i>Leptodactylus mystacinus</i>	Anura	401	Filho and Giaretta 2008	n/a	n/a	5.8	Reisman 2013	
83	a	<i>Leptodactylus podicipinus</i>	Anura	2102	Prado and Haddad 2005	n/a	5.2	Prado et al. 2000	Prado et al. 2000	
83	a	<i>Physalaemus centralis</i>	Anura	1872	Provete 2013b	n/a	n/a	3.5	Provete 2013b	
83	a	<i>Physalaemus cuvieri</i>	Anura	473.5	Barreto and Andrade 1995	n/a	n/a	2.9	Mijares et al. 2010	
83	a	<i>Pseudis platensis</i>	Anura	1834	Prado and Haddad 2005, Cynthia Prado, personal communication	n/a	27.2	Prado and Haddad 2005, Cynthia Prado, personal communication	Fabrezi et al. 2009	
83	a	<i>Pseudopaludicola falcipes</i>	Anura	32	Prado and Haddad 2005	n/a	0.3	Prado and Haddad 2005	Prado and Haddad 2005	
83	a	<i>Rhinella schneideri</i>	Anura	n/a		n/a	n/a	n/a		
83	a	<i>Scinax fuscomarginatus</i>	Anura	337	Prado and Haddad 2005	n/a	0.5	Prado and Haddad 2005	Prado and Haddad 2005	
83	a	<i>Scinax fuscovarius</i>	Anura	2892	Rodrigues et al. 2005	0.442	Domingos Rodrigues, personal communication	8.9	Rodrigues et al. 2005	Rodrigues et al. 2005
83	a	<i>Scinax similis</i>	Anura	n/a		n/a	n/a	n/a		
84	a	<i>Acris crepitans</i>	Anura	300	Gibbs et al. 2007	0.132	Gray 1983, Lemckert 2004	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
84	a	<i>Bufo americanus</i>	Anura	8000	Gibbs et al. 2007	34.1	Forester et al. 2006	n/a	7	Conant and Collins 1991; Gibbs et al. 2007

84	a	<i>Bufo fowleri</i>	Anura	3700	Green 2012	0.64	Boenke 2011	n/a	6.3	Conant and Collins 1991; Gibbs et al. 2007
84	a	<i>Hyla versicolor/chryso scelis</i>	Anura	1750	Gibbs et al. 2007, Resetarits and Wilbur 1989	9.075	Johnson et al 2007	n/a	4	Conant and Collins 1991
84	a	<i>Notophthalmus viridescens</i>	c	304	Petranka 2010	0.045	Healy 1975	n/a	8.9	Petranka 2010
84	a	<i>Pseudacris crucifer</i>	Anura	900	Gibbs et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
84	a	<i>Rana catesbeiana</i>	Anura	16000	Gibbs et al. 2007	12.94	Semlitsch and Bodie 2003	n/a	12	Conant and Collins 1991; Gibbs et al. 2007
84	a	<i>Rana clamitans</i>	Anura	3250	Gibbs et al. 2007	3.83	Lamoureux et al 2002	n/a	6.5	Conant and Collins 1991
84	a	<i>Rana palustris</i>	Anura	2500	Gibbs et al. 2007	n/a		n/a	6	Conant and Collins 1991
84	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
84	a	<i>Rana utricularia</i>	Anura	4000	Gibbs et al. 2007	706.5	Meade 2008	n/a	6.3	Gibbs et al. 2007
85	a	<i>Ambystoma maculatum</i>	Caudata	257	Petranka 2010	1.093	Semlitsch and Bodie 2003	n/a	20	Petranka 2010
85	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
86	a	<i>Crinia parinsignifera</i>	Anura	n/a		n/a		n/a	2.1	Tyler and

86	a	<i>Crinia signifera</i>	Anura	125	Hero et al. 2005	0.502	Westgate et al. 2012	n/a	2.5	Knight 2009 Hero et al. 2005
86	a	<i>Limnodynastes dumerilii</i>	Anura	3900	Hero 2013	1.96	Carthew et al. 2009	n/a	6.9	Tyler and Knight 2009
86	a	<i>Limnodynastes peronii</i>	Anura	857	Hero et al. 2005	0.159	Martin Westgate, personal communication	n/a	5.9	Tyler and Knight 2009
86	a	<i>Limnodynastes tasmaniensis</i>	Anura	1000	Wilson et al. 2012	n/a		n/a	3.8	Tyler and Knight 2009
86	a	<i>Litoria ewingii/L. paraewingi</i>	Anura	600	Hero and Bishop 2013	n/a		n/a	3.5	Tyler and Knight 2009
86	a	<i>Litoria peronii</i>	Anura	1750	Sherman et al. 2008	n/a		n/a	5.2	Tyler and Knight 2009
87	a	<i>Ambystoma maculatum</i>	Caudata	268.8	Karraker 2007	1.093	Semlitsch and Bodie 2003	n/a	20	Petranka 2010
87	a	<i>Rana sylvatica</i>	Anura	690	Karraker and Gibbs 2009	2.909	Baldwin et al 2006	n/a	5.3	Conant and Collins 1991; Gibbs et al. 2007
88	a	<i>Rana arvalis</i>	Anura	1500	Spellerberg 2002, Kuzmin 2013d	14.92	Kovar et al. 2009	n/a	7	Cooper et al. 2008
89	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008
90	a	<i>Pseudacris crucifer</i>	Anura	900	Fisher et al. 2007	5.89	Delzell 1958	n/a	2.5	Conant and Collins 1991; Gibbs et al. 2007
90	a	<i>Pseudacris triseriata</i>	Anura	1000	Gibbs et al. 2007, Cooper	0.785	Kramer 1973	n/a	3	Conant and Collins

					et al. 2008					1991; Gibbs et al. 2007
91	a	<i>Bombina variegata</i>	Anura	63	Barandun et al. 1997	1.65	Barandun and Reyer 1998	n/a	4.1	Spellerberg 2002
91	a	<i>Bufo bufo</i>	Anura	3100	Cooper et al. 2008	54.73	Sinsch 1988	n/a	11.5	Cooper et al. 2008
91	a	<i>Bufo calamita</i>	Anura	3500	Cooper et al. 2008	14.655	Miaud and Sanuy 2005	n/a	8	Cooper et al. 2008
91	a	<i>Hyla arborea</i>	Anura	1100	Kuzmin 2013b	7.864	Pellet et al. 2006	n/a	4.3	Cooper et al. 2008
91	a	<i>Ichthyosaura alpestris</i>	Caudata	150	Griffiths 1996	3.05	Kovar et al. 2009	n/a	10	Spellerberg 2002
91	a	<i>Lissotriton helveticus</i>	Caudata	365	van der Meijden and Cavagnaro 2013	1.766	Diego-Rasilla and Luengo 2007	n/a	6.5	Spellerberg 2002
91	a	<i>Lissotriton vulgaris</i>	Caudata	180	Kuzmin 2013c	2.8	Kovar et al. 2009	n/a	9	Spellerberg 2002
91	a	<i>Rana dalmatina</i>	Anura	910	Ponsero and Joly 1998	0.93	Ponsero and Joly 1998	n/a	7.5	Cooper et al. 2008
91	a	<i>Rana ridibunda</i>	Anura	1200	Cooper et al 2008	2.834	Holenweg Peter 2001	n/a	10	Cooper et al. 2008
91	a	<i>Rana temporaria</i>	Anura	2585	Cooper et al 2008	9.397	Kovar et al. 2009	n/a	10	Cooper et al. 2008

*Citations for studies included in the meta-analysis are provided in Appendix A

[†]Taxa: m = mammal, b = bird, r = reptile, a = amphibian

^aRepro = reproductive rate = mean litter or clutch size multiplied by the mean number of litters or clutches per year

^bHome Range = mean annual home range or seasonal migration distance across both sexes

^cMass = body mass = average body mass of the two sexes

^dLength = body length = average total body length of the two sexes

[§]Source = citations for studies used for species trait information are provided below

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Appendix C: Results from univariate mixed-effects meta-analytic models testing the effect of mobility (home range size and body length) and reproductive rate on species responses to wetland habitat loss at the population-level across and within taxonomic groups (Chapter 2).

Group	Moderator variable	ES _{Zr} ^a	ES _r ^b	CI _{lower}	CI _{upper}	Q _E ^c	Q _M ^d	p (Q _M)	n
All Taxa	Reproductive Rate	-0.046	-0.046	-0.067	-0.025	672.762	18.829	<0.0001	334
	Home Range Size	-0.009	-0.009	-0.032	0.015	708.151	0.536	0.464	334
	Body Length	0.119	0.118	0.061	0.177	665.864	16.022	<0.0001	334
	Reproductive Rate + Body Length ^e					661.251	20.761	<0.0001	334
	<i>Reproductive Rate</i>	-0.032	-0.032	-0.061	-0.003				334
	<i>Body Length</i>	0.056	0.056	-0.025	0.137				334
	Taxon					621.388	25.169	<0.0001	334
	<i>Mammal</i>	0.303	0.294	0.125	0.48				6
	<i>Bird</i>	0.198	0.196	0.153	0.243				115
	<i>Reptile</i>	0.087	0.087	-0.002	0.175				24
	<i>Amphibian</i>	0.074	0.074	0.044	0.104				189
	Study Type					695.378	0.649	0.421	334
	<i>Amount-based</i>	0.108	0.107	0.08	0.134				256
	<i>Configuration-based</i>	0.132	0.132	0.08	0.185				78
Sampling Effort					689.673	4.341	0.114	334	

	<i>Dependent</i>	0.138	0.137	0.104	0.172				191
	<i>Independent</i>	0.091	0.091	0.051	0.131				123
	<i>Unknown</i>	0.07	0.07	-0.008	0.148				20
	Sampled Wetland Area					711.241	1.26	0.262	334
	<i>Included</i>	0.125	0.124	0.093	0.157				185
	<i>Not Included</i>	0.096	0.096	0.057	0.135				149
	Study Region					776.636	4.632	0.201	426
	<i>Natural</i>	0.113	0.113	0.046	0.180				58
	<i>Agricultural</i>	0.136	0.136	0.099	0.174				188
	<i>Rural</i>	0.122	0.121	0.084	0.160				132
	<i>Urban</i>	0.050	0.050	-0.021	0.021				88
	Number of Scales	-0.032	-0.032	-0.084	0.020	778.399	1.479	0.224	426
Birds	Reproductive Rate	-0.305	-0.295	-0.546	-0.063	158.508	6.09	0.014	115
	Home Range Size	-0.003	-0.003	-0.032	0.025	168.645	0.052	0.819	115
	Body Length	0.152	0.151	0	0.303	160.317	3.863	0.049	115
	Body Mass	-0.008	-0.008	-0.037	0.021	165.693	0.315	0.575	115
	Reproductive Rate + Body Length ^c					155.038	7.749	0.021	115
	<i>Reproductive Rate</i>	-0.253	-0.248	-0.506	-0.001				
	<i>Body Length</i>	0.103	0.102	-0.056	0.261				
	Order					151.493	9.887	0.195	115
	<i>Anseriformes</i>	0.072	0.072	-0.258	0.403				2

	<i>Charadriiformes</i>	0.143	0.142	0.043	0.243				20
	<i>Ciconiiformes</i>	0.25	0.244	0.163	0.336				28
	<i>Falconiformes</i>	0.07	0.069	-0.224	0.363				5
	<i>Gruiformes</i>	0.115	0.114	0.023	0.207				23
	<i>Passeriformes</i>	0.238	0.234	0.143	0.333				25
	<i>Pelecaniformes</i>	0.343	0.33	0.092	0.593				3
	<i>Podicipediformes</i>	0.265	0.259	0.132	0.398				9
	Study Type					168.005	0.851	0.356	115
	<i>Amount-based</i>	0.202	0.2	0.158	0.247				100
	<i>Configuration-based</i>	0.111	0.11	-0.079	0.3				15
	Sampling Effort					168.453	0.116	0.944	115
	<i>Dependent</i>	0.193	0.19	0.132	0.253				68
	<i>Independent</i>	0.205	0.202	0.14	0.269				46
	<i>Unknown</i>	0.151	0.15	-0.28	0.583				1
	Sampled Wetland Area					168.649	0.486	0.486	115
	<i>Included</i>	0.191	0.189	0.145	0.238				85
	<i>Not Included</i>	0.237	0.232	0.119	0.354				30
Reptiles	Reproductive Rate	0.044	0.044	-0.338	0.404	35.294	0.085	0.771	24
	Home Range Size	-0.033	-0.033	-0.139	0.073	34.748	0.37	0.543	24
	Body Length	0.039	0.039	-0.212	0.289	35.278	0.093	0.761	24
	Order		0						24
	<i>Squamata</i>	0.15	0.149	-0.025	0.326	33.956	0.664	0.415	5

	<i>Testudine</i>	0.067	0.067	-0.029	0.163				19
Study Type			0			25.549	6.104	0.014	24
	<i>Amount-based</i>	0.046	0.046	-0.034	0.126				18
	<i>Configuration-based</i>	0.309	0.299	0.116	0.501				6
Sampling Effort			0			27.064	4.626	0.099	24
	<i>Area-Dependent</i>	0.26	0.254	0.082	0.438				9
	<i>Area-Independent</i>	0.042	0.042	-0.047	0.13				14
	<i>Unknown</i>	0.09	0.09	-0.279	0.46				1
Sampled Wetland Area			0			31.256	1.993	0.158	24
	<i>Included</i>	0.044	0.044	0.028	0.306				13
	<i>Not Included</i>	0.167	0.165	-0.056	0.143				11
Amphibians	Reproductive Rate	-0.011	-0.011	-0.062	0.04	411.101	0.171	0.679	189
	Home Range Size	-0.019	-0.019	-0.057	0.018	411.004	1.018	0.313	189
	Body Length	-0.025	-0.025	-0.15	0.1	411.859	0.153	0.696	189
	Order					411.422	0.183	0.669	189
	<i>Anura</i>	0.07	0.07	0.034	0.105				145
	<i>Caudata</i>	0.085	0.084	0.026	0.143				44
	Family					366.036	18.081	0.054	189
	<i>Ambystomatidae</i>	0.024	0.023	-0.074	0.121				17
	<i>Bombinatoridae</i>	-0.01	-0.01	-0.212	0.191				3
	<i>Bufo</i>	-0.054	-0.054	-0.146	0.037				23
	<i>Hylidae</i>	0.124	0.123	0.059	0.189				44

	<i>Leiuperidae</i>	0.569	0.515	-0.449	1.588				1
	<i>Myobatrachidae</i>	0.031	0.031	-0.111	0.174				10
	<i>Pelobatidae</i>	-0.036	-0.036	-0.427	0.354				1
	<i>Plethodontidae</i>	-0.277	-0.27	-1.296	0.742				1
	<i>Ranidae</i>	0.083	0.083	0.031	0.136				62
	<i>Salamandridae</i>	0.119	0.118	0.048	0.19				26
	<i>Scaphiopodidae</i>	0.681	0.592	-0.029	1.391				1
	Study Type					403.452	2.844	0.092	189
	<i>Amount-based</i>	0.057	0.057	0.021	0.093				133
	<i>Configuration-based</i>	0.114	0.114	0.058	0.17				56
	Sampling Effort					402.658	4.624	0.099	189
	<i>Dependent</i>	0.102	0.102	0.61	0.143				109
	<i>Independent</i>	0.027	0.027	-0.027	0.082				62
	<i>Unknown</i>	0.067	0.067	-0.012	0.146				18
	Sampled Wetland Area					412.169	0.313	0.576	189
	<i>Included</i>	0.083	0.083	0.039	0.127				83
	<i>Not Included</i>	0.065	0.065	0.023	0.108				106
Ranidae	Reproductive Rate	0.01	0.01	-0.083	0.103	64.59	0.042	0.838	62
	Home Range Size	-0.015	-0.015	-0.074	0.044	63.421	0.253	0.615	62
	Body Length	-0.035	-0.035	-0.429	0.359	64.627	0.03	0.862	62
	Study Type					95.585	0.84	0.359	62
	<i>Amount-based</i>	0.071	0.071	0.021	0.121				47

<i>Configuration-based</i>	0.123	0.122	0.024	0.222				15
Sampling Effort					91.994	1.834	0.400	62
<i>Dependent</i>	0.107	0.107	0.046	0.167				39
<i>Independent</i>	0.036	0.035	-0.049	0.121				18
<i>Unknown</i>	0.073	0.073	-0.039	0.186				5
Sampled Wetland Area					94.134	2.859	0.091	62
<i>Included</i>	0.125	0.124	0.058	0.191				40
<i>Not Included</i>	0.049	0.049	-0.009	0.107				22

^a mean-weighted Z-transformed correlation coefficients (ES_{Zr})

^b back-transformed correlation coefficients (ESr)

^c residual heterogeneity

^d between group/model heterogeneity

^e multiple meta-regression

Appendix D: Descriptive statistics of species traits (untransformed values) across all wetland vertebrates and within taxonomic groups used in the meta-analysis (Chapter 2).

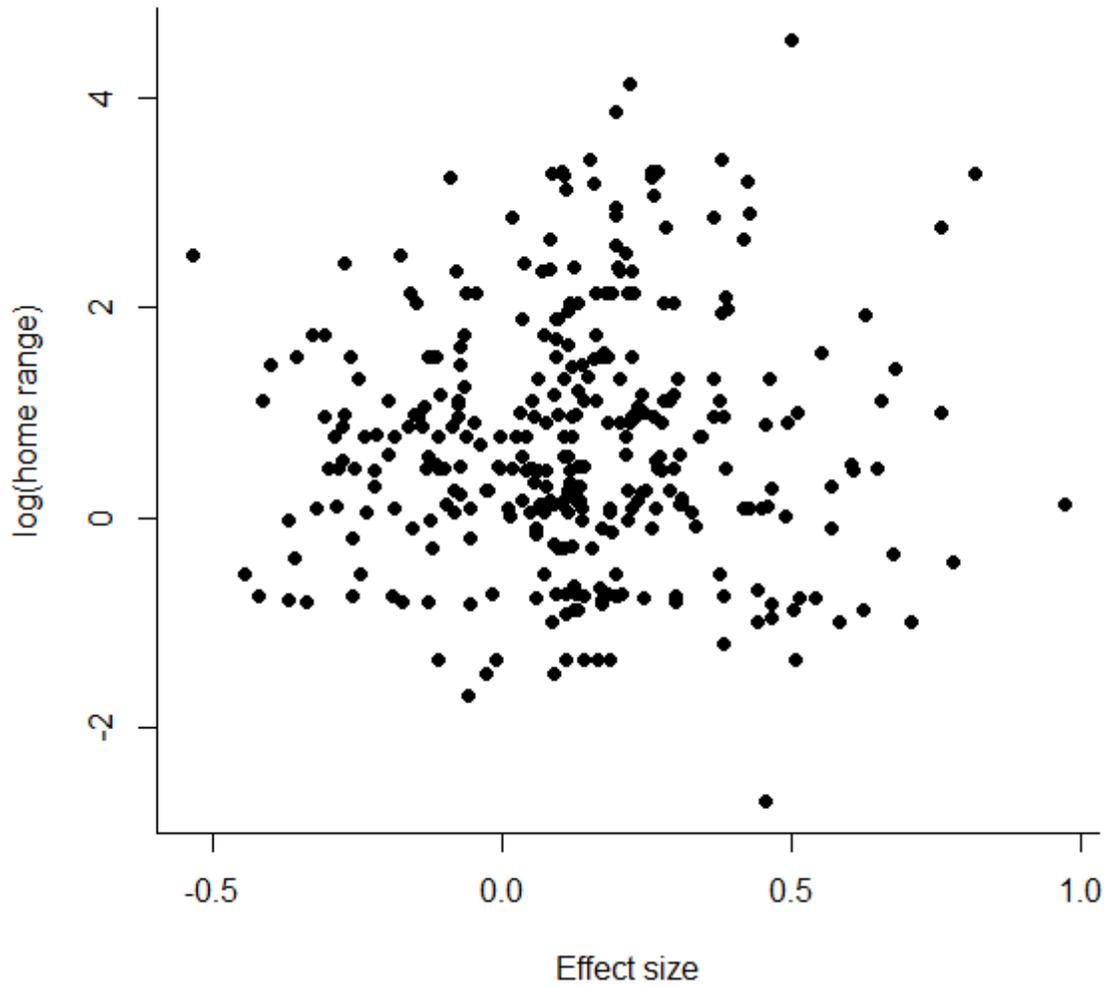
Group	Variable	Mean	Median	Min	Max
All Taxa (n = 334)	Home Range ^a	287.0	3.8	0.0	35600.0
	Body Length ^b	26.1	12.0	2.5	1200.0
	Reproductive rate ^c	1505.0	200.0	2.0	16000.0
Mammals (n = 6)	Home Range	3.0	0.5	0.1	13.0
	Body Length	38.3	25.0	12.4	120.0
	Reproductive rate	13.1	10.0	3.0	33.8
Birds (n = 115)	Home Range	775.9	9.7	0.0	35600.0
	Body Length	53.4	32.0	10.5	1200.0
	Reproductive rate	5.8	4.5	2.0	17.0
Reptiles (n = 24)	Home Range	96.1	6.1	0.6	1740.5
	Body Length	33.6	20.2	8.3	99.0
	Reproductive rate	18.1	14.4	3.5	36.6
Amphibians (n = 189)	Home Range	22.5	3.1	0.0	706.5
	Body Length	8.1	7.0	2.5	34.0
	Reproductive rate	2657.7	1000.0	40.0	16000.0

^aHome Range (ha) = mean annual home range or seasonal migration distance across both sexes

^bLength (cm) = body length = average total body length of the two sexes

^cReproductive rate = mean litter or clutch size multiplied by the mean number of litters or clutches per year

Appendix E: Scatterplot of log home range size (ha) and response to wetland habitat amount in a landscape (ES_{Z_r}) for all species included in the meta-analysis in Chapter 2, including mammals, birds, reptiles and amphibians ($n = 334$).



Appendix F: Estimated home range sizes and movement distances from radio-telemetry studies used to generate the landscape size (i.e. spatial scale) for each wetland species or species group. For Blanding’s turtle, movement distances were used because several studies excluded nesting forays and other terrestrial movements from home range calculations. Reprinted with permission of Elsevier © 2013.

Species or Species Group	Movement Distance (m)	Home Range (ha)	Source
Wetland Birds†		9.7	Bogner and Baldassarre 2002
		223.3	Griffin et al. 2009
Blanding's Turtle‡	489		Ross and Anderson 1990
	1400		Rowe and Moll 1991
	2050		Joyal et al. 2001
	1209		Piepgras and Lang 2000
	941		Rubin et al. 2001
	856		Grgurovic and Sievert 2005
	2500		Carrière 2007
	430		Beaudry et al. 2008
Eastern Musk Turtle†		50	Edmonds 1998
		155	Edmonds 1998
		6.2	Carrière 2007

† landscape size was calculated as the radius of circle with an area (ha) equal to the mean home range across studies

‡ landscape size was calculated as the mean movement distance across studies

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Appendix G: Results of principal components analysis (varimax rotation) of twelve candidate landscape-scale variables from 1000 random landscapes within the study area conducted for each species or species group. Reprinted with permission of Elsevier © 2013.

Landscapes defined for:	Principal Component	Landscape-Scale Variable Selected	Factor Loading	Variance Explained
Wetland Birds†	1	Number of Patches	0.89	27.4
	2	Wetland Amount	0.95	17.7
	3	Road Density	0.75	14.4
	4	Agriculture Amount	0.85	12.3
	5	Forest Cover	0.94	9.2
Blanding's Turtle‡	1	Number of Patches	0.87	28.2
	2	Road Density	0.84	19.4
	3	Wetland Amount	0.93	16.9
	4	Agriculture Amount	0.89	12.3
	5	Forest Cover	0.93	6.6
Eastern Musk Turtle§	1	Number of Patches	0.86	35.1
	2	Wetland Amount	0.92	16.4
	3	Road Density	0.86	13.9
	4	Forest Cover	0.89	11.9
	5	Agriculture Amount	0.96	10.4

† landscape radius = 625 m

‡ landscape radius = 1200 m

§ landscape radius = 500 m

Appendix H: Posterior probability of a non-zero coefficient [Pr(i)] for each of five landscape-scale predictors of occurrence of five wetland species. Posterior probabilities were determined by Bayesian model averaging. Landscape predictors wetland cover (area of wetland habitat as identified for each species or species group within a landscape), wetland configuration (the number of wetland habitat patches within a landscape) as a measure of habitat fragmentation per se, and three variables to measure matrix composition, forest cover (area of forest in a landscape), 4) agriculture cover (area of cropland in a landscape), 5) road density (km/km² of all road types within a landscape). Reprinted with permission of Elsevier © 2013.

Wetland Species	Pr(i)				
	Wetland Cover	Configuration	Forest Cover	Agriculture Cover	Road Density
Marsh Wren	1.0	0.09	0.32	0.43	0.18
Virginia Rail	0.83	0.20	0.08	0.31	0.09
Least Bittern	0.68	0.08	0.58	0.15	0.10
Blanding's Turtle	0.09	0.06	1.0	0.18	0.20
Eastern Musk Turtle	0.14	0.08	0.90	0.07	0.13

Appendix I: Post-hoc analysis results to compare the fit of global models of the relationship between Blanding’s turtle (*Emydoidea blandingii*) and eastern musk turtle (*Sternotherus odoratus*) occurrence and 1) landscape predictors only, 2) landscape predictors and proportion forested riparian (Riparian). Proportion forested riparian (zforriparian) is the total length of wetland edge immediately adjacent to forest in a landscape, divided by the total wetland edge length. A lower BIC indicates better model fit. Note: all variables were z-transformed. Reprinted with permission of Elsevier © 2013.

Species	Model	Parameters	BIC
Blanding's Turtle	Landscape only	Zwetland+Znp+Zforest+Zagric+Zrdens	92.94
	Landscape + Riparian	Zwetland+Znp+Zforest+Zagric+Zrdens+ zforriparian	97.03
Musk Turtle	Landscape only	Zwetland+Znp+Zforest+Zagric+Zrdens	99.67
	Landscape + Riparian	Zwetland+Znp+Zforest+Zagric+Zrdens+ zforriparian	106.12

Appendix J: Correlations of local variables affecting detection probabilities of focal wetland species with landscape-scale predictors. The expected relationship of each local variable with detectability is described. Local variables were: Julian Day, Air Temperature (°C) taken at each sampling visit, Cloud Cover, measured as the proportion of clouds visible at each sample visit, and Shoreline, measured as the amount of wetland edge in a landscape. Landscape predictors were: amount of wetland (Wetland), number of wetland patches (NP), amount of forest (Forest), amount of agriculture (Agric), and road density (RD). Reprinted with permission of Elsevier © 2013.

Species	Local Variable	Expected relationship with detectability	Correlation with Landscape Predictor				
			Wetland	NP	Forest	Agric	RD
Wetland Birds	Julian Day	Negative; vocalizations expected decrease towards end of sampling period	-0.04	-0.03	-0.12	0.09	0.04
Blanding's Turtle	Julian Day	Negative; basking expected decrease towards end of sampling period (i.e. spring; Sajwaj and Lang 2000)	-0.15	-0.1	-0.04	0.19	0.11
	Air Temp	Positive; higher air temperatures may increase aerial basking	-0.07	-0.04	0.16	0.01	0.04
	Cloud Cover	Negative; aerial basking decreases with greater cloud cover (Sajwaj and Lang 2000)	-0.11	-0.07	-0.16	0.23	0.15
	Shoreline	Increase; greater availability of wetland shoreline for aerial basking may increase detection	0.4	0.44	0.13	-0.29	-0.21
Musk Turtle	Julian Day	Negative; basking expected decrease towards end of sampling period (i.e. summer; Carrière, 2007)	-0.2	0.18	-0.01	0.08	0.03
	Air Temp	Positive; higher air temperatures may increase basking	0.19	-0.04	0.01	-0.02	-0.19

Appendix K: Studies included in the meta-analysis and associated species, effect sizes for wetland amount and forest amount (ESr), adjusted sample sizes (n), and study design categories (study type, sampling effort, and patch area) (Chapter 4).

Study*	Country	Taxa ⁺	Order	Species	Wetland ESr	Forest ESr	n	Study Type	Sampling Effort	Patch Area
1	Canada	m	Rodentia	<i>Castor canadensis</i>	0.28	-0.27	24	Amount	Independent	no
2	USA	m	Rodentia	<i>Oryzomys palustris</i>	0.19	0.03	35	Amount	Dependent	yes
3	USA	m	Rodentia	<i>Microtus pennsylvanicus</i>	0.08	-0.19	15	Amount	Dependent	yes
3	USA	m	Soricomorpha	<i>Sorex fumeus</i>	0.29	-0.22	15	Amount	Dependent	yes
3	USA	m	Rodentia	<i>Synaptomys cooperi</i>	0.13	0.54	15	Amount	Dependent	yes
4	USA	m	Lagomorpha	<i>Sylvilagus palustris hefneri</i>	0.30	0.05	39	Amount	Dependent	yes
5	USA	m	Rodentia	<i>Neofiber alleni</i>	0.42	-0.37	453	Configuration	Dependent	no
6	Japan	b	Pelecaniformes	<i>Ardea cinerea</i>	0.40	-0.05	32	Amount	Independent	yes
6	Japan	b	Pelecaniformes	<i>Egretta intermedia</i>	0.14	-0.26	32	Amount	Independent	yes
6	Japan	b	Pelecaniformes	<i>Nycticorax nycticorax</i>	0.43	-0.01	32	Amount	Independent	yes
7	USA	b	Ciconiiformes	<i>Botaurus lentiginosus</i>	0.14	-0.05	83	Amount	Independent	yes
7	USA	b	Gruiformes	<i>Fulica americana</i>	0.18	-0.23	190	Amount	Independent	yes
7	USA	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.21	-0.20	79	Amount	Independent	yes
7	USA	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.11	0.11	190	Amount	Independent	yes
7	USA	b	Gruiformes	<i>Porzana carolina</i>	0.10	-0.24	190	Amount	Independent	yes
8	Spain	b	Falconiformes	<i>Circus aeruginosus</i>	0.15	0.06	35	Configuration	Unknown	no
9	Netherlands	b	Passeriformes	<i>Acrocephalus scirpaceus</i>	0.09	0.08	316	Amount	Dependent	no
10	USA	b	Ciconiiformes	<i>Ardea herodias</i>	0.28	0.10	29	Amount	Independent	no
11	USA	b	Passeriformes	<i>Agelaius phoeniceus</i>	0.17	-0.67	11	Amount	Dependent	yes
11	USA	b	Passeriformes	<i>Cistothorus palustris</i>	0.09	0.12	11	Amount	Independent	yes

11	USA	b	Passeriformes	<i>Cistothorus platensis</i>	-0.40	0.22	11	Amount	Independent	yes
12	Brazil	b	Anseriformes	<i>Amazonetta brasiliensis</i>	0.43	-0.18	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Anas flavirostris</i>	0.28	-0.03	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Anas versicolor</i>	0.22	-0.20	5	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Aramides ypecaha</i>	0.09	-0.15	5	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Aramus guarauna</i>	0.05	-0.11	35	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Ardea alba</i>	0.08	0.24	6	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Ardea cocoi</i>	0.53	-0.15	6	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Butorides striata</i>	0.25	0.08	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Callonetta leucophrys</i>	0.12	-0.10	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Chauna torquata</i>	0.17	-0.15	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Ciconia maguari</i>	0.42	-0.22	5	Amount	Dependent	yes
12	Brazil	b	Falconiformes	<i>Circus buffoni</i>	0.14	0.03	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Dendrocygna bicolor</i>	0.22	-0.19	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Dendrocygna viduata</i>	0.27	-0.15	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Egretta thula</i>	0.25	0.14	9	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Fulica leucoptera</i>	0.02	-0.09	5	Amount	Dependent	yes
12	Brazil	b	Charadriiformes	<i>Gallinago paraguaiiae</i>	0.32	-0.38	5	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Gallinula galeata</i>	0.58	-0.15	35	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Gallinula melanops</i>	-0.10	0.06	5	Amount	Dependent	yes
12	Brazil	b	Charadriiformes	<i>Himantopus himantopus</i>	0.13	-0.13	42	Amount	Dependent	yes
12	Brazil	b	Charadriiformes	<i>Jacana jacana</i>	0.43	-0.15	42	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Mycteria americana</i>	0.19	-0.15	5	Amount	Dependent	yes
12	Brazil	b	Anseriformes	<i>Netta peposaca</i>	0.35	-0.16	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Nycticorax nycticorax</i>	0.06	-0.17	32	Amount	Dependent	yes
12	Brazil	b	Gruiformes	<i>Pardirallus sanguinolentus</i>	0.18	-0.14	5	Amount	Dependent	yes

12	Brazil	b	Ciconiiformes	<i>Phimosus infuscatus</i>	0.51	-0.13	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Platalea ajaja</i>	0.22	-0.14	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Plegadis chihi</i>	0.40	-0.17	9	Amount	Dependent	yes
12	Brazil	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.06	-0.21	35	Amount	Dependent	yes
12	Brazil	b	Podicipediformes	<i>Rollandia rolland</i>	0.43	-0.12	42	Amount	Dependent	yes
12	Brazil	b	Falconiformes	<i>Rostrhamus sociabilis</i>	0.26	-0.05	9	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Syrigma sibilatrix</i>	0.07	-0.05	5	Amount	Dependent	yes
12	Brazil	b	Ciconiiformes	<i>Theristicus caerulescens</i>	0.47	-0.21	5	Amount	Dependent	yes
13	Canada	b	Ciconiiformes	<i>Botaurus lentiginosus</i>	0.27	0.13	141	Amount	Independent	yes
13	Canada	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.47	-0.03	141	Amount	Independent	yes
13	Canada	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.30	-0.14	141	Amount	Independent	yes
13	Canada	b	Gruiformes	<i>Porzana carolina</i>	0.11	-0.28	141	Amount	Independent	yes
13	Canada	b	Gruiformes	<i>Rallus limicola</i>	0.20	-0.08	141	Amount	Independent	yes
14	USA	b	Falconiformes	<i>Circus cyaneus</i>	-0.27	0.39	17	Amount	Independent	yes
15	USA	b	Anseriformes	<i>Dendrocygna bicolor</i>	0.09	-0.23	5	Amount	Dependent	no
15	USA	b	Gruiformes	<i>Gallinula galeata</i>	0.41	-0.20	5	Amount	Dependent	no
15	USA	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.07	-0.10	5	Amount	Dependent	no
15	USA	b	Gruiformes	<i>Porphyrio martinica</i>	0.45	-0.25	5	Amount	Dependent	no
15	USA	b	Gruiformes	<i>Rallus elegans</i>	0.29	-0.25	5	Amount	Dependent	no
16	Canada	b	Passeriformes	<i>Cistothorus palustris</i>	0.53	-0.43	70	Amount	Dependent	yes
16	Canada	b	Passeriformes	<i>Geothlypis trichas</i>	0.14	0.18	70	Amount	Dependent	yes
16	Canada	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.24	-0.15	70	Amount	Dependent	yes
16	Canada	b	Passeriformes	<i>Melospiza georgiana</i>	0.24	0.15	70	Amount	Dependent	yes
16	Canada	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.30	-0.07	70	Amount	Dependent	yes
16	Canada	b	Gruiformes	<i>Rallus limicola</i>	0.29	0.04	70	Amount	Dependent	yes
16	Canada	b	Passeriformes	<i>Setophaga petechia</i>	0.07	-0.23	70	Amount	Dependent	yes

17	USA	b	Ciconiiformes	<i>Botaurus lentiginosus</i>	0.29	0.60	10	Amount	Independent	yes
17	USA	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.03	0.27	15	Amount	Independent	yes
17	USA	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.75	0.88	16	Amount	Independent	yes
17	USA	b	Gruiformes	<i>Porzana carolina</i>	0.18	0.36	16	Amount	Independent	yes
17	USA	b	Gruiformes	<i>Rallus limicola</i>	0.14	0.38	16	Amount	Independent	yes
18	Canada	b	Passeriformes	<i>Agelaius phoeniceus</i>	0.44	0.15	15	Amount	Independent	no
18	Canada	b	Passeriformes	<i>Cistothorus palustris</i>	0.61	-0.02	15	Amount	Independent	no
18	Canada	b	Gruiformes	<i>Gallinula galeata</i>	0.42	0.00	15	Amount	Independent	no
18	Canada	b	Passeriformes	<i>Geothlypis trichas</i>	-0.31	0.36	15	Amount	Independent	no
18	Canada	b	Ciconiiformes	<i>Ixobrychus exilis</i>	0.64	-0.06	15	Amount	Independent	no
18	Canada	b	Passeriformes	<i>Melospiza georgiana</i>	0.47	-0.24	15	Amount	Independent	no
18	Canada	b	Gruiformes	<i>Porzana carolina</i>	0.00	-0.09	15	Amount	Independent	no
18	Canada	b	Gruiformes	<i>Rallus limicola</i>	0.20	-0.28	15	Amount	Independent	no
19	USA	b	Gruiformes	<i>Gallinula galeata</i>	-0.05	0.17	109	Amount	Independent	yes
19	USA	b	Ciconiiformes	<i>Ixobrychus exilis</i>	-0.08	-0.13	62	Amount	Independent	yes
19	USA	b	Gruiformes	<i>Porphyrio martinica</i>	-0.06	0.14	109	Amount	Independent	yes
20	USA	b	Passeriformes	<i>Agelaius phoeniceus</i>	-0.35	-0.32	32	Amount	Dependent	yes
20	USA	b	Anseriformes	<i>Anas discors</i>	0.06	0.18	22	Amount	Dependent	yes
20	USA	b	Anseriformes	<i>Anas platyrhynchos</i>	0.08	-0.06	32	Amount	Dependent	yes
20	USA	b	Charadriiformes	<i>Chlidonias niger</i>	0.08	-0.29	22	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Cistothorus palustris</i>	0.42	-0.14	22	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Cistothorus platensis</i>	0.37	-0.10	22	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Empidonax alnorum</i>	0.12	-0.12	10	Amount	Dependent	yes
20	USA	b	Gruiformes	<i>Fulica americana</i>	-0.05	-0.11	22	Amount	Dependent	yes
20	USA	b	Charadriiformes	<i>Gallinago delicata</i>	0.20	0.14	10	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Geothlypis trichas</i>	0.65	0.11	32	Amount	Dependent	yes

20	USA	b	Gruiformes	<i>Grus canadensis</i>	0.56	0.65	10	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Melospiza georgiana</i>	0.49	0.14	32	Amount	Dependent	yes
20	USA	b	Podicipediformes	<i>Podilymbus podiceps</i>	0.09	-0.15	22	Amount	Dependent	yes
20	USA	b	Gruiformes	<i>Porzana carolina</i>	0.20	0.26	32	Amount	Dependent	yes
20	USA	b	Gruiformes	<i>Rallus limicola</i>	-0.25	-0.07	22	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Setophaga petechia</i>	0.36	0.20	32	Amount	Dependent	yes
20	USA	b	Passeriformes	<i>Xanthocephalus xanthocephalus</i>	0.04	-0.42	12	Amount	Dependent	yes
21	USA	r	Testudine	<i>Chrysemys picta marginata</i>	0.26	0.24	31	Configuration	Dependent	no
21	USA	r	Testudine	<i>Emydoidea blandingii</i>	-0.10	0.11	7	Configuration	Dependent	no
21	USA	r	Squamata	<i>Nerodia erythrogaster neglecta</i>	-0.14	0.17	12	Configuration	Dependent	no
21	USA	r	Squamata	<i>Nerodia sipedon sipedon</i>	0.21	0.28	21	Configuration	Dependent	no
22	Canada	r	Testudine	<i>Chrysemys picta marginata</i>	-0.19	0.53	20	Amount	Dependent	yes
23	USA	r	Testudine	<i>Chrysemys picta</i>	0.22	-0.43	35	Amount	Independent	no
24	USA	r	Squamata	<i>Nerodia erythrogaster neglecta</i>	0.12	-0.17	76	Amount	Independent	yes
24	USA	r	Squamata	<i>Nerodia rhombifer rhombifer</i>	0.26	-0.31	76	Amount	Independent	yes
24	USA	r	Squamata	<i>Nerodia sipedon pleuralis</i>	0.04	-0.19	76	Amount	Independent	yes
25	USA	r	Testudine	<i>Glyptemys muhlenbergii</i>	0.09	-0.14	43	Amount	Unknown	yes
16	Canada	r	Testudine	<i>Emydoidea blandingii</i>	-0.08	0.50	70	Amount	Independent	yes
16	Canada	r	Testudine	<i>Sternotherus odoratus</i>	-0.21	0.34	66	Amount	Independent	yes
26	USA	r	Testudine	<i>Apalone spinifera spinifera</i>	-0.09	0.11	30	Amount	Independent	yes
26	USA	r	Testudine	<i>Chelydra serpentina serpentina</i>	-0.13	-0.07	150	Amount	Independent	yes
26	USA	r	Testudine	<i>Chrysemys picta marginata</i>	0.16	-0.19	206	Amount	Independent	yes
26	USA	r	Testudine	<i>Trachemys scripta elegans</i>	0.21	-0.08	55	Amount	Independent	yes
27	USA	r	Testudine	<i>Chelydra serpentina</i>	0.04	-0.07	25	Amount	Independent	no
27	USA	r	Testudine	<i>Chrysemys picta</i>	0.18	0.00	25	Amount	Independent	no
28	USA	r	Testudine	<i>Chelydra serpentina</i>	-0.06	0.10	42	Amount	Independent	no

28	USA	r	Testudine	<i>Chrysemys picta</i>	-0.06	0.20	42	Amount	Independent	no
1	Canada	a	Anura	<i>Bufo boreas</i>	-0.07	0.37	9	Amount	Independent	no
1	Canada	a	Anura	<i>Pseudacris maculata</i>	-0.27	0.26	15	Amount	Independent	no
1	Canada	a	Anura	<i>Rana sylvatica</i>	-0.28	0.34	15	Amount	Independent	no
29	Switzerland	a	Anura	<i>Bufo bufo</i>	-0.30	0.13	34	Configuration	Dependent	yes
29	Switzerland	a	Anura	<i>Hyla arborea</i>	0.27	-0.23	46	Configuration	Dependent	yes
29	Switzerland	a	Caudata	<i>Ichthyosaura alpestris</i>	0.14	-0.11	49	Configuration	Dependent	yes
29	Switzerland	a	Caudata	<i>Lissotriton vulgaris</i>	0.27	-0.40	66	Configuration	Dependent	yes
29	Switzerland	a	Anura	<i>Rana esculenta</i>	0.28	-0.35	52	Configuration	Dependent	yes
29	Switzerland	a	Anura	<i>Rana lessonae</i>	0.30	-0.34	52	Configuration	Dependent	yes
29	Switzerland	a	Anura	<i>Rana temporaria</i>	-0.15	0.43	44	Configuration	Dependent	yes
29	Switzerland	a	Caudata	<i>Triturus cristatus</i>	0.11	-0.26	38	Configuration	Dependent	yes
30	USA	a	Caudata	<i>Ambystoma maculatum</i>	0.11	0.21	45	Amount	Dependent	yes
30	USA	a	Caudata	<i>Notophthalmus viridescens</i>	0.19	-0.36	67	Amount	Dependent	yes
30	USA	a	Anura	<i>Pseudacris crucifer</i>	0.21	-0.24	39	Amount	Dependent	yes
30	USA	a	Anura	<i>Rana catesbeiana</i>	0.36	-0.10	29	Amount	Dependent	yes
30	USA	a	Anura	<i>Rana clamitans</i>	0.27	-0.27	39	Amount	Dependent	yes
30	USA	a	Anura	<i>Rana palustris</i>	0.21	-0.26	29	Amount	Dependent	yes
30	USA	a	Anura	<i>Rana sylvatica</i>	0.05	0.23	45	Amount	Dependent	yes
31	USA	a	Anura	<i>Pseudacris regilla</i>	0.06	-0.01	32	Configuration	Independent	no
31	USA	a	Anura	<i>Rana catesbeiana</i>	0.31	0.10	28	Configuration	Dependent	no
31	USA	a	Anura	<i>Rana draytonii</i>	0.01	0.18	22	Configuration	Dependent	no
32	Belgium	a	Caudata	<i>Ichthyosaura alpestris</i>	-0.01	0.12	254	Configuration	Dependent	yes
32	Belgium	a	Caudata	<i>Lissotriton vulgaris</i>	0.08	-0.10	254	Configuration	Dependent	yes
33	Canada	a	Anura	<i>Bufo americanus</i>	-0.12	-0.25	36	Amount	Independent	no
33	Canada	a	Anura	<i>Hyla versicolor</i>	0.22	0.17	36	Amount	Independent	no

33	Canada	a	Anura	<i>Pseudacris crucifer</i>	0.33	0.60	36	Amount	Independent	no
33	Canada	a	Anura	<i>Rana clamitans</i>	0.14	0.38	36	Amount	Dependent	no
33	Canada	a	Anura	<i>Rana pipiens</i>	0.06	-0.36	36	Amount	Independent	no
33	Canada	a	Anura	<i>Rana sylvatica</i>	0.05	0.13	36	Amount	Independent	no
34	USA	a	Caudata	<i>Ambystoma macrodactylum</i>	-0.06	0.26	105	Amount	Independent	no
34	USA	a	Anura	<i>Pseudacris regilla</i>	-0.19	-0.12	99	Amount	Independent	no
34	USA	a	Anura	<i>Rana luteiventris</i>	-0.16	0.06	95	Amount	Independent	no
35	USA	a	Caudata	<i>Ambystoma laterale</i>	0.04	0.37	89	Amount	Dependent	yes
35	USA	a	Caudata	<i>Ambystoma maculatum</i>	-0.08	0.42	89	Amount	Dependent	yes
35	USA	a	Anura	<i>Bufo americanus</i>	0.22	-0.27	89	Amount	Independent	yes
35	USA	a	Caudata	<i>Notophthalmus viridescens</i>	-0.11	0.25	89	Amount	Dependent	yes
35	USA	a	Anura	<i>Rana clamitans</i>	0.11	0.21	89	Amount	Independent	yes
35	USA	a	Anura	<i>Rana pipiens</i>	0.18	-0.30	89	Amount	Independent	yes
35	USA	a	Anura	<i>Rana septentrionalis</i>	0.17	0.18	89	Amount	Independent	yes
35	USA	a	Anura	<i>Rana sylvatica</i>	-0.13	0.26	89	Amount	Independent	yes
36	Sweden	a	Caudata	<i>Triturus cristatus</i>	0.11	0.01	134	Amount	Independent	no
37	Australia	a	Anura	<i>Crinia signifera</i>	0.11	0.67	44	Amount	Independent	yes
37	Australia	a	Anura	<i>Limnodynastes dumerilii</i>	0.13	0.76	36	Amount	Independent	yes
37	Australia	a	Anura	<i>Limnodynastes peronii</i>	0.29	-0.10	36	Amount	Independent	yes
37	Australia	a	Anura	<i>Litoria ewingii/verreauxii</i>	-0.02	0.27	36	Amount	Independent	yes
37	Australia	a	Anura	<i>Paracrinia haswelli</i>	0.10	0.45	44	Amount	Independent	yes
38	Romania	a	Anura	<i>Bombina variegata</i>	0.11	-0.11	54	Amount	Dependent	no
38	Romania	a	Anura	<i>Bufo bufo</i>	0.16	0.24	54	Amount	Dependent	no
38	Romania	a	Anura	<i>Hyla arborea</i>	-0.05	0.07	54	Amount	Dependent	no
38	Romania	a	Caudata	<i>Lissotriton vulgaris</i>	-0.22	0.10	54	Amount	Dependent	no
38	Romania	a	Anura	<i>Pelobates fuscus</i>	-0.04	-0.09	54	Amount	Dependent	no

38	Romania	a	Anura	<i>Rana dalmatina</i>	-0.12	0.04	54	Amount	Dependent	no
38	Romania	a	Anura	<i>Rana esculenta</i>	0.12	-0.14	54	Amount	Dependent	no
38	Romania	a	Anura	<i>Rana temporaria</i>	-0.27	0.49	54	Amount	Dependent	no
38	Romania	a	Caudata	<i>Triturus cristatus</i>	-0.24	0.14	54	Amount	Dependent	no
39	Australia	a	Anura	<i>Litoria raniformis</i>	0.54	0.10	107	Configuration	Dependent	no
40	Canada	a	Anura	<i>Bufo americanus</i>	0.24	-0.20	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Hyla versicolor</i>	-0.09	0.16	34	Configuration	Dependent	no
40	Canada	a	Caudata	<i>Notophthalmus viridescens</i>	-0.10	0.17	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Pseudacris crucifer</i>	-0.21	0.48	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Pseudacris triseriata</i>	-0.06	-0.12	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Rana clamitans</i>	0.02	0.10	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Rana pipiens</i>	-0.17	0.24	34	Configuration	Dependent	no
40	Canada	a	Anura	<i>Rana sylvatica</i>	-0.13	0.32	34	Configuration	Dependent	no
41	USA	a	Caudata	<i>Ambystoma maculatum</i>	0.07	0.44	62	Amount	Unknown	no
41	USA	a	Anura	<i>Rana sylvatica</i>	-0.11	0.38	62	Amount	Unknown	no
42	Canada	a	Caudata	<i>Ambystoma laterale</i>	0.11	0.13	46	Amount	Dependent	no
42	Canada	a	Caudata	<i>Ambystoma maculatum</i>	0.32	0.32	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Bufo americanus</i>	0.09	-0.13	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Hyla versicolor</i>	0.35	0.40	46	Amount	Dependent	no
42	Canada	a	Caudata	<i>Notophthalmus viridescens</i>	0.16	0.16	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Pseudacris crucifer</i>	0.12	0.35	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Pseudacris triseriata</i>	0.25	0.18	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Rana catesbeiana</i>	0.16	-0.43	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Rana clamitans</i>	0.13	0.23	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Rana pipiens</i>	0.19	-0.25	46	Amount	Dependent	no
42	Canada	a	Anura	<i>Rana septentrionalis</i>	0.50	0.42	46	Amount	Dependent	no

42	Canada	a	Anura	<i>Rana sylvatica</i>	0.26	0.50	46	Amount	Dependent	no
43	Canada	a	Caudata	<i>Ambystoma laterale/A. maculatum</i>	-0.28	0.26	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Bufo americanus</i>	0.18	-0.18	34	Amount	Independent	no
43	Canada	a	Caudata	<i>Notophthalmus viridescens</i>	0.47	0.18	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Pseudacris crucifer</i>	-0.23	0.51	34	Amount	Independent	no
43	Canada	a	Anura	<i>Rana catesbeiana</i>	-0.39	0.33	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Rana clamitans</i>	0.23	-0.18	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Rana palustris</i>	0.33	0.27	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Rana septentrionalis</i>	0.46	-0.25	34	Amount	Dependent	no
43	Canada	a	Anura	<i>Rana sylvatica</i>	-0.25	0.44	34	Amount	Dependent	no
44	USA	a	Anura	<i>Rana pipiens</i>	-0.06	-0.16	31	Amount	Independent	no
45	France	a	Caudata	<i>Ichthyosaura alpestris</i>	0.13	0.50	79	Configuration	Independent	yes
45	France	a	Caudata	<i>Lissotriton helveticus</i>	0.24	0.31	79	Configuration	Independent	yes
45	France	a	Caudata	<i>Triturus cristatus</i>	0.30	0.16	79	Configuration	Independent	yes
46	USA	a	Caudata	<i>Ambystoma tigrinum</i>	-0.24	-0.30	39	Amount	Independent	yes
46	USA	a	Anura	<i>Bufo americanus</i>	-0.13	0.25	27	Amount	Independent	yes
46	USA	a	Anura	<i>Hyla versicolor</i>	-0.31	0.15	35	Amount	Independent	yes
46	USA	a	Anura	<i>Pseudacris crucifer</i>	-0.18	-0.18	37	Amount	Independent	yes
46	USA	a	Anura	<i>Pseudacris triseriata</i>	-0.16	-0.17	39	Amount	Independent	yes
46	USA	a	Anura	<i>Rana clamitans</i>	0.08	0.12	39	Amount	Independent	yes
46	USA	a	Anura	<i>Rana palustris</i>	-0.15	-0.27	22	Amount	Independent	yes
46	USA	a	Anura	<i>Rana pipiens</i>	0.19	-0.06	22	Amount	Independent	yes
47	USA	a	Caudata	<i>Ambystoma tigrinum</i>	-0.42	-0.31	26	Amount	Independent	yes
47	USA	a	Anura	<i>Bufo americanus</i>	-0.26	-0.29	26	Amount	Independent	yes
47	USA	a	Anura	<i>Bufo cognatus</i>	-0.38	-0.19	26	Amount	Independent	yes
47	USA	a	Anura	<i>Hyla versicolor/chrysosecelis</i>	0.26	0.73	21	Amount	Independent	yes

47	USA	a	Anura	<i>Pseudacris triseriata</i>	0.51	0.18	26	Amount	Independent	yes
47	USA	a	Anura	<i>Rana pipiens</i>	0.21	0.08	26	Amount	Independent	yes
47	USA	a	Anura	<i>Rana sylvatica</i>	0.29	0.62	26	Amount	Independent	yes
48	Canada	a	Anura	<i>Rana clamitans</i>	0.13	-0.20	21	Amount	Dependent	yes
49	USA	a	Anura	<i>Bufo fowleri</i>	-0.06	0.18	22	Amount	Independent	no
49	USA	a	Anura	<i>Hyla versicolor</i>	-0.08	0.27	22	Amount	Independent	no
49	USA	a	Anura	<i>Pseudacris crucifer</i>	0.33	0.50	22	Amount	Independent	no
49	USA	a	Anura	<i>Rana catesbeiana</i>	0.14	0.30	22	Amount	Independent	no
50	USA	a	Anura	<i>Rana virgatipes</i>	0.33	0.69	38	Amount	Independent	no
51	Switzerland	a	Anura	<i>Hyla arborea</i>	0.20	-0.15	68	Configuration	Independent	no
52	USA	a	Anura	<i>Acris crepitans</i>	0.13	0.14	61	Amount	Independent	no
52	USA	a	Anura	<i>Bufo americanus</i>	-0.11	-0.06	61	Amount	Independent	no
52	USA	a	Anura	<i>Hyla chrysoscelis</i>	0.05	-0.09	61	Amount	Independent	no
52	USA	a	Anura	<i>Hyla versicolor</i>	0.12	0.27	61	Amount	Independent	no
52	USA	a	Anura	<i>Pseudacris triseriata</i>	0.06	-0.14	61	Amount	Independent	no
52	USA	a	Anura	<i>Rana catesbeiana</i>	-0.07	-0.13	61	Amount	Independent	no
52	USA	a	Anura	<i>Rana pipiens</i>	-0.05	0.19	61	Amount	Independent	no
53	Romania	a	Anura	<i>Bombina variegata</i>	-0.02	0.20	51	Amount	Dependent	yes
53	Romania	a	Anura	<i>Bufo bufo</i>	-0.07	-0.15	26	Amount	Dependent	yes
53	Romania	a	Anura	<i>Hyla arborea</i>	0.18	-0.01	45	Amount	Dependent	yes
53	Romania	a	Caudata	<i>Lissotriton vulgaris</i>	0.54	0.32	53	Amount	Dependent	yes
53	Romania	a	Anura	<i>Rana dalmatina</i>	0.22	-0.04	51	Amount	Dependent	yes
53	Romania	a	Anura	<i>Rana esculenta</i>	0.21	-0.18	51	Amount	Dependent	yes
53	Romania	a	Anura	<i>Rana temporaria</i>	0.13	0.37	44	Amount	Dependent	yes
53	Romania	a	Caudata	<i>Salamandra salamandra</i>	0.56	0.27	53	Amount	Dependent	yes
53	Romania	a	Caudata	<i>Triturus cristatus</i>	0.35	0.14	52	Amount	Dependent	yes

54	Switzerland	a	Anura	<i>Bufo bufo</i>	-0.32	0.52	61	Configuration	Unknown	yes
54	Switzerland	a	Anura	<i>Bufo calamita</i>	0.24	0.22	61	Configuration	Unknown	yes
54	Switzerland	a	Caudata	<i>Ichthyosaura alpestris</i> <i>Ambystoma maculatum/A.</i> <i>jeffersonianum</i>	0.14	0.38	91	Configuration	Unknown	yes
55	USA	a	Caudata	<i>Bufo americanus/B. woodhouseii</i>	0.44	0.54	34	Configuration	Dependent	no
55	USA	a	Anura	<i>fowleri</i>	-0.07	0.13	27	Configuration	Dependent	no
55	USA	a	Anura	<i>Hyla versicolor</i>	-0.14	0.00	27	Configuration	Dependent	no
55	USA	a	Caudata	<i>Notophthalmus viridescens</i>	-0.01	0.40	34	Configuration	Dependent	no
55	USA	a	Anura	<i>Pseudacris crucifer</i>	0.00	0.12	34	Configuration	Dependent	no
55	USA	a	Anura	<i>Rana catesbeiana</i>	0.05	-0.18	27	Configuration	Dependent	no
55	USA	a	Anura	<i>Rana clamitans</i>	-0.13	0.04	34	Configuration	Dependent	no
55	USA	a	Anura	<i>Rana palustris</i>	-0.38	-0.10	27	Configuration	Dependent	no
55	USA	a	Anura	<i>Rana sylvatica</i>	0.37	0.55	34	Configuration	Dependent	no
56	USA	a	Anura	<i>Rana pipiens</i>	0.22	-0.09	73	Amount	Independent	no
57	USA	a	Anura	<i>Acris crepitans</i>	0.12	0.10	49	Configuration	Dependent	no
57	USA	a	Caudata	<i>Ambystoma texanum</i>	0.17	-0.10	49	Configuration	Dependent	no
57	USA	a	Anura	<i>Bufo americanus</i>	-0.34	-0.22	40	Configuration	Dependent	no
57	USA	a	Anura	<i>Hyla versicolor/H. chrysoscelis</i>	0.23	0.43	45	Configuration	Dependent	no
57	USA	a	Anura	<i>Pseudacris crucifer</i>	0.02	0.37	45	Configuration	Dependent	no
57	USA	a	Anura	<i>Pseudacris maculata</i>	-0.09	-0.23	45	Configuration	Dependent	no
57	USA	a	Anura	<i>Rana blairi/R. sphenoccephalus</i>	0.02	-0.05	38	Configuration	Dependent	no
57	USA	a	Anura	<i>Rana catesbeiana</i>	0.27	0.15	44	Configuration	Dependent	no
57	USA	a	Anura	<i>Rana clamitans</i>	0.03	-0.05	48	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Dendropsophus elianae</i>	-0.14	0.20	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Dendropsophus minutus</i>	-0.25	0.29	6	Configuration	Dependent	no

58	Brazil	a	Anura	<i>Dendropsophus nanus</i>	-0.06	0.42	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Dermatonotus muelleri</i>	-0.24	-0.12	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Elachistocleis bicolor</i>	0.14	0.21	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Eupemphix nattereri</i>	0.15	-0.43	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Hypsiboas albopunctatus</i>	0.00	0.41	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Hypsiboas raniceps</i>	0.23	0.43	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus chaquensis</i>	-0.10	0.32	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus fuscus</i>	-0.27	0.58	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus labyrinthicus</i>	-0.08	0.31	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus latrans</i>	0.27	0.22	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus mystacinus</i>	-0.67	0.25	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Leptodactylus podicipinus</i>	-0.18	0.49	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Physalaemus centralis</i>	-0.16	0.29	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Physalaemus cuvieri</i>	-0.21	0.33	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Pseudis platensis</i>	-0.14	0.23	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Pseudopaludicola falcipes</i>	0.29	0.29	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Rhinella schneideri</i>	0.11	0.38	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Scinax fuscomarginatus</i>	0.21	0.36	6	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Scinax fuscovarius</i>	-0.59	0.23	17	Configuration	Dependent	no
58	Brazil	a	Anura	<i>Scinax similis</i>	-0.21	0.16	6	Configuration	Dependent	no
59	USA	a	Anura	<i>Acris crepitans</i>	0.47	0.82	19	Amount	Dependent	no
59	USA	a	Anura	<i>Bufo americanus</i>	-0.26	0.65	17	Amount	Dependent	no
59	USA	a	Anura	<i>Bufo fowleri</i>	-0.25	-0.44	19	Amount	Dependent	no
59	USA	a	Anura	<i>Hyla versicolor/H. chrysoscelis</i>	0.37	0.56	17	Amount	Dependent	no
59	USA	a	Caudata	<i>Notophthalmus viridescens</i>	0.14	0.46	17	Amount	Dependent	no
59	USA	a	Anura	<i>Pseudacris crucifer</i>	-0.28	0.36	17	Amount	Dependent	no

59	USA	a	Anura	<i>Rana catesbeiana</i>	-0.20	0.38	17	Amount	Dependent	no
59	USA	a	Anura	<i>Rana clamitans</i>	0.11	0.28	17	Amount	Dependent	no
59	USA	a	Anura	<i>Rana palustris</i>	0.20	0.46	12	Amount	Dependent	no
59	USA	a	Anura	<i>Rana sylvatica</i>	0.57	0.67	18	Amount	Dependent	no
59	USA	a	Anura	<i>Rana utricularia</i>	0.35	0.51	12	Amount	Dependent	no
60	USA	a	Caudata	<i>Ambystoma maculatum</i>	0.05	0.27	55	Amount	Dependent	yes
60	USA	a	Anura	<i>Rana sylvatica</i>	0.05	0.31	55	Amount	Dependent	yes
61	Australia	a	Anura	<i>Crinia parinsignifera</i>	-0.05	0.36	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Crinia signifera</i>	-0.12	0.47	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Limnodynastes dumerilii</i>	-0.22	0.49	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Limnodynastes peronii</i>	-0.13	-0.26	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Limnodynastes tasmaniensis</i>	-0.28	0.10	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Litoria ewingii/L. paraewingi</i>	0.29	0.04	30	Amount	Independent	yes
61	Australia	a	Anura	<i>Litoria peronii</i>	-0.10	0.32	30	Amount	Independent	yes
62	USA	a	Caudata	<i>Ambystoma maculatum</i>	0.19	0.13	49	Amount	Dependent	no
62	USA	a	Anura	<i>Rana sylvatica</i>	-0.10	0.08	49	Amount	Dependent	no
63	Switzerland	a	Anura	<i>Bombina variegata</i>	-0.07	-0.09	480	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Bufo bufo</i>	0.07	0.10	107	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Bufo calamita</i>	-0.11	-0.23	247	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Hyla arborea</i>	0.22	-0.12	315	Amount	Unknown	yes
63	Switzerland	a	Caudata	<i>Ichthyosaura alpestris</i>	-0.07	0.15	414	Amount	Unknown	yes
63	Switzerland	a	Caudata	<i>Lissotriton helveticus</i>	-0.03	-0.12	469	Amount	Unknown	yes
63	Switzerland	a	Caudata	<i>Lissotriton vulgaris</i>	0.12	0.13	615	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Rana dalmatina</i>	0.14	0.20	520	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Rana ridibunda</i>	0.06	-0.39	423	Amount	Unknown	yes
63	Switzerland	a	Anura	<i>Rana temporaria</i>	0.10	0.15	296	Amount	Unknown	yes

*Studies cited below. Studies conducted in the same study area by the same researchers are identified as one study in the meta-analysis

[†]Taxa: m = mammal, b = bird, r = reptile, a = amphibian

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Appendix L: Distribution of correlations between wetland amount and forest amount from each study included in Chapter 4 across and within taxonomic groups.

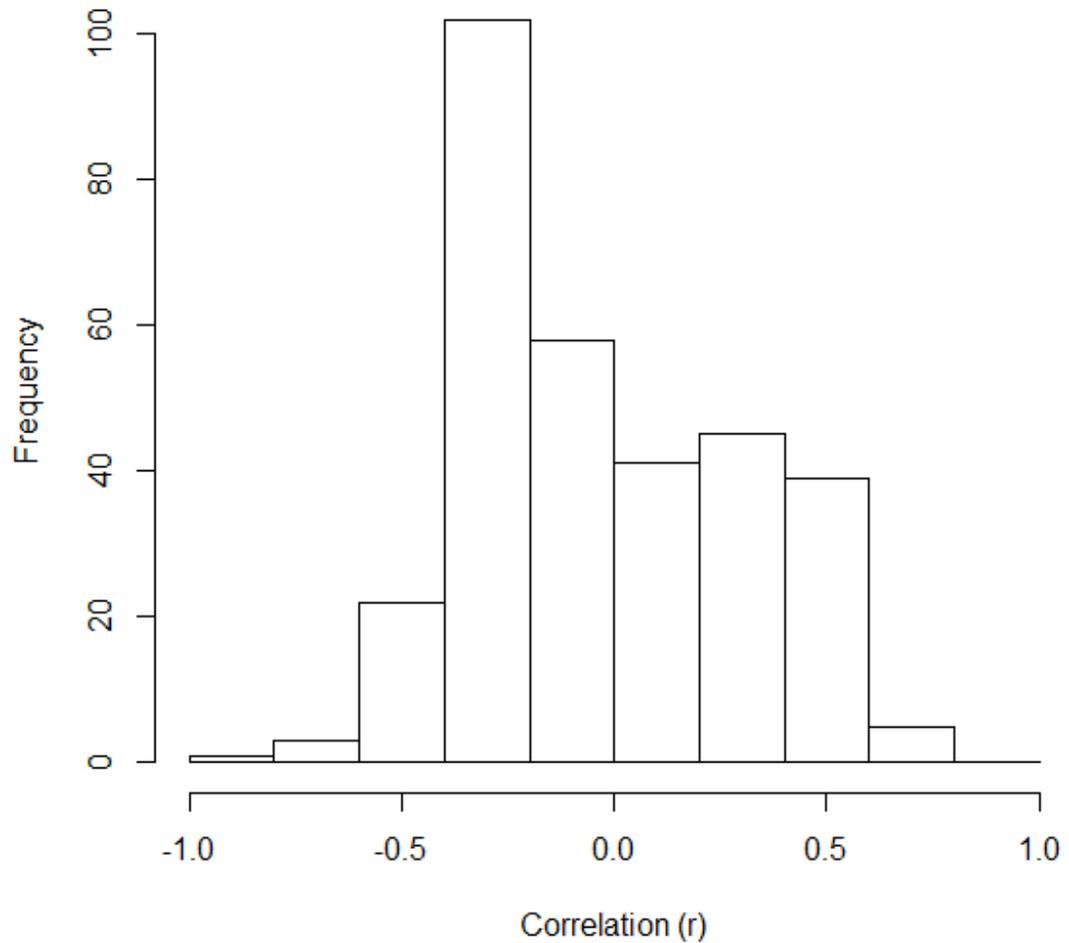


Figure L1: Frequency distribution of Pearson correlation coefficients (r), where each r is the correlation between wetland amount and forest amount in a landscape from each study across all wetland-dependent taxa.

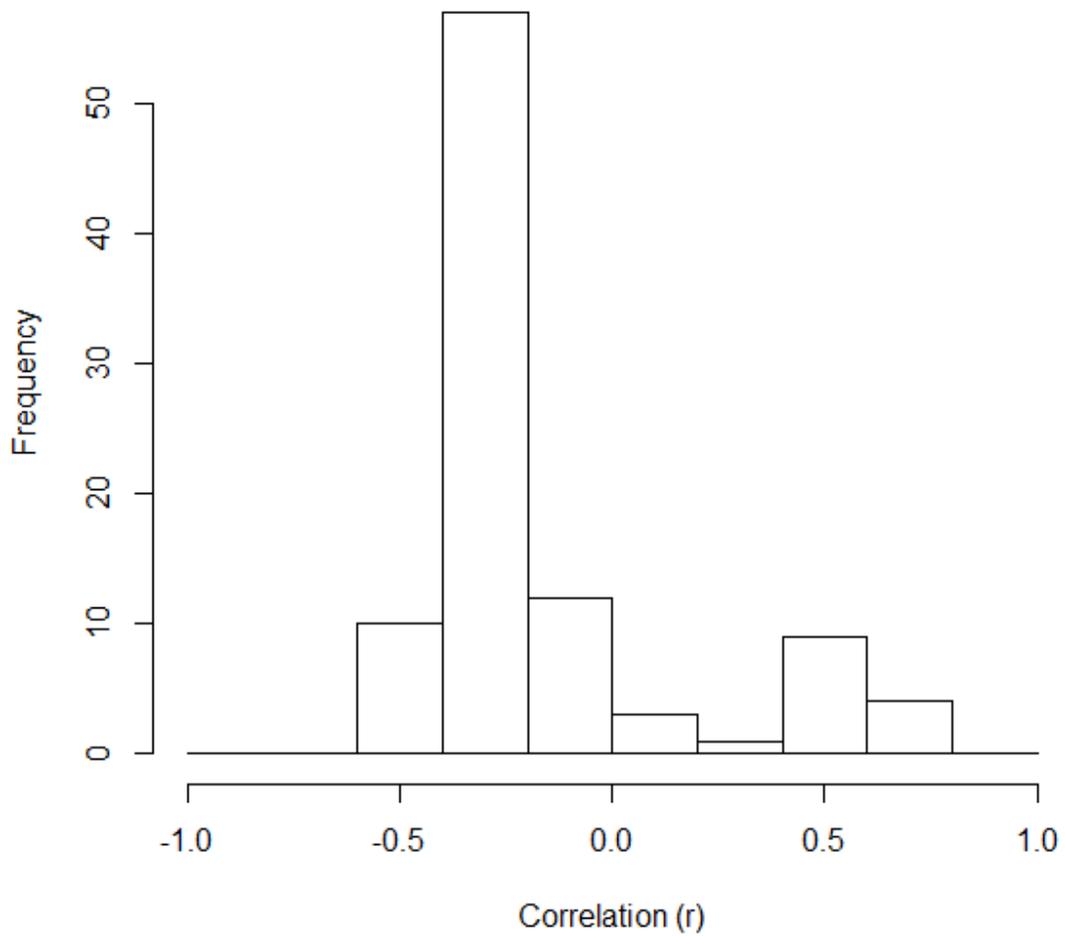


Figure L2: Frequency distribution of Pearson correlation coefficients (r), where each r is the correlation between wetland amount and forest amount in a landscape from each study conducted on wetland-dependent birds.

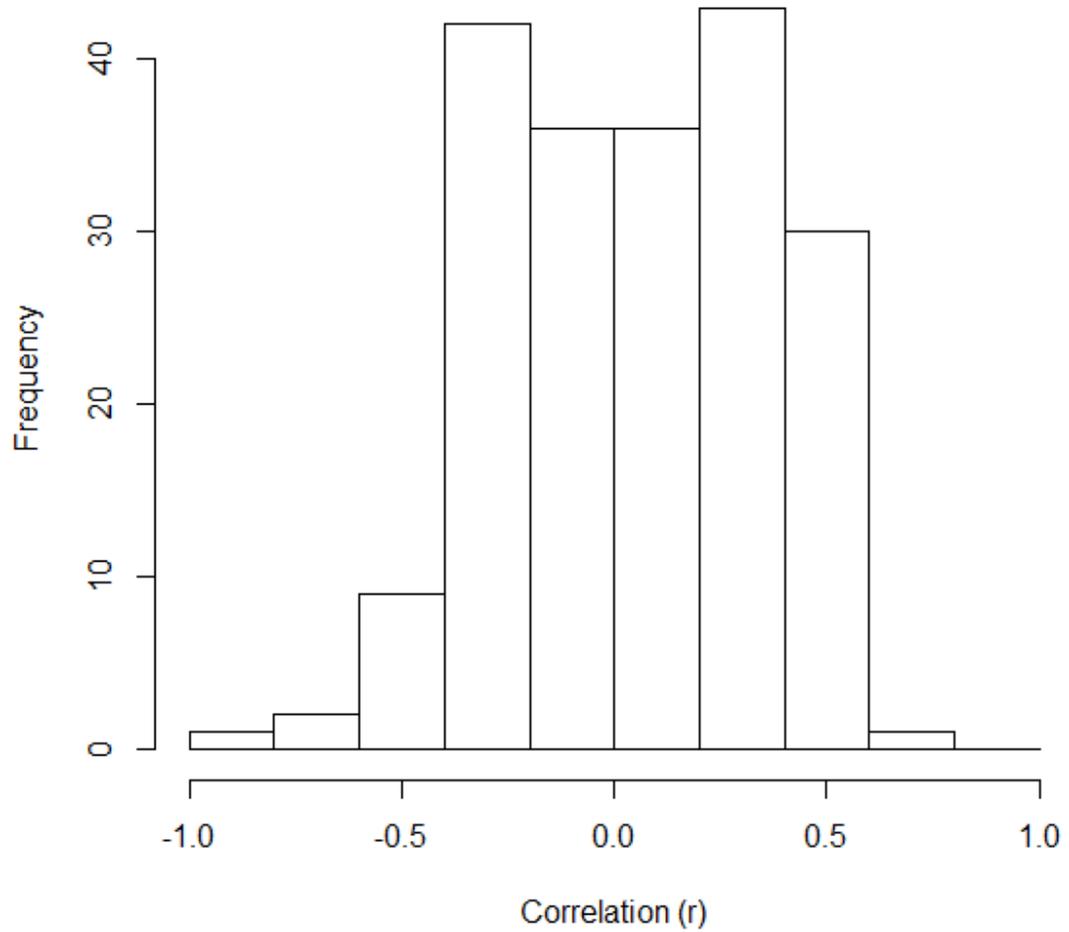


Figure L3: Frequency distribution of Pearson correlation coefficients (r), where each r is the correlation between wetland amount and forest amount in a landscape from each study conducted on wetland-dependent amphibians.

Appendix M: Results from univariate mixed-effects meta-analytic models testing the effect of study design moderators and taxonomic class on wetland species responses to wetland amount and forest amount at the landscape scale (Chapter 4).

Moderator variable	ES _{Zr}	ES _r	CI _{lower}	CI _{upper}	Q _E	Q _M	p (Q _M)	n
Study Type ¹					653.691	0.657	0.418	330
<i>Amount-based</i>	0.109	0.108	0.080	0.138				255
<i>Configuration-based</i>	0.082	0.082	0.025	0.140				75
Sampling Effort					630.186	5.849	0.055	330
<i>Dependent</i>	0.132	0.132	0.096	0.168				200
<i>Independent</i>	0.082	0.081	0.041	0.122				113
<i>Unknown</i>	0.040	0.040	-0.042	0.123				17
Sampled Wetland Area					659.471	2.832	0.092	330
<i>Included</i>	0.122	0.121	0.037	0.117				178
<i>Not Included</i>	0.077	0.077	0.088	0.156				152
Taxon - Wetland Amount					558.3011	30.5681	< 0.0001	330
<i>Mammal</i>	0.306	0.297	0.136	0.476				7
<i>Bird</i>	0.215	0.211	0.1629	0.2665				98
<i>Reptile</i>	0.048	0.047	-0.045	0.140				20
<i>Amphibian</i>	0.066	0.066	0.366	0.0961				205
Taxon - Forest Amount					1047.358	18.7129	0.0003	330
<i>Mammal</i>	-0.102	-0.102	-0.327	0.123				7
<i>Bird</i>	-0.027	-0.027	-0.0949	0.0406				98

<i>Reptile</i>	0.028	0.028	-0.096	0.152	20
<i>Amphibian</i>	0.129	0.129	0.0895	0.169	205
