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RELATIONSHIPS BETWEEN INTENSIVE BIOMASS PRODUCTION AND BIODIVERSITY IN NORTH AMERICAN FORESTS – A LITERATURE REVIEW

TECHNICAL BULLETIN NO. 992

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PRESIDENT'S NOTE

Due to increasing cost of fossil fuels and concerns about carbon emissions, there is growing interest in deriving energy from cellulose materials, including forest biomass. This has led to concerns within the environmental and forestry communities about sustainability of biomass harvesting practices and implications for biological diversity. As a result, states and other entities, including sustainable forestry certification programs, are developing biomass harvesting guidelines that address a range of issues, including measures to protect biological diversity. Unfortunately, knowledge about effects of intensive biomass production systems on wildlife and plant communities and the technical basis for biomass harvesting guidelines are limited.

As an initial step in addressing this information gap, NCASI recently supported a review of literature related to biodiversity response to biomass production options in North American forests. The authors conducted a literature review and meta-analysis of manipulative and observational studies. They found that most taxa responded positively to thinning treatments but that reducing coarse woody debris will likely decrease bird diversity. However, other taxa may not respond strongly to reductions in coarse woody debris and overall biodiversity responses may be minimal. Short-rotation woody crops may have a lower diversity of birds and mammals than managed forests, but there is considerable uncertainty. The authors found no studies of biodiversity response to intercropping native warm season grasses in commercial forests. The authors note that additional research of added or increased frequency of harvesting operations, and research at larger spatial scales, would strengthen understanding of biodiversity responses and the technical basis for harvesting guidelines.

This technical bulletin represents the final report for a project supported through the NCASI Sustainable Forestry and Eastern Wildlife Program and the NCASI Western Wildlife Program. The details contained in this report will be useful to those seeking to understand the state of knowledge about potential response of biodiversity to intensive biomass production systems and will serve as a guide for future research priorities.

Ronald A. Yeske

October 2011

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NOTE DU PRÉSIDENT

En raison du coût croissant des combustibles fossiles et des préoccupations liées aux émissions de carbone, la production d'énergie à partir de matériaux cellulosiques, notamment à partir de la biomasse forestière, connaît un intérêt accru, ce qui a amené les communautés environnementales et forestières à s'inquiéter de la durabilité des pratiques de récolte de la biomasse et de l'impact de ces pratiques sur la diversité biologique. C'est pourquoi des instances gouvernementales et d'autres entités, notamment les organismes responsables des programmes de certification sur l'aménagement forestier durable, élaborent présentement des lignes directrices sur la récolte de la biomasse qui ont pour but de répondre à une série de préoccupations, dont l'instauration de mesures pour protéger la diversité biologique. Malheureusement, les connaissances actuelles sur l'impact des systèmes de récolte intensive de la biomasse sur la faune et la flore et les bases techniques sur lesquelles reposent les lignes directrices sont limitées.

Comme première étape pour combler ce manque d'informations, NCASI a récemment soutenu une revue de la littérature sur la réponse de la biodiversité aux méthodes de récolte de la biomasse dans les forêts de l'Amérique du Nord. Les auteurs ont passé la littérature en revue et ont effectué une méta-analyse d'études contrôlées et d'études par observation. Ils ont constaté que la plupart des taxons répondaient positivement aux traitements d'éclaircie, mais que la diversité des oiseaux diminuerait probablement s'il y avait moins de débris grossiers ligneux. Par contre, les autres taxons ne répondraient peut-être pas aussi fortement à une réduction des débris grossiers ligneux et la réponse globale de la biodiversité pourrait être minimale. Il est possible que les peuplements forestiers à courte rotation présentent une moins grande diversité d'oiseaux et de mammifères que les forêts aménagées, mais il existe encore beaucoup d'incertitude à ce sujet. Les auteurs n'ont pas recensé d'études sur la réponse de la biodiversité à la culture intercalaire de graminées indigènes de saison chaude dans les forêts commerciales. Les auteurs font remarquer qu'en réalisant des études sur l'impact qu'ont des récoltes plus fréquentes et des études à des échelles spatiales plus vastes, on aurait une meilleure compréhension de la réponse de la biodiversité et on renforcerait les bases techniques sur lesquelles reposent les lignes directrices.

Le présent Bulletin technique constitue le dernier volet d'un projet soutenu par le programme de NCASI sur la foresterie durable et la faune de l'est des États-Unis et du programme de NCASI sur la faune de l'ouest des États-Unis. Les informations dans ce rapport seront utiles à ceux et celles qui cherchent à mieux comprendre l'état des connaissances sur la réponse possible de la biodiversité aux systèmes de récolte intensive de biomasse et serviront de guide pour déterminer les priorités futures en matière de recherche.

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Octobre 2011

RELATIONSHIPS BETWEEN INTENSIVE BIOMASS PRODUCTION AND BIODIVERSITY IN NORTH AMERICAN FORESTS – A LITERATURE REVIEW

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ABSTRACT

Demand for alternative energy sources has led to increased interest in intensive biomass production. Biomass feedstocks may be produced in forests through a variety of practices such as thinning and fuels treatments, use of harvest residues including fine (foliage, small limbs and trees) and coarse woody debris (snags and downed logs), establishment and harvesting of short-rotation woody crops, and harvesting of natural biomass or intercropped herbaceous plant species between crop tree rows in intensively managed stands. When applied across a broad spatial extent, intensive biomass production in forests, which support a large proportion of biodiversity, has the potential to alter species composition, nutrient cycling, and overall biodiversity. However, potential effects of biomass harvesting on biodiversity are not well known. Therefore, to examine potential impacts and assess information gaps, we used a literature review and meta-analyses of manipulative and observational studies to assess potential biodiversity responses to practices associated with intensive biomass production systems in North American forests. We calculated 1,507 effect sizes from 68 studies. Biodiversity responses varied among taxa and production systems reviewed. Most taxa responded positively to thinning treatments. Reducing coarse woody debris will likely decrease bird diversity, but other taxa may not respond strongly. If reductions in coarse woody debris from actual harvests are less than the 70 – 95% used in experimental studies, then overall biodiversity responses may be minimal. Short-rotation woody crops may have lower diversity of birds and mammals than managed forests, but there is considerable uncertainty. We found no studies of biodiversity response to intercropping of native, warm season grasses in commercial forests. A small number of studies or strong geographic bias in available studies for some practices increased uncertainty about consistency of observed responses across different landscapes of North America. Additional research at larger spatial scales and of added or increased frequency of harvesting operations would strengthen understanding of biodiversity response and the technical basis for biomass harvesting/establishment guidelines.

KEYWORDS

amphibians, bioenergy, biomass, biological diversity, birds, coarse woody debris, forest management, fuels treatments, intensive biomass production, *Panicum virgatum,* reptiles, short-rotation woody crops, small mammals, switchgrass, thinning

RELATED NCASI PUBLICATIONS

None.

RELATIONS ENTRE LA RÉCOLTE INTENSIVE DE BIOMASSE ET LA BIODIVERSITÉ DANS LES FORÊTS NORD-AMÉRICAINES – UNE REVUE DE LA LITTÉRATURE

BULLETIN TECHNIQUE N^{O} 992
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RÉSUMÉ

Il existe un intérêt de plus en plus grand pour la récolte intensive de biomasse en raison d'une demande accrue pour d'autres sources d'énergie. Il est possible de récolter la biomasse en forêt en faisant appel à diverses pratiques comme, par exemple, les éclaircies et les mesures de réduction du danger de feu, l'utilisation de résidus provenant de la récolte du bois dont les fines (feuillage, petites branches et petits arbres) et les débris grossiers ligneux (chicots et arbres morts), la culture et la récolte de peuplements à courte rotation, et la récolte de la biomasse naturelle ou d'espèces de plantes herbacées cultivées entre les rangées de l'arbre du peuplement final dans des forêts aménagées de façon intensive. Lorsqu'elle se fait à une large échelle spatiale, la récolte intensive de biomasse en forêt, milieu qui abrite une grande partie de la biodiversité, peut potentiellement modifier la composition taxinomique, le cycle des éléments nutritifs et l'ensemble de la biodiversité. Par contre, on connait peu l'impact que pourrait avoir la récolte de la biomasse sur la biodiversité. Pour être en mesure d'évaluer l'impact potentiel et de définir les lacunes en matière d'information, nous avons donc effectué une revue de la littérature et des méta-analyses d'études contrôlées et d'études par observation pour déterminer la réponse possible de la biodiversité aux pratiques associées aux systèmes de récolte intensive de la biomasse dans les forêts de l'Amérique du Nord. Nous avons calculé 1 507 valeurs d'effet qui ont été tirées de 68 études. La réponse de la biodiversité a varié selon les taxons et les systèmes de récolte. La plupart des taxons ont répondu positivement aux traitements d'éclaircie. La diversité des oiseaux diminuerait probablement s'il y avait moins de débris grossiers ligneux, mais les autres taxons ne répondraient peut-être pas aussi fortement à cette réduction. Si les récoltes actuelles retirent moins de débris grossiers ligneux que la quantité utilisée dans les études expérimentales (70 – 95%), la réponse de la biodiversité est peut-être minimale. Il est possible que les peuplements forestiers à courte rotation présentent une moins grande diversité d'oiseaux et de mammifères que les forêts aménagées, mais il existe encore beaucoup d'incertitude à ce sujet. Nous n'avons pas recensé d'études sur la réponse de la biodiversité à la culture intercalaire de graminées indigènes de saison chaude dans les forêts commerciales. Quelques études ou certains biais géographiques dans des études publiées sur certaines pratiques ont accru l'incertitude sur la cohérence des réponses observées dans différents paysages en Amérique du Nord. En réalisant des études sur l'impact qu'ont des récoltes plus fréquentes et en réalisant des études à des échelles spatiales plus vastes, on aurait une meilleure compréhension de la réponse de la biodiversité et on renforcerait les bases techniques sur lesquelles reposent les lignes directrices sur la récolte de la biomasse.

MOTS-CLÉS

aménagement forestier, amphibiens, bioénergie, biomasse, débris grossiers ligneux, diversité biologique, éclaircie, mesures de réduction du danger de feu, oiseaux, panic raide, *Panicum virgatum,* petits mammifères, peuplements forestiers à courte rotation, récolte intensive de la biomasse, reptiles

AUTRES PUBLICATIONS DE NCASI

Aucune

CONTENTS

TABLES

FIGURES

RELATIONSHIPS BETWEEN INTENSIVE BIOMASS PRODUCTION AND BIODIVERSITY IN NORTH AMERICAN FORESTS – A LITERATURE REVIEW

1.0 INTRODUCTION

Energy policy in the United States has increasingly promoted developing plant-based biofuels to complement, and potentially provide alternatives to, fossil fuels. One result has been more interest in intensive biomass production to meet anticipated growth in demand. Native grasses, woody species and forestry residues currently show promise as local energy sources for wood products facilities and for producing marketable biofuels (e.g., Winandy et al. 2008). Future development of cellulosic processes may further increase demand for intensive biomass production. Large-scale adoption of intensive biomass production in forests, however, would potentially alter management, species composition, physical structure, and landscape configuration of forests in some regions of the US. In some cases, tree species composition may remain similar, but timing of harvest, number of entries, and quantity of residual woody material left on site may be affected. Because forest lands support a large component of biodiversity in many regions, it is important to understand what is known about forest biodiversity response to practices associated with biomass production systems and what additional information is needed by forest managers and policymakers.

Our objective was to summarize documented relationships between intensive production of forest biomass and forest biodiversity in the US and identify knowledge gaps. This review is important because recent breakthroughs in cellulosic processes (e.g., Lau and Dale 2009) and recent additions to infrastructure suggest viable markets may soon emerge. Thus, there is a need to understand potential responses to increased bioenergy harvests (Berndes, Hoogwijk, and van den Broek 2003).

1.1 Potential Scope and Intensity of Increased Biomass Production in the United States

Availability of woody biomass for energy production depends largely on the amount and sustainability of biomass stocks and location of facilities that use it (i.e., costs of transporting the woody biomass to power generators). Gan and Smith (2006) estimated that over one-half of the nation's recoverable logging residues were present in forests of the Southeast and South-Central regions of United States. Thus, the greatest demand for woody biomass will likely be in the Southeast and South-Central regions of the US with a concomitant increase in biomass harvest. Other regions with potential for high demand for woody biomass include the Northeast and Pacific Northwest (Gan and Smith 2006). Previous publications (e.g., Cook, Beyea, and Keeler 1991; Hall 1997; Field, Campbell, and Lobell 2007) further discuss the potential scope of intensive biomass harvests in the US and possible impacts on biological diversity.

1.2 Biomass Harvesting Guidelines

In response to a growing and uncertain market, seven US states have developed biomass harvest guidelines that are similar in many ways (see Table 1.1). These guidelines focus on harvest methods that are likely to increase in the near term and that need further research and evaluation.

All guidelines emphasize retention of coarse (CWD) and fine woody debris (FWD), reflecting the expectation that collection of harvest residues will likely increase in the near future. Four of seven guidelines suggest that existing litter (prior to harvest), stumps and roots should be retained and disturbed as little as possible; five of seven recommend retaining as much of the existing FWD as possible (typically with little specific guidance); and six of seven provide recommendations for retaining harvest residue FWD. These recommendations range from retaining 1/6 - 1/3 of treetops as FWD to retaining as much as 33% of FWD on site.

Table 1.1 Comparison of Existing Biomass Harvest Guidelines in the United States
[*Italics* indicate recommendations that reference existing guidelines for general harvests.]

Provisions	ME	MA	MI	MN	MO	PA	WI
Litter, stumps and roots	Retain	--	--	Retain	--	Retain	Retain
FWD – Existing	Retain as much as possible	--	--	Retain as much as possible	Retain as much as possible	Retain as much as possible	Retain as much as possible
FWD – harvest residue	Retain as much as possible	--	Retain 1/3 – 1/6 tops	Retain 20%	Retain ≥ 33 %	Retain ≥ 10 %	Retain ≥ 10 %
CWD – existing	Retain as much as possible	--	Retain as much as possible	Retain as much as possible	--	--	Retain as much as possible
CWD – harvest residue	Retain as much as possible on site	General recommenda-tions only	Fell 2 – 5 leave logs / acre	*Retain 2 – 5 down logs*	Retain ≥ 33 %	Retain 10% - 30% Fell 2 – 5 leave logs / acre	--
Snags	*Retain all possible*	General recommendations	10 live & snag trees/acre	*Retain all possible*	Habitat specific	1 - 5 snags /acre	*≥ 3 / acre*
Retention	Retain live cavity trees	10-20%	10 live & snag trees/acre	*5% clumps OR 6 – 12 /acre*	Size class specific	5 cavity trees/acre	*≥ 5%*
Specific mast tree guidelines	YES	--	YES	--	YES	--	*YES*
Guidelines also include special considerations for:							
Discouraging re-entry	YES	YES	--	YES	YES	YES	--
Primary nesting season restrictions	--	--	--	--	--	--	--
T & E and/or rare habitat types	--	--	YES	YES	YES	YES	YES
Thinning	--	YES	--	--	YES	--	--
Harvest of understory vegetation	--	--	--	--	--	--	--
Short-rotation woody crops	--	--	--	--	--	YES	--
Intercropping	--	--	--	--	--	--	--

All guidelines address retention of CWD because it is a primary source of fiber in biomass harvests. Four of seven recommend retaining as much existing CWD as possible because it may already be colonized by insects, be soft enough for animals to excavate cavities, and may already be used as nesting or denning sites. Six of seven provide recommendations for retaining harvest residue CWD. These recommendations range from general (e.g., retain as much as possible) to specific (e.g., down logs/acre or % of harvest area). In some cases, biomass harvest recommendations simply refer to recommendations in existing timber harvest guidelines.

Snags (dead standing trees) and live trees are important components to retain for wildlife, but are also potential sources of woody biomass. All biomass harvest guidelines recommend retention of snags and green trees, although in many cases, the biomass guidelines simply reference existing timber harvest guidelines. Recommendations for snag and green tree retention range from general (retain as many as possible) to habitat- and size-specific (e.g., stems/acre or % of harvest area). Four of seven provide additional recommendations for retaining mast-producing trees.

Most guidelines (five of seven) explicitly encourage that biomass harvest occur concurrently with normal thinning or timber harvest operations to reduce impacts of multiple re-entries. Most plans also provide guidance for conservation-priority forests containing threatened and endangered species, sensitive plant communities, or rare habitat types. Only two guidelines specifically address woody biomass harvests during thinning operations, one set addresses harvest of short-rotation woody crops, and none address removal of understory vegetation or intercropping biomass plants with timber.

1.3 General Methodology for Literature Review and Meta-Analyses

We searched the literature for papers that characterized biodiversity responses to at least one of four treatments related to biomass harvesting: removal of forest harvest residues (CWD manipulations), thinning, intercropping, and short-rotation woody crops. We identified relevant studies by searching Wildlife and Ecology Worldwide, Web of Science, USDA Forest Service TreeSearch, and Google Scholar databases. We supplemented searches by examining bibliographies of articles and existing literature reviews (e.g., Jones, Hanberry, and Demarais 2009) for relevant studies. Biodiversity responses included species richness, species diversity, abundance of taxa or groups of species (guilds), and abundance of individual species for birds, mammals, reptiles, amphibians, and invertebrates.

We used meta-analysis (Borenstein et al. 2009) to summarize biodiversity response to the four biomass harvest treatments. Specific meta-analysis procedures for each biomass treatment were dictated by availability of literature and detailed methods are described in each section. We included both manipulative experiments (wildlife diversity and abundance measured before and after treatments) and observational studies of management practices (manipulated stands paired post-hoc with unmanipulated controls), hereafter referred to as "management experiments." We calculated response ratios (Hedges, Gurevitch, and Curtis 1999) and conducted all meta-analyses using Meta-Win software (Rosenberg, Adams, and Gurevitch 2000).

2.0 REMOVAL OF FOREST HARVEST RESIDUES

2.1 Background and Definitions

Traditional forest harvest operations often produce large amounts of woody residue consisting of treetops, limbs, slash, foliage, and felled non-crop trees and small-diameter trees that cannot be sold at value great enough to justify the costs of removing it from the site (i.e., unmerchantable stems). However, harvesting (i.e., removing) could become economically feasible because these residues have potential to help meet increasing demand for biofuel and allow the forest industry to participate in the emerging economic market for biomass feedstocks (see Section 1.1 above).

Forest harvest residues include growing stock volume cut or knocked down during harvest, low-quality commercial trees, deadwood and non-commercial tree species typically left at the harvest site (Gan and Smith 2006). Coarse woody debris (CWD) has been defined various ways and some consider it to include both down wood and standing snags (e.g., Loeb 1999). In this review, we will distinguish between snags and down coarse woody debris (DCWD). We consider ***snags*** to be standing dead trees ≥ 1.8 m in height and ≥ 10.2 cm diameter at breast height (dbh) following Thomas et al. (1979), although others may use slightly different girth and height criteria. We consider ***down coarse woody debris*** **(DCWD**) to be downed deadwood such as logs, stumps, piles of limbs, and other woody material of a minimum size found on the forest floor. Although no universally recognized size criteria exist (Jones, Hanberry, and Demarais 2009), most studies we reviewed defined DCWD as > 10 cm in dbh and > 60 cm in length. We define ***fine woody debris*** (FWD) as down, dead woody material < 10 cm in dbh or < 60 cm in length.

Although pilot and experimental biomass harvests have been conducted across North America (Arnosti et al. 2008; Evans and Finkral 2009), little is known about how forest biodiversity responds to removal of forest harvest residues. A primary mechanism for biodiversity response would likely occur through changes in numbers of snags and amount of DCWD and FWD. Harvest residues may represent a substantial input of DCWD. Thus, removal of harvest residues may impact amount of DCWD present during years following biomass harvest.

Although detailed information about biodiversity response to harvest residue removals has not been collected, importance of DCWD and snags to cavity-nesting birds and other wildlife has long been recognized (Harmon et al. 1986; Freedman et al. 1996; Russell et al. 2004; Jones, Hanberry, and Demarais 2009). As a result, some research efforts have experimentally manipulated levels of snags and DCWD in a way that closely mimics changes likely to occur from biomass harvests. To summarize biodiversity response to removal of snags and DCWD, we reviewed the literature and conducted a meta-analysis of experimental studies.

2.2 Methodology

We reviewed the literature for papers comparing biodiversity responses to experimental manipulations of DCWD and/or snags. Diversity responses included diversity metrics (i.e., species richness, diversity, or evenness), abundance of taxa or groups of species (guilds), and single-species abundance estimates. We included responses of birds, mammals, reptiles, amphibians and invertebrates. We included both manipulative experiments and management experiments where harvested areas were compared to appropriate unharvested controls. We included studies of salvage logging – harvest of merchantable residues after large forest fires to recover economic value of wood – because it is a viable management option in forests in western North America. Additionally, demand for woody biomass may increase frequency (and intensity) of salvage logging. Also, salvage logging mimics – to a certain extent – CWD manipulations likely to occur in biomass harvests, especially post-disturbance. Salvage logging experiments reduce standing dead biomass similar to what might occur when non-crop trees are removed at harvest and potentially reduce the future stock of snags.

We used Wildlife & Ecology Worldwide, USDA Forest Service TreeSearch and Google Scholar databases to search for relevant studies. We searched for the following forestry and biodiversity terms in article abstracts: coarse woody debris, fine woody debris, snags, harvest residue, slash, salvage logging, biodiversity, diversity, richness, wildlife, birds, avian, amphibians, reptiles, invertebrates, insects and mammals. We supplemented searches by examining bibliographies of articles for additional references.

We found 26 studies suitable for use in a meta-analysis (i.e., provided sample size and standard deviations for biodiversity responses) with the following experimental manipulations: removal of DCWD, addition of DCWD, snag removal (including salvage logging), snag addition and removal of both snags and DCWD (Table 2.1). These studies provided 745 individual effects sizes (Table 2.2). Because responses to habitat manipulations can vary greatly among taxa and among species within a taxon, we considered

different biodiversity measures (e.g., diversity, abundance, richness) from the same study to be independent effects (Bender, Contreras, and Fahrig 1998). For birds, we considered studies that presented analysis of breeding and winter bird responses as separate "studies" because behavior, habitat requirements, and composition of bird communities can be very different during those seasons. When studies presented comparisons in different types of forest, we similarly treated these as separate experiments because species' responses can vary among different forest types. When studies presented results of two separate experiments on the same study areas, but separated in time, we also treated those as independent experiments. When studies presented comparisons for a metric in consecutive years, we calculated the overall mean effect and standard deviation using the pooled variance. Some studies compared more than one treatment to the same control (e.g., Lohr, Gauthreaux, and Kilgo 2002), so these effect sizes would not be independent because they included data from the same control sites. To account for this, we conducted meta-analysis for each taxon across all manipulations and separately for each type of manipulation (see Table 2.2). Because of the number of studies and effect sizes available for birds, we were able to do additional meta-analysis by season and by nesting strategy (strong excavator, weak excavator, secondary cavity-nester, and non-cavity-nester following Martin, Aitken, and Wiebe [2004]).

Table 2.1 Summary of Manipulative Studies Used in Meta-Analysis

Study	Location	Forest Type	Taxa	Effect Sizes[a]
Caine & Marion 1991	Florida	Slash Pine	Birds	1, 2, 12, 0
Castro & Wise 2010	Kentucky	Mixed Hardwood	Invertebrates	3, 19, 0, 0
Haggard & Gaines 2001	Washington	Ponderosa Pine	Birds	0, 0, 13, 0
Hanula et al. 2006	South Carolina	Loblolly Pine	Invertebrates	4, 13, 0, 0
Horn & Hanula 2008	South Carolina	Loblolly Pine	Invertebrates	0, 0, 1, 9
Hutto & Gallo 2006	Montana	Ponderosa Pine	Birds	0, 1, 17, 0
Koivula & Schmiegelow 2007	Alberta	Boreal Mixedwoods	Birds	0, 1, 2, 0
Loeb 1999	South Carolina	Longleaf Pine	Mammals	1, 1, 11, 0
Lohr et al. 2002, Summer	South Carolina	Loblolly Pine	Birds	4, 14, 58, 0
Lohr et al. 2002, Winter	South Carolina	Loblolly Pine	Birds	4, 13, 56, 0
McPeek et al. 1987, Summer	Kentucky	Mixed Hardwoods	Birds	0, 0, 6, 0
McPeek et al. 1987, Winter	Kentucky	Mixed Hardwoods	Birds	0, 0, 6, 0
Morissette et al. 2002	Saskatchewan	Boreal Mixedwood	Birds	0, 0, 10, 0
Morissette et al. 2002	Saskatchewan	Jack Pine	Birds	0, 0, 20, 0
Moseley et al. 2005	South Carolina	Loblolly Pine	Invertebrates	0, 36, 0, 0
Moseley et al. 2008	South Carolina	Loblolly Pine	Mammals	0, 0, 9, 0
Osbourne & Anderson 2002	West Virginia	Mixed Hardwoods	Mammals	8, 2, 10, 0
Owens et al. 2008, Phase I	South Carolina	Loblolly Pine	Reptiles Amphibians	12, 18, 30, 0
Owens et al. 2008, Phase II	South Carolina	Loblolly Pine	Reptiles Amphibians	12, 18, 33, 0
Saab et al. 2007	Idaho	Ponderosa Pine	Birds	0, 0, 7, 0
Schwab et al. 2006	Labrador	Boreal Mixedwood	Birds	1, 1, 7, 0
Stepnisky 2003, Summer	Alberta	Boreal Mixedwood	Birds	0, 0, 3, 0
Stepnisky 2003, Winter	Alberta	Boreal Mixedwood	Birds	0, 0, 4, 0
Todd & Andrews 2008	South Carolina	Loblolly Pine	Reptile	0, 0, 2, 0
Todd et al. 2008	South Carolina	Loblolly Pine	Invertebrate	0, 0, 1, 0
Ulyshen & Hanula 2009	South Carolina	Loblolly Pine	Invertebrates	80, 150, 0, 0

[a] Number of effect sizes from each study for diversity, guild abundance, species abundance, and biomass, respectively.

Table 2.2 Effects of CWD Manipulations on Biodiversity by Taxa and Manipulation Type
[*Italics* indicates effect sizes from South Carolina only.]

	DCWD Removal	DCWD Addition	Snag Removal	Snag Addition	DCWD + Snags Removed	All Manipulations Combined
Birds						
Diversity[a]	*0.87 (n=4)* **		--	--	*0.78 (n=4)* **	0.78 (n=10) **
Abundance	*0.84 (n=13)* **	--	0.58 (n=10) **	0.32 (n=2) **	*0.61 (n=14)* **	0.65 (n=39) **
Species	*0.93 (n=57)*		0.81 (n=75) **	0.85 (n=24) **	*0.74 (n=57)* **	0.83 (n=213) **
	k = 2		k = 9	k = 3	*k = 2*	k = 14
Mammals						
Diversity	1.07 (n=4) **	0.98 (n=4)		--	--	1.03 (n=9)
Abundance	--	--	--	--	--	0.81 (n=3)
Species	1.16 (n=5)	0.98 (n=8) **		*1.02 (n=3)* **	*0.96 (n=16)*	1.00 (n=30)
	k = 1	k = 1		*k=1*	*k=2*	k = 4
Reptiles						
Diversity	*0.97 (n=2)* **	*0.89 (n=2)* **		*1.20 (n=4)* **	*1.02 (n=4)*	*1.03 (n=12)*
Abundance	*0.99 (n=3)*	*0.98 (n=3)* **	--	*1.05 (n=8)* **	*1.00 (n=6)*	*1.01 (n=18)*
Species	*1.00 (n=4)*	*1.00 (n=4)*		*1.01 (n=4)* **	*1.01 (n=10)* **	*1.01 (n=26)*
	k = 1	*k = 1*		*k = 2*	*k = 3*	*k = 3*
Amphibians						
Diversity	*0.95 (n=2)* **	*1.06 (n=2)* **		*1.09 (n=4)* **	*0.99 (n=4)*	*1.03 (n=12)*
Abundance	*0.91 (n=3)* **	*1.01 (n=3)*	--	*1.03 (n=6)*	*0.97 (n=6)* **	*0.98 (n=18)*
Species	*0.97 (n=6)* **	*1.00 (n=7)*		*1.00 (n=13)*	*1.00 (n=13)*	*1.00 (n=39)*
	k = 1	*k = 1*		*k = 1*	*k = 1*	*k = 2*
Invertebrates						
Diversity	0.95 (*n*=3)**	*1.00 (n=28)*		*1.04 (n=24)* **	*0.97 (n=32)* **	*1.00 (n=87)*
Abundance	0.86 (*n*=19)**	*1.09 (n=68)* **	--	*1.09 (n=50)* **	*0.92 (n=82)* **	*1.01 (n=219)*
Biomass	--	--		--	*0.83 (n=9)* **	*0.83 (n=9)* **
	k = 1	*k = 2*		*k = 1*	*k = 5*	*k = 6*

(Continued on next page. See notes at end of table.)

Table 2.2 Continued

	DCWD Removal	DCWD Addition	Snag Removal	Snag Addition	DCWD + Snags Removed	All Manipulations Combined
All Taxa Combined						
Diversity	0.99 (n=12)	0.99 (n=36)	--	1.05 (n=33) **	*0.96 (n=45)* **	0.99 (n=127)
Abundance	0.90 (n=20) **	1.03 (n=75)	0.58 (n=10) **	1.05 (n=64) **	*0.89 (n=109)* **	0.93 (n=278) **
Species	0.97 (n=79) **	1.00 (n=19)	0.81 (n=75) **	1.00 (n=48)	*0.92 (n=95)* **	0.91 (n=309) **
Biomass	--	--	--	--	*0.82 (n=9)* **	0.81 (n=9) **
	k = 4	k = 4	k = 9	k = 7	*k = 12*	k = 25

[a] Diversity includes species richness, diversity, and evenness responses; abundance includes abundance of taxa, species groups and guilds; species includes abundance of individual species; and biomass includes biomass of species or taxonomic groupings.
** Indicates bootstrap confidence intervals did not include 1.00; k = # of papers, n = # of effect sizes.

We conducted all meta-analyses using Meta-Win software (Rosenberg, Adams, and Gurevitch 2000). For each of 745 responses, we calculated a response ratio which is the ratio of the experimental to control groups (Hedges, Gurevitch, and Curtis 1999). For each response, we coded the low CWD treatment as the control so that response ratios could be consistently interpreted. For example, when DCWD was experimentally removed, we coded the removal group as the "control" group for meta-analysis. Response ratios < 1.00 indicate a negative response to lower CWD levels and ratios > 1.00 indicate a positive response to lower CWD. Because some means were zero, we added 1 to all means before calculating effect sizes. We used bootstrap confidence intervals and considered a combined effect to be significant if the confidence interval did not include 1.00. Some of our sub-group analyses are based on one or two studies so results should be treated with caution. However, even when the number of studies is small, summary effect sizes are still better than vote counting or other forms of summary (Borenstein et al. 2009).

2.3 Impacts of Snag and DCWD Harvest on Birds

2.3.1 *Results of Bird Species Meta-Analysis*

We found 262 bird responses (effect sizes) from 14 papers involving manipulations of snags, DCWD or both (Table 2.2). With the exception of invertebrates (315 responses, six papers), this was far more than the next most commonly studied taxon (amphibians, 69 responses, two papers), and thus we have more information about potential responses of birds to removal of harvest residues than other taxa. Most effect sizes (55%) were from studies in South Carolina. But effect sizes for birds when South Carolina studies were omitted (Figure 2.1b) were similar to those using all locations (Figure 2.1a, Table 2.2).

Bird communities were consistently less diverse and bird guilds were less abundant on treatments with lower levels of snags and/or DCWD (Table 2.2). Responses were consistently significant across all types of manipulations (e.g., DCWD removal, snag addition, etc.) and across diversity metrics (diversity, abundance of guilds, and species-level responses). With the exception of removing DCWD only, individual species abundances were also consistently lower on low CWD treatments (Table 2.2). Effects were much greater for birds than for other taxa (Table 2.2, Figure 2.1a). Thus, birds likely would respond negatively, at least in the short term at the stand level, to reductions in the density of snags or abundance of DCWD similar to that which might result from removal of forest harvest residues.

Across 85 effect sizes from nine studies of salvage logging, abundance of guilds and species were consistently lower on salvage-logged stands compared to unsalvaged controls (Table 2.2). None of the studies analyzed diversity metrics, and most focused on abundance of cavity-nesters. In particular, negative effects were consistently demonstrated for fire-dependent species like three-toed woodpecker (*Picoides dorsalis*) and black-backed woodpecker (*P. arcticus*) (Hutto 1995; Imbeau, Savard, and Gagnon 1999; Hobson and Schieck 1999; Morissette et al. 2002; Hutto and Gallo 2006; Schwab et al. 2006; Koivula and Schmiegelow 2007; Saab, Russell, and Dudley 2007). Multiple effect sizes for these two fire-dependent species did not skew overall results for species' responses because mean effect sizes were similar when we omitted effects sizes for these two species. Because salvage logging decreased abundance of all but a few cavity-nesters in these studies, we expect that diversity metrics – had they been reported – would have been lower as well.

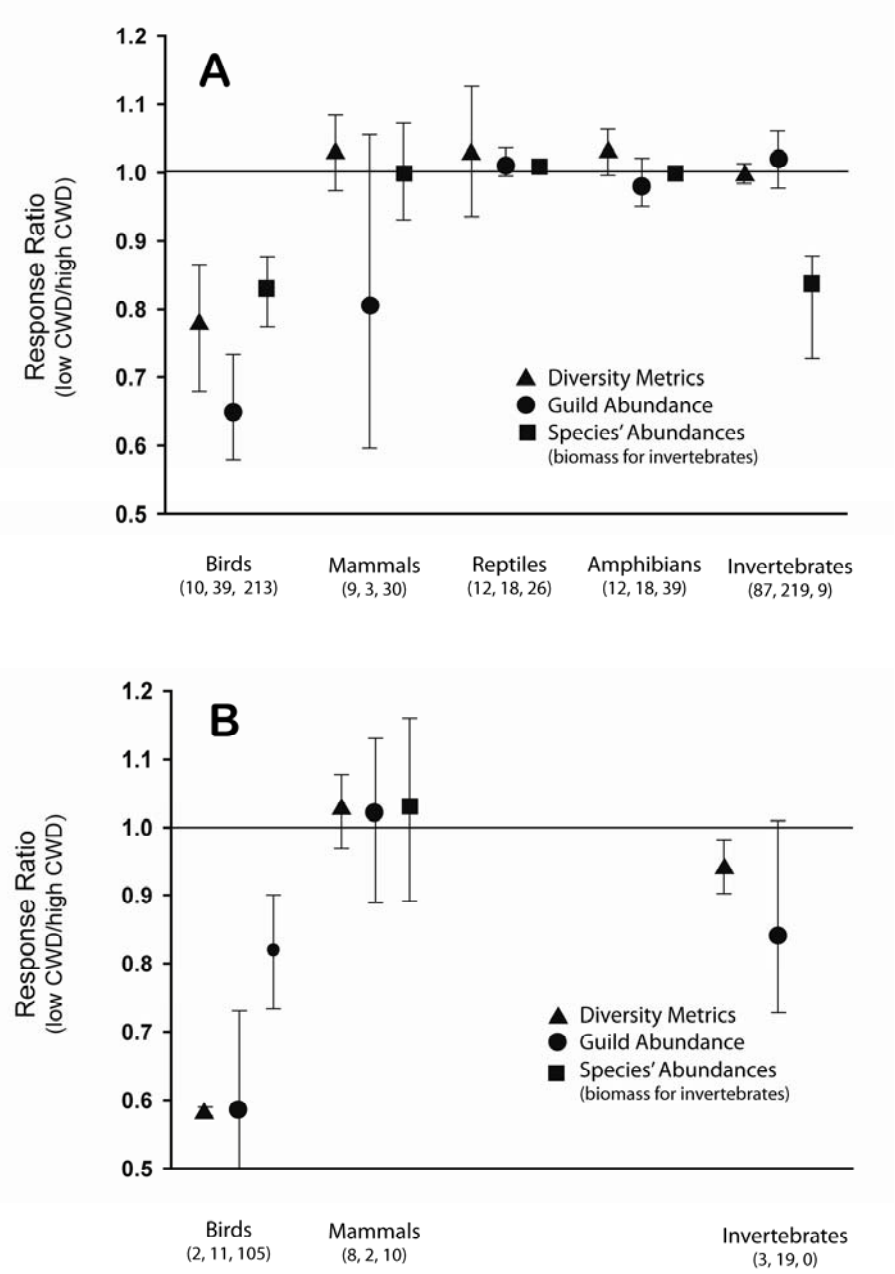

Figure 2.1 Summary Effect Sizes for Birds, Mammals, Reptiles, Amphibians and Invertebrates across All Manipulations – (a) Summary Effect Sizes Including All Studies; (b) Summary Effect Sizes with South Carolina Studies Omitted

Because salvage logging recovers merchantable standing timber, snag densities are typically reduced, but DCWD may be increased (e.g., Morissette et al. 2002) depending on how much harvest slash is left and/or scattered across the stand (McIver and Starr 2001). However, cumulative effect sizes for bird metrics from salvage logging experiments were similar to those from snag manipulations (0.81 vs. 0.78, $P = 0.502$), indicating that including salvage logging experiments in our meta-analyses did not bias our conclusions about removal of snags and DCWD.

2.3.2 *Effects of Season and Nesting Strategy on Bird Responses*

During the breeding season, strong excavators, weak excavators, secondary cavity nesters and non-cavity nesters all responded negatively to fewer snags and less DCWD (Figure 2.2), indicating birds that do not nest in snags (or DCWD) may respond negatively to reductions in snags and DCWD. Winter effect sizes were smaller or more variable, and only the effect size for weak excavators did not include 1.00 (Figure 2.2).

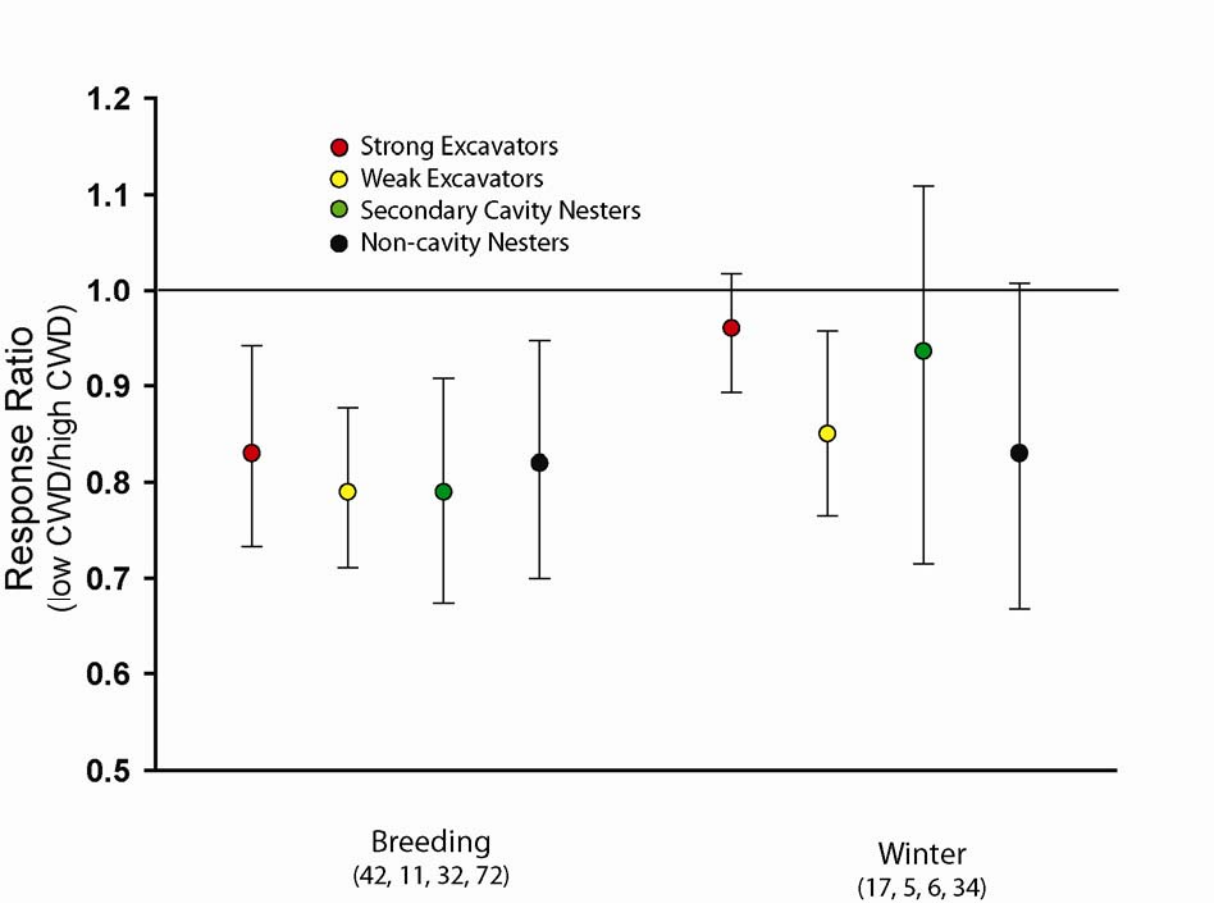

Figure 2.2 Summary Effect Sizes for Birds across All Manipulation Types

2.3.3 *Discussion of Bird Response*

Importance of snags to birds as sites for nesting, perching, foraging, and other functions is well-established. For example, primary cavity nesters (both strong and weak excavators, *sensu* Martin, Aitken, and Wiebe [2004]) excavate nest cavities in large, dead trees and secondary cavity nesters also depend on these cavities for breeding sites. Breeding opportunities may be limited by lack of cavities in some habitat types (e.g., Holt and Martin 1997). In a recent review of bird-forestry relationships, Sallabanks and Arnett (2005) reported that many more species decreased than increased when snags were removed. Our

analyses confirm these relationships and support loss of nesting substrates as a likely mechanism for these effects.

Birds also appear to respond negatively to decreases in DCWD, possibly through different mechanisms than those driving response to snags. During the breeding season, several non-cavity nesting species (e.g., eastern towhee [*Pipilo erythrophthalmus*], eastern wood-pewee [*Contopus virens*]) were less abundant when only DWCD was removed (Table 2.2; see also Lohr, Gauthreaux, and Kilgo 2002). Standing snags and DCWD may be important perching or foraging structure for a variety of birds possessing different life history traits. The causal mechanisms for these relationships have not been identified, but cavity- and open-nesters alike use both standing and down woody debris for foraging, perching and communication (Lohr, Gauthreaux, and Kilgo 2002 and references therein). One hypothesis is that removal of CWD reduces invertebrate abundance (see Section 2.7 below), thereby reducing habitat quality for birds other than just cavity-nesters. Thus, snags (and DCWD) should also be recognized as potential foraging substrates. Additionally, number of snags (or amount of DCWD) needed to meet foraging needs may be different from what is needed to meet nesting requirements (Hutto and Gallo 2006).

During winter, effects of CWD manipulations were not as large, and confidence intervals (for the most part) contained 1.00 (Figure 2.2). There are three hypotheses why this might happen. First, birds are typically non-territorial during winter and are thus not as strongly tied to a particular habitat type and forage over larger spatial areas (Lohr, Gauthreaux, and Kilgo 2002). Second, open canopies associated with snag and DCWD reduction may facilitate winter flock formation and improve predator vigilance, mitigating – at least in part – effects of snag removal on invertebrates or habitat structure (Lohr, Gauthreaux, and Kilgo 2002). Third, wintering bird communities contain a different suite of species which could account for at least part of differences in response. While ecologically plausible, these hypotheses need further testing to be confirmed. Also, stand-level studies address neither effects of nearby snag resources in non-biomass stands nor effects of how biomass harvests may be distributed across the landscape.

2.4 Impacts of Snag and DCWD Harvest on Small Mammals

2.4.1 *Results of Mammal Species Meta-Analysis*

We found 42 mammal responses (effect sizes) from four papers involving manipulations of snags, DCWD, or both. The four papers represent a limited amount of data on mammal diversity and abundance. Thus, overall effect size estimates should be treated with caution. Twelve of the 32 mammal effect sizes were from studies in South Carolina. But effect sizes for mammals when South Carolina studies were omitted (Figure 2.1b) were similar to those using all studies (Figure 2.1a, Table 2.2).

Overall effect sizes indicated little or no consistent response of mammal diversity to CWD manipulations (Table 2.2, Figure 2.1a), while meta-analysis summaries for DCWD removals (diversity only) and snag additions (species-level responses only) indicated higher values for mammal metrics in the low CWD treatments. When effect size estimates did not include 1.00, response ratios were much smaller than those for birds, indicating that mammals responded less strongly to CWD manipulations.

Within the individual manipulative studies we reviewed, effects of CWD manipulations were mixed. Loeb (1999) found higher overall numbers of cotton mice (*Peromyscus gossypinus*), short-tailed shrews (*Blarina carolinensis*) and cotton rats (*Sigmodon hispidus*) in some years on unsalvaged plots with higher density of snags and DCWD, but diversity was not affected. Osborne and Anderson (2002) found no negative effects of DCWD removal on mammals, and higher mammalian species richness. Likewise, Moseley et al. (2008) found that one of three species of shrews was lower on DCWD removals in one year of the study, but otherwise removal did not affect shrews. DCWD additions either decreased mammal abundance or had no effect. Hence, effects of removing harvest residues on mammals may be

hard to predict. At this time, there is no strong experimental evidence to suggest that diversity or abundance of mammals would be negatively affected by manipulation of snags and/or DCWD.

2.4.2 *Discussion of Small Mammal Response*

Correlative studies relating mammalian abundance and diversity to quantity of DCWD are common but report a variety of outcomes (see review by Osbourne and Anderson 2002). As a result, it is not surprising that effects from manipulative experiments were also mixed. Mammals often use DCWD as cover and for travel corridors (Waldien, Hayes, and Huso 2006), especially when predation risk is high (Zollner and Crane 2003). In addition, DCWD harbors food used by small mammals (e.g., Seastedt, Reddy, and Kline 1989). A positive correlation between volume of DCWD and abundance of small mammals has been demonstrated for several species of shrews, rats, and mice (Carey and Johnson 1995; Lee 1995; Maidens, Menzel, and Laerm 1998; Butts and McComb 2000; McCay 2000; McCay and Komoroski 2004; Cromer et al. 2007), but this relationship has not held in a number of other situations (Menzel et al. 1999, Bowman et al. 2000, Billig and Servello 2002, Payer and Harrison 2003, McCay and Komoroski 2004). Thus, mammal response to CWD manipulations may be highly context-dependent (e.g. differences in biophysical setting, predator density, etc.) or be mitigated by other factors.

Correlative studies not included in our meta-analyses suggest two important considerations relevant to biomass harvesting. First, some correlative studies hint that response of mammals to DCWD may be stronger in intensively managed forests where DCWD volumes are typically lower (Carey and Johnson 1995; Bowman et al. 2000). This is important because biomass harvests are most likely to occur in intensively managed forests. Second, mammals are often most strongly related to older, more decayed, and larger DCWD (Maidens et al. 1998; Bowman et al. 2000; Butts and McComb 2000; McCay 2000; Cromer et al. 2007). Given that existing biomass harvest guidelines typically restrict removal of existing DCWD, biomass harvests may not appreciably alter amount of old DCWD in the short term. Whether or not residue removal would change amount of old, decayed wood several years (or more) post-harvest is unknown.

2.5 Impacts of Snag and DCWD Harvest on Reptiles

2.5.1 *Results of Reptile Species Meta-Analysis*

We found 56 effect sizes for reptiles from three papers, all in loblolly pine forests in South Carolina. With all treatments combined, there were no significant effects on reptile diversity or abundance (Table 2.2 and Figure 2.1a). However, meta-analysis of individual manipulation types revealed that in both DCWD removals and DWCD additions, reptile diversity was slightly lower in treatments with lower DCWD (Table 2.2). Snag additions negatively impacted reptile diversity, abundance and species-level abundance. Snags and DCWD may have different effects on reptiles, and this underscores the importance of treating them separately.

2.5.2 *Discussion of Reptile Response*

As reflected in our literature review, reptiles have been less well-studied than other taxa (Russell et al. 2004; Jones, Hanberry, and Demarais 2009). Most forest reptiles use DCWD (Harmon et al. 1986) and reptile diversity and abundance may be associated with higher levels of DCWD (Enge and Marion 1986; Crosswhite, Fox, and Thill 2004). Reptiles likely use CWD for refugia, foraging substrates (i.e., insect and small mammal prey), basking, and mating displays (Harmon et al. 1986). As poikilotherms, they may benefit from the thermoregulatory and moisture-retaining functions of DCWD (and litter) in the same ways that have been documented for amphibians (deMaynadier and Hunter 1995; Russell et al. 2004; Semlitsch et al. 2009), but data specific to reptiles are sorely lacking.

We reviewed three manipulative experiments from South Carolina (Owens et al. 2008; Todd and Andrews 2008). Reptile diversity (and abundance) responded to manipulations of DCWD, increasing – but only slightly – when DCWD was increased, and decreasing when DCWD was removed (Owens et al. 200; Todd and Andrews 2008). However, reptile diversity and abundance responded in the opposite fashion when snags were removed, possibly because fewer snags decreased predation pressure from birds (Owens et al. 2008). The reptile community as a whole showed no response because in response to manipulations some species increased in abundance and others decreased (Owens et al. 2008).

Current evidence indicates that reptile responses to changes in DCWD and snags may not be large, but this evidence is based on a few studies from one region and needs to be confirmed by other studies. How CWD manipulations affect reptiles via changes in other taxa is another important research question (Owens et al 2008; Todd et al. 2008).

2.6 Impacts of Snag and DCWD Harvest on Amphibians

2.6.1 *Results of Amphibian Species Meta-Analysis*

We found 69 effect sizes for amphibians from 2 papers in South Carolina (both studies used the same experimental units but differed in time (see Owens et al. 2008). Over all manipulations combined, we found no significant response of amphibian diversity, guild abundance, or species abundances to CWD manipulations (Table 2.2, Figure 2.1a). Analysis of manipulation types separately indicated a small negative response of amphibian metrics to DCWD removal (RR = 0.91 – 0.97, Table 2.2) and a small negative response of guild abundance to removal of all CWD (RR = 0.97). In contrast, amphibian richness was slightly decreased when DWCD or snags were added (Table 2.1).

2.6.2 *Discussion of Amphibian Response*

Most forest amphibians use DCWD (Harmon et al. 1986). Decaying wood provides moist microhabitats where amphibians have a lower risk of desiccation (Harpole and Haas 1999; Semlitsch et al. 2009) which can lead to either higher survival (Rothermel and Luhring 2005) or lower evacuation rates from clearcuts (Semlitsch et al. 2008). However, amphibian responses to CWD manipulations in the studies we reviewed were both small and conflicting. For example, amphibian richness decreased both when DWCD was added and when it was removed. These conflicting responses may be caused in part because effect sizes were only from one forest type (loblolly pine forest in South Carolina), differential response of different species of amphibians, and a preference by many amphibians for old, more decayed DCWD (Owens et al. 2008, and see discussion below). A second possible explanation is that amphibians responded to the disturbance *per se* associated with manipulating woody debris.

Another large experimental study (LEAP – Land-use Effects on Amphibian Populations) compared partial cuts, clearcuts with DCWD retained and clearcuts with DCWD removed to unharvested controls in Missouri, South Carolina, and Maine (see Semlitsch et al. 2009 for summary). Although they did not provide diversity or abundance estimates that could be used in our meta-analysis, they did examine migration, water loss, survival, breeding success and other vital rates that are relevant to discussion of forest harvest residues. Because two of their treatments (clearcut retained and clearcut DCWD removed) mimicked what might happen during biomass harvest, their findings warrant discussion.

Retaining DCWD in clearcuts generally reduced impacts on amphibians relative to clearcuts with DCWD removed (Semlitsch et al. 2009). Retaining DCWD reduced desiccation risk and increased survival immediately after clearcutting (Rittenhouse et al. 2008), reduced evacuation rates (Semlitsch et al. 2008), and increased probability that migrating amphibians used clearcut patches (Todd et al. 2009). All three of these mechanisms could potentially influence amphibian diversity if DCWD removal was implemented at larger scales than this study (2 – 4 ha study plots, Semlitsch et al. 2008).

However, differences in amphibian response between retaining vs. removing DCWD after clearcutting were much smaller in magnitude compared to differences between clearcuts and partial harvests and controls (Semlitsch et al. 2009). Patrick, Hunter, and Calhoun (2006), working in Maine, found that spotted salamander (*Ambystoma maculatum*) had higher capture rates in DCWD-retained plots, but there were no differences for other species. DCWD retention did not influence habitat selection and movement patterns of northern leopard frogs (*Lithobates pipiens*) in Maine (Blomquist and Hunter 2009), nor did it affect breeding effort of gray treefrogs (*Hyla versicolor*) in Missouri (Hocking and Semlitsch 2007).

Response of amphibians to DCWD manipulations may depend on interactions with other habitat features, surrounding conditions, and temporally variable environmental conditions. For example, use of DCWD by mole salamanders (*A. talpoideum*) in loblolly pine forests was greatest when litter layer was reduced (Moseley et al. 2004), suggesting that DCWD may be important to salamanders only when their primary refuge – litter – is not available.

Importance of DCWD to forest amphibians may also vary with time. The lack of amphibian response to DCWD manipulations may be due to their preference for old decayed DCWD (Herbeck and Larsen 1999; Grialou, West, and Wilkins 2000; Hicks and Pearson 2003; McKenny, Keeton, and Donovan 2006) when compared to the newly created CWD found in most experimental manipulations (just as with mammals, see above). In Vermont, red-backed salamanders (*Plethodon cinereus*) were not affected by forest treatments designed to enhance DCWD and structural heterogeneity in general (McKenny, Keeton, and Donovan 2006). Yet, volume of highly decayed DCWD (but not newly created DCWD) was still highly correlated (within treatments) to salamander abundance (McKenny, Keeton, and Donovan 2006). Thus, new DCWD created in manipulative experiments may not influence amphibian diversity immediately. Removals for biomass harvest may not elicit strong responses from amphibian in the short-term, but long-term responses (several years post harvest) could be much stronger if removing harvest residue changes the long-term availability of older, decayed DCWD in managed forests.

2.7 Impacts of Snag and DCWD Harvest on Invertebrates

2.7.1 *Results of Invertebrate Species Meta-Analysis*

We found 315 effect sizes for invertebrates from six papers. The large number of effect sizes reflects the inherent diversity of arthropods, rather than a wide geographical distribution of effect sizes, as all were from studies conducted in South Carolina. Invertebrates were rarely distinguished at the species level. Thus, species metrics are lacking for invertebrates and diversity metrics represent taxonomic diversity, morphospecies richness (Hanula, Horn, and Wade 2006), and abundance of orders or families.

Over all manipulations, only invertebrate biomass was significantly reduced on low CWD treatments (RR = 0.83, Table 2.2, Figure 2.1a). Analysis of manipulation types revealed that diversity and abundance of invertebrates responded negatively when DCWD and snags were both removed. In contrast, diversity responded negatively when DCWD or snags were added. Thus, there was little evidence over all manipulations that invertebrate diversity or abundance responded strongly to CWD manipulations. Biomass was substantially reduced when CWD was removed, but this response was based on only N = 9 effect sizes from one study (Table 2.1). Most effect sizes (93%) were from studies in South Carolina. But effect sizes for invertebrates in the one study outside South Carolina (Figure 2.1b) were more strongly negative compared to when we included all locations (Figure 2.1a, Table 2.2).

2.7.2 *Discussion of Invertebrate Response*

With the exception of biomass, we found a few, contrasting, small responses of invertebrates to CWD manipulations. Because all effect sizes were from loblolly pine forests in South Carolina, it would be unwise to extrapolate these results to other regions and forest types. However, two interesting items emerged from these studies that warrant additional study.

First, invertebrate biomass – rather than diversity – may show greatest response to DCWD and snags manipulations related to biomass harvests. Horn and Hanula (2008:165) found that removing deadwood "affected lots of groups in small amounts resulting in a cumulative reduction of available prey." This overall reduction in invertebrate prey is consistent with – and could be one cause of – the consistently negative response by birds to removal or reduction of down and standing deadwood (e.g. Lohr, Gauthreaux, and Kilgo 2002). More research about the links among deadwood, invertebrate prey, and birds is needed to establish this as a causal relationship.

Second, removal or reduction of DCWD and snags may alter predator-prey relationships among insect groups and other taxa. For example, imported fire ants (*Solenopsis invicta*) are more likely to colonize clearcut forests in the Southeast, but less so when DCWD is retained (Todd et al. 2008). Thus, biomass harvest of forest residues could potentially facilitate invasions by undesirable insects. CWD reduction also may reduce invertebrate prey for birds (see discussion in Section 2.3) and for primarily predatory insects by decreasing prey abundance, altering prey distribution, or by making prey more difficult to find (Ulyshen and Hanula 2009). More research on DCWD and snag manipulations and response of species interactions is needed.

2.8 Impacts of FWD Harvest on Biodiversity

Fine woody debris (< 10 cm dbh or < 60 cm length) is typically considered an important structural part of forest ecosystems. FWD from limbs and treetops can accumulate after management activities such as thinning and commercial harvests and thus, FWD is also a potentially viable source of biofuel. Although typically considered an important habitat feature for many types of wildlife, we found only one study that examined wildlife response to changes in FWD. Manning and Edge (2008) compared small mammal use of three arrangements of post-harvest FWD: piled; lopped and scattered; and pile burned.

Although both deer mice (*Peromyscus maniculatus*) and western red-backed voles (*Clethrionomys californicus*) used FWD disproportionately to its availability, different arrangements did not translate to differences at the population level (Manning and Edge 2008). Removing FWD in Appalachian forests decreased spider density (Castro and Wise 2009). Although diversity was not affected, FWD manipulations affected community composition as different spider guilds responded both positively and negatively (Castro and Wise 2009). Additionally, community composition was affected by FWD manipulations.

Because so little is known about how animals respond to changes in FWD, it is difficult to accurately predict how biodiversity will respond to changes in FWD related to removing harvest residues. Animal responses to changes in FWD may be determined in part by amount of DCWD present in the stand. Species richness of wood-inhabiting invertebrates may increase with increased FWD, but only when levels of DCWD are low (Kruys and Jonsson 1999). Different components of forest harvest residue (e.g., DCWD, FWD, litter) may interact – rather than act independently – to influence biodiversity responses.

2.9 Indirect, Long-Term Response to Changes in Forest Productivity

Removing harvest residues also removes nutrients, which can potentially decrease productivity of the next stand of trees (e.g., see Scott and Dean 2006; Eisenbies et al. 2009 for reviews). Although a complete discussion on effects of harvest residue removal on site productivity is outside the scope of our review, decreases in future site productivity could potentially result in changes in forest structure and/or plant communities. Because many animals respond to habitat quality or habitat cues provided by vegetation structure, this may indirectly influence biodiversity. Fortunately, decreased productivity from residue removal can be ameliorated through fertilization (Scott and Dean 2006; Eisenbies et al. 2009). In intensively managed forests, the economic incentive to ameliorate nutrient losses is large, and productivity-related changes to plant communities would likely be ameliorated as a consequence of maintaining forest productivity.

2.10 Geographic Limitations, Empirical Knowledge Gaps and Research Needs

Experimental evidence we reviewed was primarily from the Southeast and Pacific Northwest (Figure 2.3.). In fact, 585 (79%) of our effect sizes were from South Carolina. Fortunately, the Southeast and Northwest are where most forest harvest residues are located (Gan and Smith 2006; Woodall, Heath, and Smith 2008). Even so, it would be unwise to extrapolate our findings to forested regions that have not been studied (e.g., Great Lakes or Northeastern US). We conducted a second meta-analysis omitting effect sizes from South Carolina (Figure 2.1b), and arrived at similar conclusions for birds and mammals. This indicates that the geographical bias did not substantially influence our conclusions for these taxa. However, effects sizes for invertebrates were more negative in one study from Kentucky (Castro and Wise 2010), while effect sizes for reptiles and amphibians were limited to South Carolina. Additional studies in the Southeast and other regions are needed.

Figure 2.3 Distribution across North America of Effects Sizes from Studies Included in Coarse Woody Debris Meta-Analyses

Below we describe a series of empirical knowledge gaps and research needs that limit our ability to accurately predict how forest harvest residue removal might impact biodiversity in managed forests.

2.10.1 *How Many Snags, DCWD, and FWD Actually Remain after Biofuel Harvests?*

Although quantitative data are scarce, some pilot projects report that post-harvest levels of woody debris may equal or even be greater than pre-harvest levels (Arnosti et al. 2008), especially when harvest guidelines include provisions for retention of CWD. For example, in two pilot projects where all submerchantable timber and up to 80% of down woody material were harvested, snag density was unaffected, FWD decreased by < 20%, and DCWD changed -0.4% and + 13%, respectively (Arnosti et al. 2008). Even when all down woody material was removed in conjunction with a 50% harvest of submerchantable timber and shrubs, DCWD was only 4.5% less abundant post-harvest (Arnosti et al. 2008). These are very small reductions compared to the manipulative studies we reviewed (≥ 90% reductions in some studies, e.g. Todd and Andrews 2008). Considering that impacts on diversity and abundance were not large in the manipulative experiments we reviewed (with the exception of birds), it is possible that biofuel harvests may not substantially alter biodiversity in managed forests, especially when following guidelines that provide for retention of minimum levels of woody debris (see Section 1.1). It will be difficult to accurately predict impacts of forest harvest residue removal on biodiversity until such information has been gathered.

2.10.2 *Long-Term Studies Are Lacking*

Lack of long-term studies limits our ability to predict effects of removing harvest residues in three ways. First, long-term studies are needed to describe decay dynamics of CWD after residue removals several years post-harvest (Moseley et al. 2008; Owens et al. 2008). This is critical because older, more decayed DCWD may be most important for wildlife (Bowman et al. 2000; McKenny, Keeton, and Donovan 2006), but how harvest residue removal impacts temporal succession of DCWD in different types of forest is unknown. Second, large year-to-year fluctuations unrelated to forest management are common (especially in mammals) and may mask underlying differences related to DCWD manipulations in one- to three-year studies (Osborne and Anderson 2002). Third, climate-related fluctuations in diversity and abundance may similarly mask effects in short-term experiments (Moseley et al. 2008), particularly for taxa such as amphibians. Results from one- to three-year experiments may not accurately predict DCWD and snag dynamics or biodiversity responses during other periods with different climate conditions. Long-term experiments are needed to examine these mechanisms.

2.10.3 *How do Predator-Prey Relationships Change in Response to CWD Manipulations*?

Communities and populations may be impacted indirectly by changes in predators (or prey) rather than from changes in habitat structure. For example, non-cavity-nesting birds may be impacted by diminished foraging substrates and fewer invertebrates when DCWD is decreased (Lohr, Gauthreaux, and Kilgo 2002). In contrast, reptiles may increase when standing snags are removed because decreased avian predation may offset loss of CDWD habitat (Owens et al. 2008). Among insects, predator taxa may similarly benefit most from DCWD (Ulyshen and Hanula 2009) In another study, the invasive red imported fire ant – a known predator on birds, mammals, reptiles and amphibians (Allen, Epperson, and Garmestani 2004; Suarez, Yeh, and Case 2005) – was more abundant in DCWD removal plots (Todd et al. 2008).

2.10.4 W*hat are the Interactions Among Snags, DCWD, FWD, and Litter?*

Manipulative experiments to determine how litter, FWD, DCWD and snags interact to provide habitat needs of animals (especially amphibians and reptiles) are needed. Future manipulative experiments should examine role of FWD (about which little is known), vary two or more of these four habitat components in factorial designs to determine interactions (Kruys and Jonsson 1999; McKenny, Keeton, and Donovan

2006) and manipulate each component over a range of values to test for thresholds (Semlitsch et al. 2008).

2.10.5 *How do Results from Small Experiments Scale Up to Operational Extents?*

Most manipulative studies have used relatively small experimental units embedded in an unharvested matrix (e.g., 9.3 ha; Owens et al. 2008). Quite possibly, if CWD was removed over larger areas (as would happen if harvest residue removal became common), these large, CWD-poor areas might be less readily colonized from surrounding areas compared to the smaller plots used in experiments (Ulyshen and Hanula 2009). If so, small-scale experiments may underestimate extent and magnitude of biodiversity response to biomass harvests. Conversely, small-scale experiments may overestimate biodiversity response over large areas if biomass harvests occur only in a small percent of stands in a landscape. Large, operational-scale experiments are clearly warranted. These experiments should also be conducted in a variety of landscape contexts and should include measurements of reproduction and survival so that the source/sink status of harvested areas (relative to the surrounding landscape) can be determined.

2.11 Summary of Biodiversity Impacts and Management Implications

Based on our analyses, birds would be most likely to respond negatively, at least in the short term, to extensive removal of forest harvest residues, if removals resulted in substantially less CWD or snags across large spatial scales. Our analyses also tentatively suggest potential for decreased biomass of invertebrates (but not diversity or abundance of taxonomic groups), but this result was based on 9 effect sizes from a single experiment. We found little evidence for large or consistent responses by mammals, reptiles, and amphibians to CWD and snag manipulations.

Several operational realities may limit negative impacts of biomass harvesting. First, biomass harvesting will not likely happen everywhere or happen at the same time. Second, even in landscapes where biomass harvest is frequently practiced, deadwood resources (snags, DCWD, and FWD) will likely be abundant in other portions of the landscape where biomass harvesting does not occur (or has not occurred recently). Thus, deadwood resources should be available in much of the landscape at any given point in time. Third, the increasingly diverse (and fragmented) forest ownership in many parts of the US will help ensure high landscape-scale diversity (i.e., mosaic of forest ages, types, and uses) which is necessary to maintain high biodiversity. Additionally, biomass harvests will most likely occur under the auspices of sustainability programs that will require consideration for biodiversity. Although current experimental evidence and operational realities suggest that removal of forest harvest residues has potential to be a sustainable option for meeting biofuel demand, the geographical limitations and knowledge gaps (see above) prevent any reasonably certain predictions about how biodiversity would respond to widespread removal of forest harvest residues.

3.0 THINNING AND FUELS TREATMENTS

3.1 Background and Definitions

Forest thinning is a silvicultural treatment that reduces tree density primarily to improve tree growth, to enhance forest health, or for economic reasons (Helms 1998). Forests naturally thin through tree mortality resulting from competition in dense stands. Stands can be thinned before competitive self-thinning to meet objectives relative to economic needs, biodiversity conservation (Hayes et al. 1997; Carey and Wilson 2001; Hayes, Weikel, and Huso 2003), and forest restoration (Hayes, Weikel, and Huso 2003; Harrod et al. 2009). Wood products resulting from thinning operations are used in a variety of ways, although currently up to 60% of harvested material remains on site (Parikka 2004). An increase in availability of biofuels-processing facilities may increase removal and use of thinned material (USDA

Forest Service 2005), which may partially offset harvest cost while meeting some of the increasing demand for biofuels (Page-Dumroese, Jurgensen, and Terry 2010).

Thinning can increase structural complexity of young forests, subsequently increasing wildlife species diversity (Spies et al. 1991; Hayes et al. 1997; Tappeiner et al. 1997). Thinning produces a variety of short- and long-term changes to forest structure, the most obvious of which is a decrease in tree density and increase in forest canopy gaps and abundance and diversity of midstory trees (Artman 2003; Agee and Skinner 2005; Hayes, Weikel, and Huso 2003; Harrod et al. 2009). The more profound effect for wildlife species may be related to development of more complex understory vegetation due to increased light availability below the canopy (Doerr and Sandburg 1986; Bailey and Tappeiner 1998; Wilson and Carey 2000; Garman 2001; Homyack, Harrison, and Krohn 2005). Despite the favorable response of many species to thinning treatments, causal relationships between complexity of understory vegetation and increased species abundance or diversity have not been identified (Wilson, Anderson, and Puettmann 2009). In addition, variable thinning intensities and harvest patterns (e.g., variable density thinning, clumped retention, or patch cuts) can produce favorable forest stand conditions for a variety of fauna (Carey and Wilson 2001; Garman 2001; Carey 2003).

Thinning can be represented in three broad categories: pre-commercial thinning, commercial thinning, and fuels treatment thinning. The frequency with which each of these strategies is used across a landscape depends on landowner objectives, forest type, physiographic region, and other considerations. Often, a combination of these silvicultural treatments is used to achieve wood fiber, biodiversity, and forest health goals.

Managing regenerating stands often requires thinning of overstocked stands to maximize commercial harvest wood volume. **Pre-commercial thinning** (PCT) is the removal of trees, not for immediate financial return, but to reduce stocking density which allows increased growth of more desirable crop trees (Helms 1998). Pre-commercial thinning occurs early in stand development, either before or just after canopy closure. The removal of sub-dominant sapling trees allows production of merchantable wood to increase substantially throughout the remainder of the rotation period (Reukema 1975). Pre-commercial thinning is commonly used in the Northwest (especially in Douglas fir forest types [Briggs 2007]), increasingly used in Acadian forests of the Northeast (Homyack, Harrison, and Krohn 2007), decreasingly used on industrial forest lands in the Upper Midwest (D'Amato et al. 2008), and not common in commercial forests of the Southeast (Folegatti, Smidt, and Dubois 2007).

Commercial thinning is a partial-cutting process that produces merchantable material at least equal to the value of direct costs of harvesting (Helms 1998). Commercial thinning can occur at any time following canopy closure (Artman 2003). Two-stage or multiple-entry overstory removal through commercial thinning can be employed to encourage understory development that simulates late seral forest characteristics at earlier ages (Thysell and Carey 2001; Poage and Tappeiner, II 2002; Hagar, Howlin, and Ganio 2004). However, few data have been presented to document success of thinning in producing such conditions (Lindh and Muir 2004). The extent to which thinning of merchantable trees will be used for biofuels production is also unknown, and will likely depend heavily on fluctuating markets.

A **fuels treatment** is any manipulation or removal of wildland fuels to reduce likelihood of ignition or to lessen potential damage and resistance to control (Helms 1998). As a result of decades of fire suppression, fuels treatment forest thinning is increasingly used across the Western US and Canada as a mechanism to reduce forest understory density and restore forest health (Agee and Skinner 2005; USDA Forest Service 2005). Mechanical thinning of understory vegetation is becoming commonplace in dry forests of the Southwest (USDA Forest Service 2005). Fuels treatments remove dense sapling trees and other woody understory vegetation to reduce ladder fuels that can lead to uncharacteristic stand-replacing wildfire (Agee and Skinner 2005). However, depending on length of time that fire has been suppressed from the

stand, fuels treatment thinning can include thinning of merchantable trees to decrease crown density and add more wood volume to the timber sale (Skog and Barbour 2006). As a result, volume of wood removed in fuels treatment thinning is widely variable, and likely differs significantly by region and forest type. However, total basal area removed is often less than for commercial and pre-commercial thinning treatments. Biomass removal as a fuels reduction treatment has been shown to be effective at decreasing near-term fire risk, but results may be mixed over longer time periods (Reinhardt, Holsinger, and Keane 2010).

Although pilot and experimental biomass harvests have been conducted across North America (Arnosti et al. 2008; Evans and Finkral 2009), knowledge of how biodiversity responds to forest thinning is incomplete. Although the southeastern US is the leading timber-producing region of the United States (Prestemon and Abt 2002), and thinning is a common silvicultural practice in all regions, most research on effects of thinning on wildlife species has been conducted in the US Northwest. Reviews of forest thinning effects to date have been regional or local in geographic scope and primarily qualitative in their assessment (Hayes et al. 1997; Harrison 1999; Muir et al. 2002; Thompson, Baker, and Ter-Mikaelian 2003). However, detailed information about biodiversity response to forest thinning has recently been assessed quantitatively for the southwestern United States (Kalies, Chambers, and Covington 2010).

Our objective was to assess response of wildlife species diversity and abundance to different types of forest thinning. Most current research, however, offers a snapshot assessment of forest thinning effects. Effects of forest thinning operations on measures of diversity are often highly dependent on time since harvest, as many harvests will have a negative short-term effect on both species abundance and diversity (Wilson and Puettmann 2007). Thus, we used a continent-wide meta-analytic approach that represents a more comprehensive assessment of effects of biomass thinning harvests on terrestrial biodiversity across a variety of forest types and taxa.

3.2 Methodology

We reviewed the literature for papers that compared biodiversity responses to various thinning treatments. Biodiversity responses included species richness, diversity, abundance of taxa or groups of species (guilds), and abundance of individual species for birds, mammals, reptiles, amphibians, and invertebrates. We included both manipulative experiments (wildlife diversity and abundance measured before and after thinning treatments) and management experiments (stands paired post-hoc and thinned areas are compared to unthinned controls). The controls presented were most commonly unthinned harvest-aged stands (30-75 years old). We included studies of precommercial, commercial, and fuels treatment thinning.

We used Wildlife and Ecology Worldwide, Web of Science, USDA Forest Service TreeSearch, and Google Scholar databases to search for relevant studies. We searched for the following forestry and biodiversity terms in article abstracts: amphibian, avian, bird, biodiversity, diversity, fuels treatment, insect, invertebrate, mammal, precommercial thinning, shelterwood, reptile, richness, selection harvest, and thinning. We supplemented searches by examining bibliographies of articles for additional references.

We found 33 studies (k=33) relative to effects of forest thinning on wildlife species that provided control and treatment means, sample size, and standard deviations for biodiversity responses making them suitable for meta-analyses (Table 3.1). Several otherwise suitable studies did not report standard deviations or standard error measures. In some cases, treatment and control means were provided with an associated two sample t-test statistic, p-value and degrees of freedom. When this occurred, we used the pooled variance in place of individual treatment and control standard deviation measures. When neither standard deviation nor test-statistic/p-values were reported, we contacted the authors and, when the data were available, we calculated error values from raw data. If error measures could not be back-calculated and raw data were not available, we did not include the studies in analyses, but did include them in the

discussion. When studies presented comparisons for a metric in consecutive years, we calculated overall mean effect and standard deviation using the pooled variance.

Because responses to habitat manipulations can vary greatly among taxa and among species within taxa, we considered different biodiversity measures (e.g., diversity, guild abundance, species abundance) from the same study to be independent effects (Bender, Contreras, and Fahrig 1998). For birds, we also calculated two separate measures of effect size for species measured in summer and winter because behavior, habitat use, and composition of bird communities often differ during those two seasons.

When studies presented comparisons of different intensities of thinning, we similarly treated these as separate experiments because species' responses can vary among these treatments. However, many studies reporting results from different thinning intensities used the same control stands to compare each thinning intensity. To account for this lack of independence, we conducted meta-analyses for each taxon across all thinning intensities and separately for each thinning intensity. Because we considered myriad forest types with different potential stocking densities, we determined thinning intensity by calculating percent of unthinned (control) stand basal area or trees per hectare remaining in thinned (treatment) stands and categorized them as heavy (0-33%), moderate (34-66%), or light thins (67-100%). In addition, we calculated the average timing of data collection (years post-treatment) for each taxa.

Of the 33 studies we selected, 19 were manipulative experiments and 14 were management experiments. The literature included 11 precommercial thinning, 12 commercial thinning, and 10 fuels treatment studies, resulting in 505 individual effects sizes (Table 3.1). None of the studies reviewed were specifically designed to test thinning as a biomass removal technique. However, precommercial, commercial, and fuels-treatment thinning are possible mechanisms for biomass harvest. As a result, we included all types of thinning in a cumulative meta-analysis. In addition, we analyzed fuels treatment studies separately because that technique is more likely to be used for stand restoration than for commercial harvest. Most studies were from the Northwest region (16), with fewer in other regions (Southeast [10]; Southwest [4]; Northeast [3]). See Figure 3.1.

Table 3.1 Summary of Manipulative Studies Used in Meta-Analysis

Study	Location	Forest Type	Taxa	Effect Sizes[a]	Thinning Intensity[b]	Commercial Thin?	Fuel Treatment Thin?	Data Collection
Amacher et al. 2008	California	Mixed conifer	Mammals	0,0,0,4	Light	Y	Y	1 yr post-treatment
Artman 2003	Washington	Hemlock/ Douglas fir	Breeding birds	0,0,3,11	Moderate	Y	N	3 to 5 yrs post-treatment
Carey and Wilson 2001	Washington	Hemlock/ Douglas fir	Mammals	0,0,0,7	Light	Y	N	1 to 4 years post-treatment
Dellasala et al. 1996 A	SE Alaska	Hemlock/ Douglas fir	Breeding birds	1,1,0,15	Moderate	N (PCT)	N	3 to 5 yrs post-treatment
Dellasala et al. 1996 B	SE Alaska	Hemlock/ Douglas fir	Winter birds	1,1,0,3	Moderate	N (PCT)	N	3 to 5 yrs post-treatment
Easton and Martin 1998	British Columbia	Cedar/ Hemlock	Breeding birds	0,1,11,0	Light	N (understory)	N	1 to 3 yrs post-treatment
Ford et al. 2000	North Carolina	Appalachian N. Red Oak	Amphibians and Mammals	0,0,0,10	Heavy	Y	N	2 years post-treatment
Garman 2001 A	Oregon	Hemlock/ Douglas fir	Amphibians and Mammals	1,1,1,10	Moderate	N (PCT)	N	5, 6 and 8 years post-treatment
Garman 2001 B	Oregon	Hemlock/ Douglas fir	Amphibians and Mammals	1,1,1,10	Heavy	N (PCT)	N	5, 6 and 8 years post-treatment
Greenberg et al. 2007	North Carolina	Oak-Hickory hardwood	Birds and Mammals	0,0,1,18	Light	N (understory)	Y	1 to 4 years post-treatment
Hagar et al. 1996 A	Oregon	Hemlock/ Douglas fir	Breeding birds	1,1,0,20	Light	Y	N	5 to 15 years post-treatment
Hagar et al. 1996 B	Oregon	Hemlock/ Douglas fir	Winter birds	1,1,0,6	Light	Y	N	5 to 15 years post-treatment
Hagar et al. 2004 A	Oregon	Hemlock/ Douglas fir	Breeding birds	1,1,0,24	Moderate	N (PCT)	N	1 to 4 years post-treatment
Hagar et al. 2004 B	Oregon	Hemlock/ Douglas fir	Breeding birds	1,1,0,25	Heavy	N (PCT)	N	1 to 4 years post-treatment
Homyack et al. 2005	Maine	Acadian	Mammals	0,0,0,4	Light	N (PCT)	N	1 to 17 years post-treatment
Hurteau et al. 2008	Arizona	Ponderosa pine	Breeding birds	1,0,0,5	Moderate	N (understory)	Y	2 to 4 years post-treatment

(Continued on next page. See notes at end of table.)

Table 3.1 Continued

Study	Location	Forest Type	Taxa	Effect Sizes[a]	Thinning Intensity[b]	Commercial Thin?	Fuel Treatment Thin?	Data Collection
Kilpatrick et al. 2004	South Carolina	Southern pine	Amphibians and Reptiles	1,1,0,9	Light	N (understory)	Y	1 to 2 years post-treatment
Klenner and Sullivan 2003	British Columbia	Subalpine spruce-fir	Mammals	0,0,0,4	Light	Y	N	1 to 3 yrs post-treatment
Larson 2001 A	Oregon	Hemlock/ Douglas fir	Mammals	0,0,0,10	Moderate	Y	N	4 to 6 years post-treatment
Larson 2001 B	Oregon	Hemlock/ Douglas fir	Mammals	0,0,0,10	Heavy	Y	N	4 to 6 years post-treatment
Loeb and Waldrop 2008	South Carolina	Southern pine	Mammals	0,1,0,3	Light	N (understory)	Y	1 to 2 years post-treatment
Matthews et al. 2010	North Carolina	Mixed Oak-Hickory-Pine	Amphibians and Reptiles	2,2,5,4	Light	N (understory)	Y	3 years post-treatment
Norton & Hannon 1997 A	Alberta	Aspen	Breeding birds	0,0,6,0	Moderate	Y	N	1 to 2 years post-treatment
Norton and Hannon 1997 B	Alberta	Aspen	Breeding birds	0,0,6,0	Heavy	Y	N	1 to 2 years post-treatment
Perry and Thill 2005	Arkansas	Mixed pine-hardwood	Mammals	0,1,0,5	Moderate	Y	N	1.5 to 5.5 years post-treatment
Ransome et al. 2004 A	British Columbia	Lodgepole pine	Mammals	0,0,0,2	Moderate	N (PCT)	N	12 to 14 years post-treatment
Ransome et al. 2004 B	British Columbia	Lodgepole pine	Mammals	0,0,0,4	Heavy	N (PCT)	N	12 to 14 years post-treatment
Siegel and DeSante 2003	California	Mixed conifer	Breeding birds	0,1,4,35	Moderate	Y	Y	5 to 8 years post-treatment
Sullivan et al. 2005 A	British Columbia	Lodgepole pine	Mammals	2,1,0,3	Moderate	N (PCT)	N	12 to 14 years post-treatment
Sullivan et al. 2005 B	British Columbia	Lodgepole pine	Mammals	4,2,0,6	Heavy	N (PCT)	N	12 to 14 years post-treatment
Sullivan et al. 2007 A	British Columbia	Lodgepole pine	Mammals	0,0,0,3	Moderate	N (PCT)	N	12 to 15 years post-treatment
Sullivan et al. 2007 B	British Columbia	Lodgepole pine	Mammals	0,0,0,6	Heavy	N (PCT)	N	12 to 15 years post-treatment

(Continued on next page. See notes at end of table.)

Table 3.1 Continued

Study	Location	Forest Type	Taxa	Effect Sizes[a]	Thinning Intensity[b]	Commercial Thin?	Fuel Treatment Thin?	Data Collection
Suzuki 2001	Oregon	Hemlock/ Douglas fir	Amphibians & Mammals	0,2,0,14	Moderate	N (PCT)	N	1 to 2 years post-treatment
Suzuki and Hayes 2003	Oregon	Hemlock/ Douglas fir	Mammals	0,1,0,8	Light	N (PCT)	N	7 to 24 years post-treatment
Tibbels and Kurta 2003	Michigan	Red pine	Insects and Mammals	0,2,4,0	Moderate	Y	N	5 to 11 years post-treatment
Todd and Andrews 2008	South Carolina	Coastal pine plantation	Reptiles	0,0,0,2	Light	Y	N	0.5 to 2.5 years post-treatment
Twedt and Somershoe 2009	Louisiana	Bottomland hardwood	Breeding birds	0,0,0,14	Light	Y	N	1-12 years post-treatment
Wampler et al. 2008 A	New Mexico	Mixed conifer	Mammals	1,1,0,3	Light	N (understory)	Y	2 to 3 years post treatment
Wampler et al. 2008 B	New Mexico	Mixed conifer	Mammals	1,1,0,3	Moderate	Y	Y	2 to 3 years post treatment
Welsh et al. 1992	Massa-chusetts	Massachusetts oak	Breeding birds	0,0,3,9	Light	N (cordwood)	N	1 to 10 years post treatment
Wilson et al. 1995	Arkansas	Pine grassland	Breeding birds	1,0,0,23	Light	N (WSI)	Y	1 to 2 years post-treatment
Yi 2007 A	Oregon	Hemlock/ Douglas fir	Insects	2,0,19,0	Moderate	Y	N	4 to 5 years post-treatment
Yi 2007 B	Oregon	Hemlock/ Douglas fir	Insects	2,0,18,0	Heavy	Y	N	4 to 5 years post-treatment
Zebehazy et al. 2004 A	South Carolina	Southern pine	Breeding birds	1,1,0,11	Light	N (understory)	Y	1 to 2 years post-treatment
Zebehazy et al. 2004 B	South Carolina	Southern pine	Winter birds	1,1,0,7	Light	N (understory)	Y	1 to 2 years post-treatment

[a] Numbers indicate effect sizes for diversity/richness, total taxa abundance, guild abundance and species abundance respectively.
[b] Determined by the percent of unthinned (control) stand basal area or trees per hectare remaining in thinned (treatment) stands (Heavy: 0-33; Moderate: 34-66; Light: 67-100)

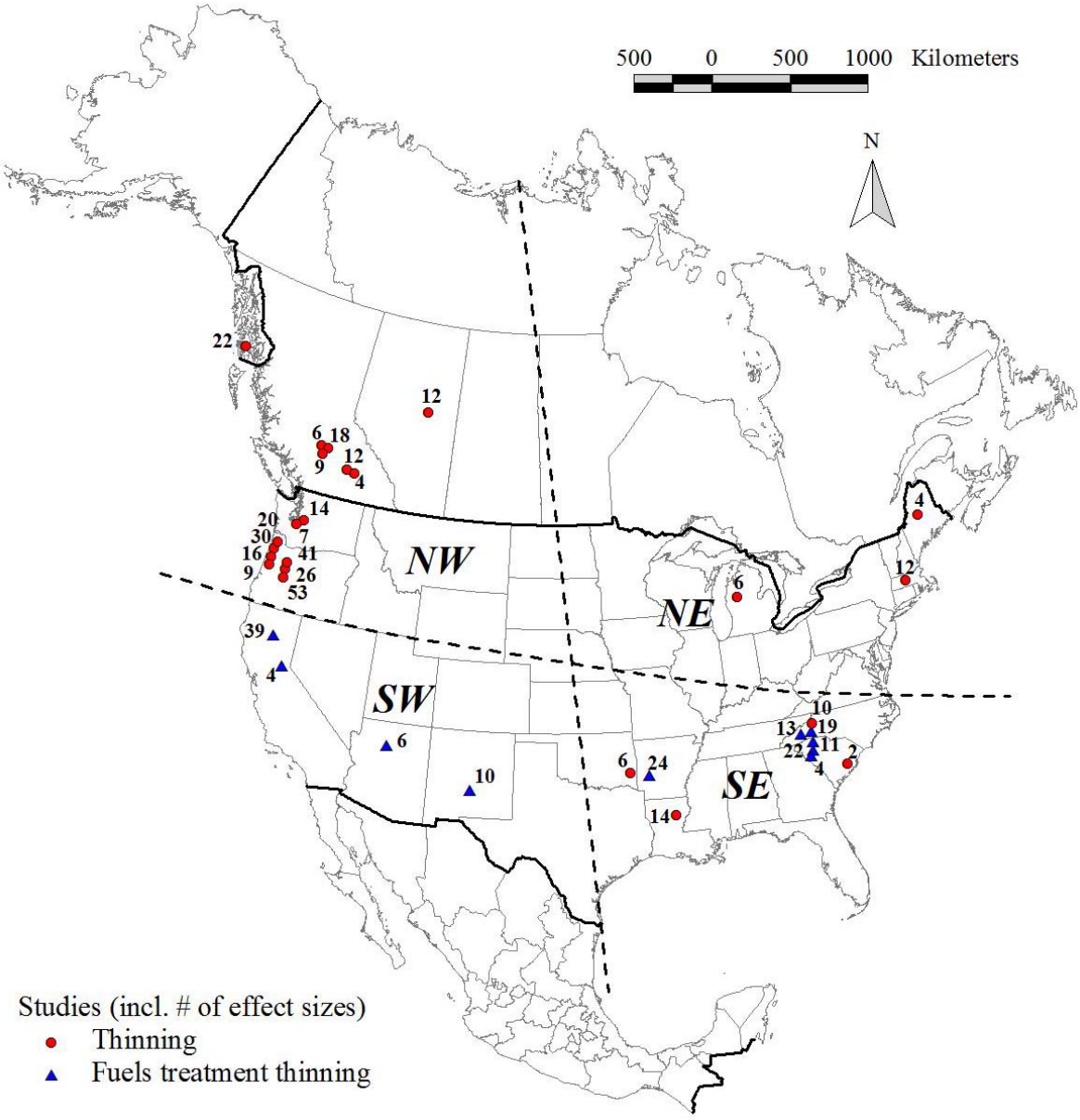

Figure 3.1 Distribution of Effects Sizes in North America

We conducted all meta-analyses using Meta-Win software (Rosenberg, Adams, and Gurevitch 2000). For each of the 505 responses, we calculated a response ratio which is the ratio of the experimental to control groups (Hedges, Gurevitch, and Curtis 1999). For each response, we treated non-thinned stands as the control. Thus, response ratios < 1.00 indicate a negative response to forest thinning, and ratios > 1.00 indicate a positive response to forest thinning. Because some means were zero, we added 1 to all means before calculating effect sizes. We used bootstrap confidence intervals and considered a combined effect to be significant if the confidence interval did not include 1.00. Some meta-analyses are based on multiple

effect sizes originating from only one or two studies. Following the suggestions of Borenstein et al. (2009:364), we report meta-analysis results in such situations but also provide limitations for its application.

3.3 Response of Birds to Forest Thinning

3.3.1 *Results of Breeding and Wintering Bird Species Meta-Analysis*

We found 274 bird responses (effect sizes) from 13 studies involving comparisons of thinned and unthinned forest stands with a significant cumulative effect size of 1.11 (Table 3.1, Figure 3.2). This was nearly double the number of effect sizes for the next most commonly reported taxon (mammals; n=149), but effect sizes came from fewer studies (k=13 for birds; k=17 for mammals). Despite the large number of effect sizes, most were for abundance measures of individual bird species, leaving a limited number of measures of taxa/guild abundance and diversity. The hairy woodpecker (*Picoides villosus*) was the most commonly reported individual species (k=6) with a significant cumulative effect size of 1.28.

Most effect sizes (52%) were from the Northwest (Table 3.2, Figure 3.1). However, studies from the Northwest represented a variety of forest types (e.g., lodgepole pine, aspen, coastal hemlock/fir, interior hemlock/cedar, and mixed conifer). Mean timing of data collection from the 13 studies was 3.81 (± 0.69) years post-thinning treatment. Response data were collected 1-15 years post-treatment, but only three studies (Hagar, McComb, and Emmingham 1996; Siegel and DeSante 2003; Twedt and Somershoe 2009) investigated response to thinning beyond five years post-treatment. Bird communities consistently responded favorably to forest thinning treatments overall (Figure 3.2). Across all regions and treatments, effects were significantly greater than 1.00 for all diversity and abundance measures except taxa/guild abundance (Table 3.2, Figure 3.2). Magnitudes of effects were generally comparable for birds and all other taxa (Figure 3.2). Studies from the Southwest reported significantly positive effects of thinning treatments on guild and species abundance but cumulative effect sizes for diversity measures were not significantly different from 1.00 (k=2). Thinning also proved to be a positive influence on bird species diversity measures in the Northwest, and bird taxa/guild abundance measures in the Southeast and Northeast (Table 3.2). Thinning intensity played a significant role in bird response. Birds responded favorably to light and moderate thinning (Table 3.2). Heavy thinning led to the only significantly negative responses for both taxa/guild abundance and the cumulative effect measure (Table 3.3). However, these results should be viewed with caution as they are based on only two studies from the Northwest region (Hagar Howlin, and Ganio 2004; Norton and Hannon 1997) where birds were measured 1-4 years post-treatment.

Breeding (k=13) and wintering (k=3) birds had similar responses to forest thinning both in magnitude and variation of effect sizes (Figure 3.3). Wintering birds were only represented by three studies (two in the Northwest and one in the Southeast). However, wintering bird diversity, taxa/guild, and cumulative effect sizes reported were significantly greater than 1.00 (Figure 3.3). Of thinning treatments included in analysis, fuels treatment thinning had the most favorable effect on bird species abundance and diversity (Figure 3.4). Neither precommercial nor commercial (non-fuels treatment) thinning effect sizes were significantly different from 1.00 (Figure 3.4).

Table 3.2 Summary of Effects of Forest Thinning on Biodiversity by Taxa and Region

	Northwest	Southwest	Southeast	Northeast	All Regions
Birds	**k=6**	**k=2**	**k=4**	**k=1**	**k=13**
Diversity	1.15 (n=6)**	0.90 *(n=1)*	1.08 (n=3)	--	1.12 (n=10)**
Taxa/guild abundance	0.98 (n=33)	1.48 (n=5)**	1.14 (n=2)**	1.20 (n=3)**	1.07 (n=43)
Species abundance	1.03 (n=104)	1.23 (n=39)**	1.08 (n=69)	1.04 (n=9)	1.12 (n=221)**
Cumulative	1.02 (n=143)	1.27 (n=45)**	1.08 (n=74)	1.08 (n=12)	1.11 (n=274)**
Mammals	**k=9**	**k=2**	**k=4**	**k=2**	**k=17**
Diversity	0.98 (n=8)	1.52 (n=2)**	--	--	1.06 (n=10)
Taxa/guild abundance	0.99 (n=9)	1.90 (n=2)**	2.56 (n=3)**	1.54 *(n=1)*	1.35 (n=15)**
Species abundance	1.03 (n=91)	1.69 (n=10)**	1.37 (n=19)**	1.91 (n=4)**	1.10 (n-124)**
Cumulative	1.02 (n=108)	1.67 (n=14)**	1.44 (n=22)**	1.86 (n=5)**	1.10 (n=149)**
Reptiles			**k=3**		**k=3**
Diversity			0.96 (n=1)		0.96 (n=1)
Taxa/guild abundance	--	--	0.90 (n=5)	--	0.90 (n=5)
Species abundance			1.58 (n=11)**		1.58 (n=11)**
Cumulative			1.38 (n=17)**		1.38 (n=17)**
Amphibians	**k=2**		**k=3**		**k=5**
Diversity	--		0.98 (n=2)		0.98 (n=2)
Taxa/guild abundance	0.96 *(n=1)*	--	0.61 (n=3)	--	0.95 (n=4)
Species abundance	1.19 (n=6)		0.87 (n=7)		0.94 (n=13)
Cumulative	1.15 (n=7)		0.92 (n=12)		0.94 (n=19)
Invertebrates	**k=1**			**k=1**	**k=2**
Diversity	1.11 (n=4)			--	1.11 (n=4)
Order/guild biomass	1.19 (n=37)**	--	--	1.02 (n=5)	1.09 (n=42)**
Species biomass	--			--	--
Cumulative	1.15 (n=41)**			1.02 (n=5)	1.10 (n=46)**
All Taxa Combined	**k=16**	**k=4**	**k=10**	**k= 3**	**k=33**
Diversity	1.07 (n=18)	1.34 (n=3)**	1.06 (n=6)	--	1.09 (n=27)**
Taxa /guild abundance	1.01 (n=80)	1.52 (n=7)**	1.42 (n=13)	1.06 (n=9)**	1.09 (n=109)
Species abundance	1.03 (n=201)	1.26 (n=49)**	1.10 (n=106)	1.16 (n=13)	1.09 (n=369)**
Cumulative	1.03 (n=299)	1.30 (n=59)**	1.11 (n=125)**	1.10 (n=22)**	1.09 (n=505)**

** Indicates bootstrap confidence intervals (1000 iterations) did not include 1.00; k = # of studies, n = # of effect sizes.

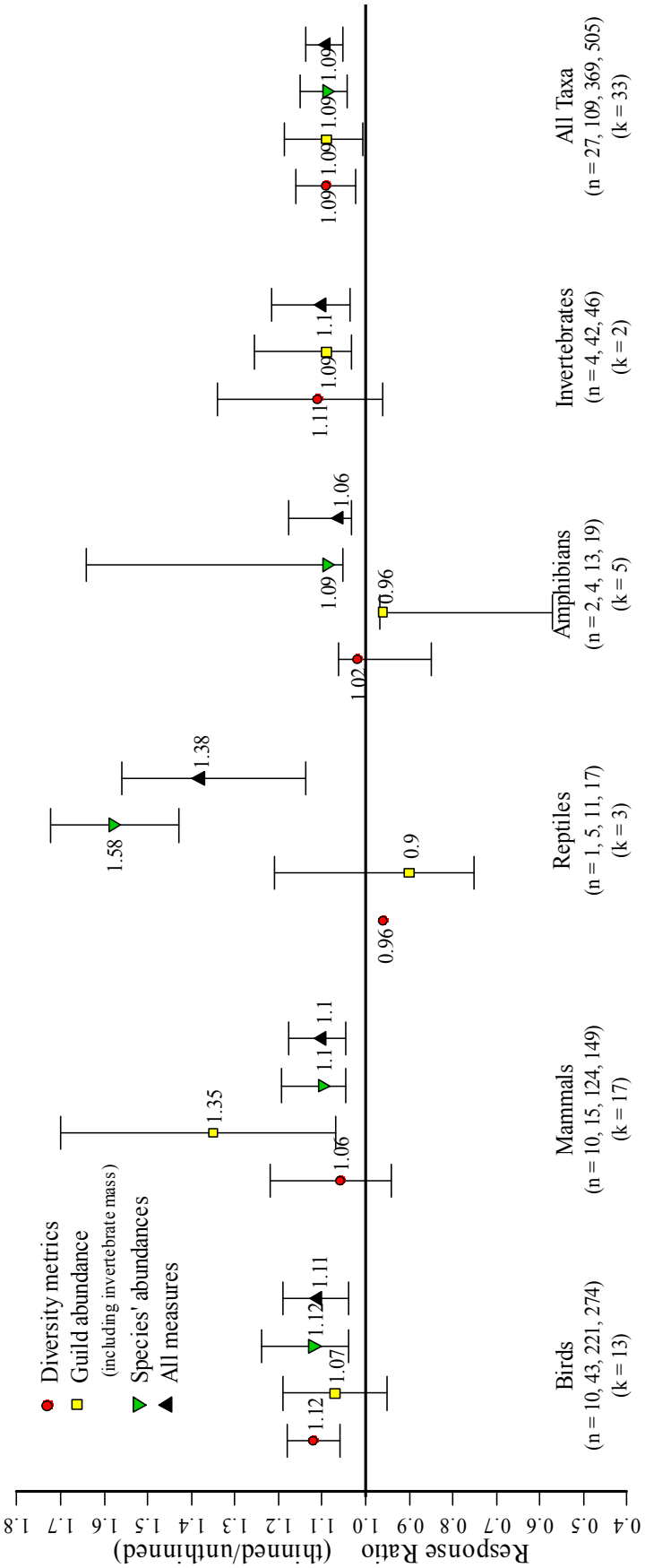

Figure 3.2 Summary Effect Sizes for Birds, Mammals, Amphibians and Invertebrates across All Manipulations [Means plotted without confidence intervals represent only one effect size (n=1).]

Table 3.3 Summary of Effects of Forest Thinning on Biodiversity by Taxa and Thinning Intensity

	Light Thin	Moderate Thin	Heavy Thin	All Treatments
Birds	**k=7**	**k=6**	**k=2**	**k=13**
Diversity	1.09 (n=5)**	1.13 (n=4)	1.37 (n=1)	1.12 (n=10)**
Taxa/guild abundance	1.13 (n=19)**	1.19 (n=17)**	0.72 (n=7)**	1.07 (n=43)
Species abundance	1.08 (n=104)	1.18 (n=92)**	0.87 (n=25)	1.12 (n=221)**
Cumulative	1.09 (n=128)**	1.18 (n=113)**	0.80 (n=33)**	1.11 (n=274)**
Mammals	**k=8**	**k=9**	**k=6**	**k=17**
Diversity	1.50 *(n=1)*	1.02 (n=4)	1.03 (n=5)	1.06 (n=10)
Taxa/guild abundance	1.93 (n=4)**	1.37 (n=7)**	0.93 (n=4)	1.35 (n=15)**
Species abundance	1.24 (n=37)	1.08 (n=46)**	1.08 (n=41)	1.10 (n=124)**
Cumulative	1.31 (n=42)**	1.08 (n=57)**	1.06 (n=50)	1.10 (n=149)**
Reptiles	**k=3**	**k=0**	**k=0**	**k=3**
Diversity	0.96 (n=1)			0.96 (n=1)
Taxa/guild abundance	0.90 (n=5)	--	--	0.90 (n=5)
Species abundance	1.58 (n=11)**			1.58 (n=11)**
Cumulative	1.38 (n=17)**			1.38 (n=17)**
Amphibians	**k=2**	**k=2**	**k=2**	**k=5**
Diversity	0.98 (n=2)	--	--	0.98 (n=2)
Taxa/guild abundance	0.61 (n=3)	*0.96 (n=1)*	--	0.95 (n=4)
Species abundance	2.26 (n=4)	1.17 (n=4)	0.77 (n=5)	0.94 (n=13)
Cumulative	1.07 (n=9)	1.13 (n=5)	0.77 (n=5)	0.94 (n=19)
Invertebrates	**k=0**	**k=2**	**k=1**	**k=2**
Diversity		0.96 (n=2)	1.22 (n=2)**	1.11 (n=4)
Order/guild biomass	--	1.06 (n=24)**	1.21 (n=18)**	1.09 (n=42)**
Species biomass		--	--	--
Cumulative		1.04 (n=26)	1.22 (n=20)**	1.10 (n=46)**
All Taxa Combined	**k=16**	**k=16**	**k=9**	**k=33**
Diversity	1.12 (n=9)**	1.06 (n=10)	1.11 (n=8)	1.09 (n=27)**
Taxa/guild abundance	1.16 (n=31)**	1.16 (n=49)**	0.85 (n=29)	1.09 (n=109)
Species abundance	1.09 (n=156)**	1.14 (n=142)**	1.02 (n=71)	1.09 (n=369)**
Cumulative	1.10 (n=196)**	1.13 (n=201)**	0.99 (n=108)	1.09 (n=505)**

NOTE: Thinning intensity is determined by the percent of unthinned (control) stand basal area or trees per hectare remaining in thinned (treatment) stands (Heavy: 0-33; Moderate: 34-66; Light: 67-100).
** Indicates bootstrap confidence intervals (1000 iterations) did not include 1.00; k = # of studies, n = # of effect sizes.

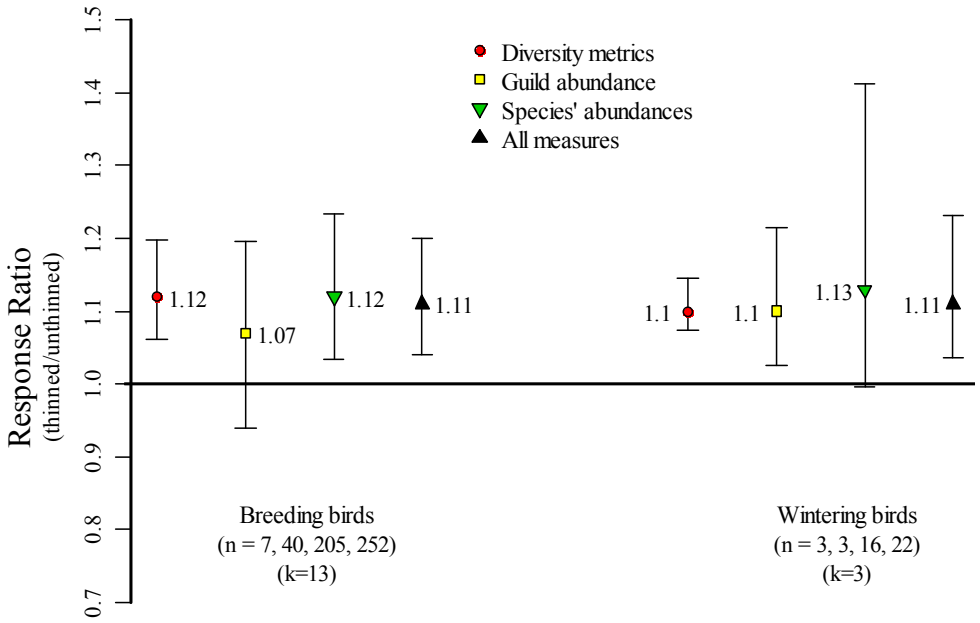

Figure 3.3 Summary Effect Sizes for Breeding and Wintering Birds across All Treatments

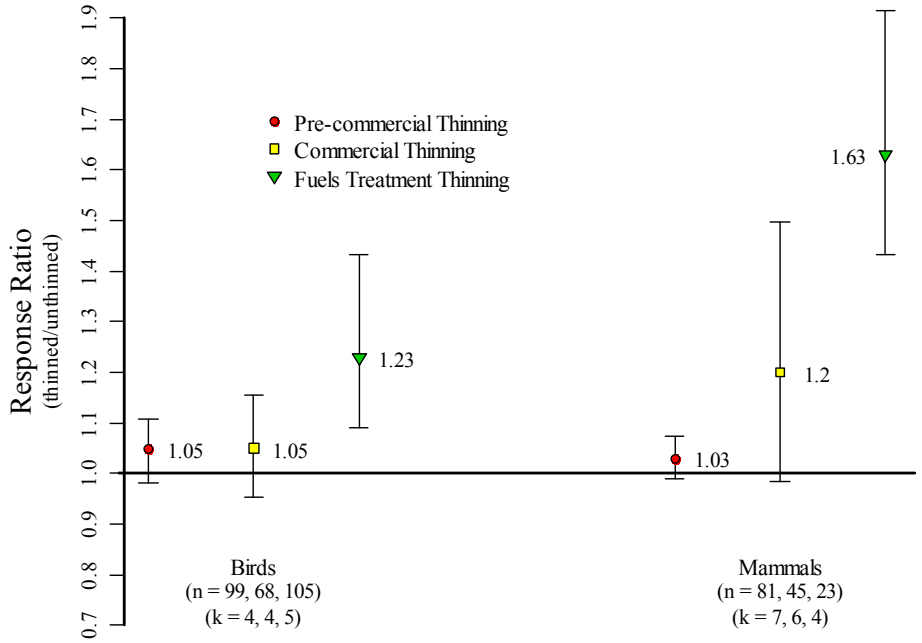

Figure 3.4 Bird and Mammal Effect Sizes for Pre-Commercial, Commercial and Fuels Treatment Thinning

3.3.2 *Discussion of Bird Response*

Positive responses by many bird species to forest thinning have been well documented (Hayes et al. 1997; Hunter 2001; Hagar, Howlin, and Ganio 2004; Hayes, Weikel, and Huso 2003; Kalies, Chambers, and Covington 2010). Proposed mechanisms for increased abundance and diversity of bird species in thinned stands include increased regeneration and development of shrub and understory layers from greater light access to the canopy floor (Hayes et al. 1997) or increased horizontal or vertical variation in forest structure (McComb and Noble 1980; Sullivan et al. 2002; Carey 2003). Others have proposed that thinning can cause a more rapid return to conditions simulating older seral stages which in turn can increase number of species using the diversified habitat (Barbour et al. 1997; Bailey and Tappeiner 1998).

Effect sizes significantly < 1.00 occurred only for studies where > 66% of basal area or trees per hectare were removed during thinning. Tree- and shrub-inhabiting birds may respond negatively to heavier thinning intensities (Norton and Hannon 1997) or certain treatments or forest types (Christian et al. 1996). However, duration of time between thinning treatment and measurement of avifauna may play a substantial role in negative responses observed (Hagar, Howlin, and Ganio 2004; Greenberg, Miller, and Waldrop 2007). Studies reporting negative avian responses to heavy intensity thinning observed bird species 1-2 years (Norton and Hannon 1997) and 1-4 years (Hagar, Howlin, and Ganio 2004) post-treatment. As a result, any negative responses may be due in part to the short-term nature of the survey effort. Most thinning operations will have an initial short-term negative effect on biodiversity due to understory disturbance caused by the operation itself (Hagar, Howlin, and Ganio 2004). Response of brown-headed nuthatches (*Sitta pusilla*) to thinning is immediate and positive, but other factors, such as number of snags, may ultimately determine their abundance (Wilson and Watts 1999). Although diversity measures may often increase with thinning, consideration needs to be given to species of high conservation priority that may be negatively affected, either directly or indirectly, by thinning (e.g., Swainson's warbler [*Limnothlypis swainsonii*], brown creeper [*Certhia Americana*]) (Hayes, Weikel, and Huso 2003; Bassett-Touchell and Stouffer 2006).

Across all methods of thinning harvest, fuels treatment thinning resulted in largest effect sizes for birds suggesting a strong positive response for avian species diversity and abundance. In stands thinned as a fuels treatment, Siegel and DeSante (2003) found canopy-, cavity-, and especially shrub-nesting avian species in higher abundance than in comparable unthinned stands. In drier forest types of the Southwest, removal of young conifer saplings and small trees in a fuels treatment thin resulted in redevelopment of shrub growth and elevated densities of birds (Siegel and DeSante 2003). In the Southeast, response to fuels treatment thinning appears to be influenced by treatment intensity, whether thinning is followed with a burn, and the guild of birds species investigated (Greenberg, Miller, and Waldrop 2007; Zebehazy, Lanham, and Waldrop 2004).

3.4 Response of Mammals to Forest Thinning

3.4.1 *Results of Mammalian Species Meta-Analysis*

We found 149 mammal responses (effect sizes) from 17 studies involving comparisons of thinned and unthinned forest stands (Table 3.1). There were more published mammal studies but fewer individual effect sizes than for birds. Measured effects came primarily from studies of small mammals (k=14; n=129) but also included large herbivores (Sullivan et al. 2007) and bats (Tibbels and Kurta 2003; Loeb and Waldrop 2008). The deer mouse was the most frequently reported individual species (k=12), with a significant cumulative effect size of 1.52.

Studies were available from all regions for the cumulative meta-analyses. Average timing of data collection from the 17 studies was 5.93 (± 1.48) years post-thinning treatment. Number of effect size measures of mammalian taxa/guild abundance and diversity were somewhat limited, thus limiting conclusions based upon meta-analysis. Mammalian diversity and abundance were higher in thinned stands than unthinned controls across most regions (Table 3.2). However, magnitude of mammalian response to thinning treatments varied significantly between regions. Most studies were in the Northwest (72%; k=9; Figure 3.1), where there was no significant mammalian species abundance or diversity response to thinning treatments. All other regions, however, reported summary effects significantly greater than 1.00, suggesting a strong positive response of mammalian diversity and abundance to the variety of thinning treatments applied.

There was little difference in mammalian response by thinning intensity (Table 3.3). However, there was a gradual decrease in summary effect sizes reported (all were greater than 1.00) from light through heavy thinning intensities, with the latter being not significantly different from 1.00. Effect sizes for all three treatment types were above 1.00. Response of mammals to fuels treatment thinning was significantly greater than 1.00 and it was also significantly greater than measured mammalian response to precommercial thinning (Figure 3.4).

3.4.2 *Discussion of Mammal Response*

Numerous studies have revealed a positive response of small mammals to forest thinning (Zwolak 2009). Thinning is proposed to be beneficial to open-habitat and generalist small mammal species through increased light to and productivity of understory vegetation. Increased understory shrub and herbaceous vegetation increases forage and cover for deer mouse **(***Peromyscus maniculatus***),** jumping mouse (*Zapus hudsonius*), and most vole species (Wilson and Carey 2000; Suzuki and Hayes 2003; Homyack, Harrison, and Krohn 2005), although response to this increase may be short-lived (Suzuki and Hayes 2003).

In ponderosa pine forests of the Southwest (2-3 years post treatment), species responses to thinning treatments varied (Converse, Block, and White 2006), but total small mammal density was higher in thinned stands (Converse, White, and Block 2006). Bats are also typically favored by thinning operations across geographies through increased access to flying insects (Humes, Hayes, and Collopy 1999; Tibbels and Kurta 2003; Loeb and Waldrop 2008), but species-specific responses must be considered (Patriquin and Barclay 2003). Thinning also often leads to no change in or increased densities of common small mammal species (Homyack, Harrison, and Krohn 2005). However, relatively little is known about influences of thinning intensity on response of small or large mammal populations (Suzuki and Hayes 2003).

Although commercial thinning resulting in open canopies and increased understory growth may favor measures of mammalian species abundance or diversity, it may not improve habitat conditions for species associated with closed-canopy conditions (Lehmkuhl, Loggers, and Creighton 2002). However, intermediate or variable density treatments may produce habitat for generalists and closed canopy or arboreal specialists (Carey and Wilson 2001; Lehmkuhl, Loggers, and Creighton 2002; Carey 2003). As an example, Ransome et al. (2004) reported lowest abundance of both northern flying squirrels (*Glaucomys sabrinus*) and American red squirrels (*Tamiasciurus hudsonicus*) in heavily thinned and unthinned lodgepole pine forests, whereas highest abundances were recorded in a moderate thinning treatment. Although typically associated with low intensity harvest, precommercial thinning has been shown to reduce small mammal species diversity in some instances (Etcheverry, Ouellet, and Crête 2005). However, pre-commercial thinning can lead to late-seral conditions developing at an earlier age, which may ultimately benefit species associated with older forests.

The thinning operation itself changes understory characteristics (e.g., prey availability, vegetative cover, and microclimate) that are linked to demographic parameters of many small mammals. As a result, thinning can initially have significant short-term effects on abundance and diversity of small mammals (both positive and negative) that do not persist (Greenberg, Neary, and Harris 2006; Greenberg, Miller, and Wallop 2007). Conversely, Garman (2001) reported a short-term influx of small mammals in thinned stands that disappeared beyond three years post-harvest. Our meta-analysis results confirm reported positive medium- and long-term response to forest thinning by mammals across all forest types and thinning intensities. Despite the generally positive response by mammals to forest thinning, some direct and indirect effects of forest thinning on species of conservation concern may warrant further review (e.g., northern flying squirrel (*Glaucomys sabrinus*) habitat connectivity and food resources [Carey 2000; Gomez, Anthony, and Hayes 2005] and snowshoe hare/Canada lynx (*Lepus americanus*/*Lynx canadensis*) population dynamics [Griffin and Mills 2007; Hodges 2000a, 2000b; Homyack, Harrison, and Krohn 2007]).

3.5 Response of Reptiles to Forest Thinning

3.5.1 *Results of Reptilian Species Meta-Analysis*

We found 17 reptile responses (effect sizes) from three studies in the southeastern US (Table 3.1) (Kilpatrick et al. 2004; Todd and Andrews 2008; Matthews et al. 2010). Cumulative effect size for 11 species abundance, 5 taxa/guild abundance and 1 diversity measure was 1.38 (Table 3.2). Twelve of 17 individual effect sizes reported were greater than 1.00.

3.5.2 *Discussion of Reptile Response*

Many reptile populations are potentially experiencing declines (Gibbon et al. 2000). However, research documenting response of reptiles to timber harvest is limited (Russell et al. 2004; Todd and Andrews 2008). Solar radiation and thermal cover are important habitat characteristics for reptiles (Kiester 1971). Standard clearcutting provides ample solar radiation for morning sunning, but may not provide adequate night thermal cover in some regions. Thinning, on the other hand, may provide a more moderate environment for many reptile species than closed-canopy forest stands or recently clearcut stands (Todd and Andrews 2008). Many lizard species, some of which have been reported in decline, have been shown to be in higher abundance on recently harvested stands (Greenberg, Neary, and Harris 1994; Kilpatrick et al. 2004).

The highest species richness and abundance of North American herpetofauna are in the southeastern US (Kiester 1971). The indication from available research (primarily from that region) is that forest harvest can variously affect reptile species depending on their life histories (Renken et al. 2004). However, more research would be required to draw conclusions about response to different thinning intensities and regional differences in reptile response to various thinning treatments.

3.6 Response of Amphibians to Forest Thinning

3.6.1 *Results of Amphibian Species Meta-Analysis*

We found 19 amphibian responses from five studies, two in the northwestern US (Garman 2001; Suzuki 2001), and three in the southeastern US (Table 3.1) (Ford et al. 2000; Kilpatrick et al. 2004; Matthews et al. 2010). Cumulative effect size for all abundance and diversity measures was 0.94, but was not significantly different from 1.00 (Table 3.2). However, the cumulative taxa/guild abundance effect size (0.95) was significantly less than 1.00 (Table 32). Suzuki (2001) and Matthews et al. (2010) reported lower total amphibian abundance in thinned stands than unthinned stands. Twelve of 19 reported effects were less than 1.00.

Fowler's toad (*Bufo fowleri*), eastern narrow-mouthed toad (*Engystoma carolinense*) (Kilpatrick et al. 2004) and Ensatina (*Ensatina eschscholtzii*) (Garman 2001) had the strongest positive responses to forest thinning. The Ensatina (incl. all subspecies) was the only species measured in different thinning intensities and studies (Garman 2001; Suzuki 2001). Suzuki (2001) found slightly lower abundances of Ensatina in thinned stands than unthinned control stands 1-2 years post-harvest. Conversely, Garman (2001) measured Ensatina response to thinning treatments 5-8 years post-harvest and found a strong positive effect of moderate and heavy thinning intensities.

3.6.2 *Discussion of Amphibian Response*

Salamanders represented 11 of 19 effect sizes used to summarize amphibian response to thinning. Salamanders, particularly plethodontid salamanders, are often more abundant in closed-canopy forests and later successional stages (Corn and Bury 1989; Ash 1997; Aubry 2000; Semlitsch et al. 2009). Declines of up to 80% for some salamanders and species richness declines of up to 50% have been reported following even-age timber harvest in some forest types (Petranka, Eldridge, and Haley 1993). In a comprehensive review of amphibian response to forest management in North America, deMaynadier and Hunter (1995) report that short-term, stand-level response of salamanders to timber harvest is typically negative, especially for clearcutting, usually through the mechanisms of reduced leaf litter, canopy cover, and soil moisture (deMaynadier and Hunter 1995; Pough et al. 1987; Ash 1997; Semlitsch et al. 2009). Pough et al. (1987) showed a strong linear relationship of understory vegetation and leaf litter depth with above-ground salamander activity, and Ash (1997) reports the timing of amphibian return to previously harvested stands closely follows re-development of the litter layer.

Less information is available on amphibian response to partial harvest or thinning. Some suggest that detrimental effects of stand disturbance (e.g., soil compaction, stream sedimentation) on amphibian populations persist even when the disturbance is a less severe partial cut (Harpole and Haas 1999; Semlitsch et al. 2009). However, Brooks and Kyker-Snowman (2008) found forest floor temperature and humidity to be similar between partial, selection-based timber harvests and unharvested control stands. Several studies report mixed or even positive effects of thinning on amphibian populations (Pough et al. 1987; Grialou, West, and Wilkins 2000; Renken et al. 2004; McKenny, Keeton, and Donovan 2006) suggesting that thinning harvests can maintain forest amphibian populations. In a study comparing thinning with riparian buffers to unharvested control stands, Kluber, Olson, and Puettmann (2008) found no treatment effect across several species of amphibians. The less extreme response to thinning (when compared to even-aged regeneration harvests) by the understory and forest litter layers may explain the more moderate, short-term amphibian response to thinning when compared to even-age regeneration harvests (Petranka, Eldridge, and Haley 1993). Enhanced productivity of herbaceous and shrub forest understory also can create favorable soil moisture conditions for amphibian species (Zheng et al. 2000).

Although an increasing number of studies in managed forest settings have focused on amphibians (e.g., Russell et al. 2004; Kroll 2009), we found few reporting experimental results needed for meta-analyses. Nevertheless, the studies we evaluated, and the lack of significant response, suggest the biophysical characteristics necessary for moisture sensitive amphibian species may still be retained in thinned forests (Ford et al. 2000).

3.7 Response of Invertebrates to Forest Thinning

3.7.1 *Results of Invertebrate Species Meta-Analysis*

We found 46 invertebrate responses (effect sizes) from two studies, one in the northwestern US (Yi 2007), and another in the upper midwestern US (Table 3.1) (Tibbels and Kurta 2003). Thinned stands reported significantly higher biomass of invertebrates than unthinned stands for 35 of 42 order biomass effect sizes. Cumulative effect size of 1.10 for 42 order biomass measures and four measures of order

diversity was significantly greater than 1.00 (Table 3.2). Response magnitude was similar for both studies.

3.7.2 *Discussion of Invertebrate Response*

Insects are affected in a variety of ways by changes to the forest canopy, understory, and litter layers, and can themselves be significant drivers of forest productivity and nutrient cycling (Hunter 2002). Diversity of arthropod functional groups can be a good measure of overall habitat complexity (Hunter 2002; Yi 2007). However, effects of forest thinning on invertebrates are not well understood (Duguay, Wood, and Miller 2000; Schowalter, Zhang, and Rykken 2003; Yi 2007). Mechanisms for increases or declines of certain invertebrates in response to forest thinning are often specific to the functional group being examined. Some examples include increases in abundance of herbivorous arthropods in recently thinned stands due to increased availability of canopy level forage and declines in populations of detritovores and some predators due to reduced habitat and food resources (Progar, Schowalter, and Work 1999). Thinning that changes community composition and structure of understory vegetation can increase diversity and abundance of some insect groups in the short term (Taki et al. 2010).

Depending on their life history characteristics, invertebrate communities have been shown to respond positively (Yi 2007), negatively (Niemela, Langor, and Spence 1993), or minimally (Schowalter, Zhang, and Rykken 2003; Apigian, Dahlsten, and Stephens 2006) to forest thinning and other canopy-opening disturbances. The research we summarized, including responses for several different types of arthropods (i.e., herbivores, predators, detritovores), demonstrated a significant positive summary response to forest thinning treatments. However, our results are limited in geographic scope and in number of reporting studies.

3.8 Fuels Treatment Thinning

Excessive fuel loading in stand understories has become a significant issue especially in regions with historically frequent fire return intervals that have been altered through suppression (Waldrop et al. 2004). As a result, thinning to reduce fuel loads will likely continue to occur especially throughout forests in the western U.S. (USDA Forest Service 2005; Evans and Finkral 2009). Fuels treatment thinning is distinct from other types of thinning not only in its selective regional application and limited potential for economic gain but in the intensity of disturbance. For that reason, we offer a separate discussion of response of diversity and species abundance to fuels treatment thinning.

Thinning intensity in seven of 10 fuels treatment studies was considered light (0-33% of basal area removed). The remainder were of moderate intensity (34- 66% basal area removed). Our results show that studies of fuels treatment thinning had significantly higher taxa/guild abundance and cumulative effect sizes than non-fuels treatment thinning experiments (Figure 3.5). Across 152 effect sizes of fuels treatment thinning, abundance and diversity measures were higher in fuels treatment thinned stands than unthinned (control) stands (Figure 3.5). Proposed mechanisms for this substantial increase in abundance and diversity are similar to those for precommercial thinning and include increases in forest productivity, reduced competitive dominance, and redevelopment of the understory shrub and herbaceous layers. The differences in magnitude of response for fuels treatment versus precommercial thinning may be due to forest type and regional differences as much as the treatment itself.

Some concerns with widespread application of fuels treatment thinning remain. Mechanical fuels treatment thinning will typically result in fewer snags than a prescribed burn or thin/burn treatment (Greenberg, Miller, and Waldrop 2007). In addition, soil compaction from increased stand entries could occur, although reducing number of skid trails would likely reduce potential for this impact (Moghaddas and Stephens 2008). Our analysis and other research (Converse, Block, and White 2006; Converse, White, and Block 2006) suggest at the very minimum, short-term gains for total mammal abundance, species diversity and forest health after fuels treatment thinning.

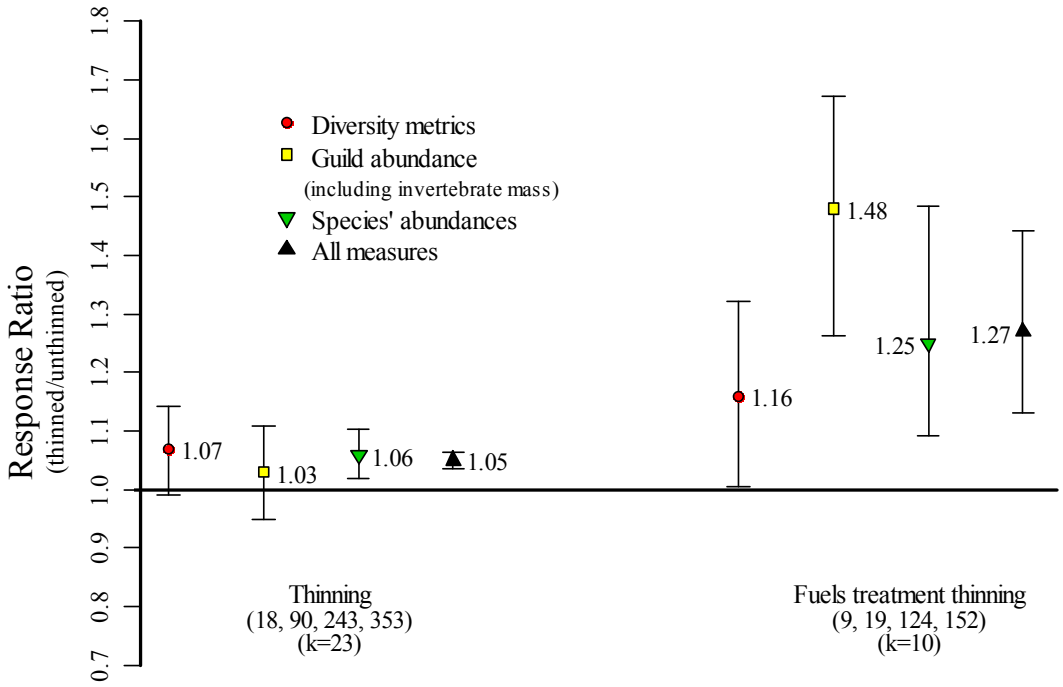

Figure 3.5 Summary Effect Sizes for Standard Thinning and Fuels Treatment Thinning across All Taxa

3.9 Response of Plant Species Diversity to Forest Thinning

Response of plant species diversity to forest thinning is often positive, but has been less studied than faunal diversity (Halpern and Spies 1995; Thomas et al. 1999). In the northwestern US and Canada, species richness of understory vegetation in thinned stands was similar to (Deal 2001) or greater than (Thomas et al. 1999) uncut control stands. In structurally complex temperate rain forests of the northwestern US, thinning increased growth of important mid-canopy layers (Comfort, Roberts, and Harrington 2010). Lodgepole pine forests of the Northwest Interior exhibited few differences in plant species diversity or composition between thinned and unthinned stands (Sullivan et al. 2002). In boreal forests, the peak plant species richness occurred in early seral stages. As forest succession continued, pre-commercial thinning sustained high levels of plant diversity (Weidenfalk and Weslien 2009).

Plant species richness in ponderosa pine forests of the southwestern US was least in unmanaged stands and increased with greater thinning intensity (Griffis et al. 2001). However, exotic species were a large part of the increase in richness for harvested stands, and number of native shrub species decreased significantly with treatment intensity (Griffis et al. 2001). In Sierran mixed conifer forests, canopy closure, used as a measure of thinning intensity, was shown to be negatively related to plant species richness (Battles et al. 2001). In addition, plant species composition varied significantly with intensity of thinning treatments. High intensity treatments maximized species richness but understory vegetation typical of late seral stands was more abundant in lightly thinned or control stands. Furthermore, control stands had lower proportions of exotic species (Battles et al. 2001). By three years post-treatment, thinned

stands showed significantly higher plant species richness than control stands in the Piedmont of South Carolina (Phillips and Waldrop 2008).

In summary, response of plant species diversity to forest thinning across much of North America is likely to be positive, but will differ by forest type and treatment intensity. Plant species composition and abundance of exotic species are also likely to vary with thinning intensity.

3.10 Summary of Biodiversity Effects and Management Implications

Though harvesting live trees for biofuels production as part of a sustainable forest management program disturbs ecological processes to some extent, such disturbances do not negatively affect biological diversity in most cases (Janowiak and Webster 2010). Our results show that, across most thinning intensities and forest types, thinning positively affects abundance and diversity of a variety of taxa. The magnitude of response to forest thinning, either positive or negative, is often small. It is important to recognize that some species of higher conservation concern may be either positively or negatively affected by thinning and that simple diversity and richness measures may not be sufficient for fully understanding effects of thinning on biodiversity. Furthermore, thinning (as with any silvicultural practice) is not implemented simultaneously across the landscape. As a result, biomass thinning harvest across a range of intensities will likely result in increased species abundance and diversity in most forest types.

Disturbance can increase species diversity at stand and landscape scales by creating a variety of habitat types through a mosaic of forest development stages (Hunter 1999; Franklin et al. 2002; Loehle et al. 2002; Lindenmayer, Franklin, and Fischer 2006). However, species response to disturbance can depend on biophysical setting of the landscape (McWethy, Hansen, Verschuyl 2010). In highly productive systems with lengthy inter-disturbance periods, a few species can begin to dominate the community, leading to reduced levels of diversity (Huston 1999, 2004; Odion and Sarr 2007). Forest thinning for biofuels production in highly productive forests may provide disturbance necessary to counteract competitive dominance. Alternately, in less productive forests, more care may be required to blend objectives for biomass harvest with those for maintenance of biological diversity (Janowiak and Webster 2010; Page-Dumroese, Jurgensen, and Terry 2010). Disturbance intensity and biophysical setting are likely to be strong determinants of response by wildlife and vegetation to biomass thinning harvests (Greenberg, Miller, and Waldrop 2007). Thinning designed to promote species diversity will likely need locally tailored prescriptions of intensity and pattern (Hagar, Howlin, and Ganio 2004).

Forested regions of North America harbor a significant proportion of terrestrial biodiversity (Hansen and Rotella 1999). Much of this land is privately owned and is under increasing pressure from rural residential development (Huston 2005; Gude et al. 2006). Thinning for biofuel production may offer land managers an additional economic incentive to retain their ownership in forest cover and the opportunity to address other silvicultural and ownership objectives (Page-Dumroese, Jurgensen, and Terry 2010). However, stand accessibility, terrain, transportation costs, and availability of processing plants are just a few of the factors that will influence long-term viability of thinning for biofuels production (USDA Forest Service 2005).

3.11 Geographic Limitations, Empirical Knowledge Gaps and Research Needs

The meta-analysis we completed for reptiles and amphibians suffers from a limited number of studies and non-uniform geographic distribution of results. Few meaningful conclusions should be drawn from the meta-analysis of reptile responses originating from only three published studies in the Southeast. Many of the responses included in this analysis were of abundance measures for single species. However, abundance is not always related to habitat quality and may only reflect short-term occupancy of the sampled stand (Van Horne 1983). Furthermore, as a result of common species being disproportionately

included due to availability of data, analysis of single species responses may or may not provide an accurate picture of biodiversity response (Lennon et al. 2004; Prendergast et al. 1993).

4.0 SHORT-ROTATION WOODY CROPS

4.1 Background and Definitions

Short-rotation woody cropping (SRWC) refers to silvicultural systems designed to produce woody biomass using short harvest cycles (1-15 years), intensive silvicultural techniques (e.g., fertilization, irrigation, and competition control), genetically improved planting material, and often coppice regeneration (Dickmann 2006). Many species used for SRWC re-sprout vigorously after harvest. In North America, *Populus sp.* (poplars and cottonwood) and willow (*Salix* spp.) currently show promise, and most research on both production and biodiversity response to SRWCs has focused on these two species. Other candidate SRWC species include loblolly pine (*Pinus taeda*), alder (*Alnus* sp.), black locust (*Robinia pseudoacacia*), silver maple (*Acer saccharinum*), sycamore (*Platanus occidentalis*), *Eucalyptus* spp., and sweetgum (*Liquidambar styraciflua*), although much less is known about their biofuel potential and relationship to biodiversity (Philips et al. 1995; Dickmann 2006). Short-rotation woody crop systems may be profitable in the southeastern, midwestern, Pacific Northwest and boreal regions of North America (Weih 2004; Dickmann 2006; Dale et al. 2010).

Using SRWCs for bioenergy is not a new concept (e.g., Christian et al. 1994), but fluctuations in fossil fuel prices, social pressure to develop alternate energy sources, government incentives, and emerging carbon markets have renewed interest in SRWCs on privately owned and intensively managed forests of North America. Biodiversity response to SRWCs will depend in large part on what type of habitat is replaced, landscape context, and scale of analysis (Dale et al. 2010). For example, SRWCs that replace cropland will likely increase stand-level diversity, but if SRWCs replace mature forests, native grasslands or other habitat types with high local diversity, then stand-level diversity will likely decrease (Christian et al. 1994; Fletcher et al. 2011). Biodiversity at larger spatial scales might respond differently to these same practices based on scale of analysis and characteristics of the landscape analyzed.

Bioenergy feedstocks may be derived from forest management systems that vary in their intensity in terms of silvicultural inputs and implications for ecosystems (Burger 2002). In this review, our objective was to examine potential stand-level biodiversity response to conversion of existing, intensively managed forests to SRWCs. We leave discussion of how landscape-level conversions may influence biodiversity to others (e.g., Fletcher et al. 2011). Our analysis will be restricted to intensively managed forests in North America, with a focus on *Populus* species because information about biodiversity response to other species is lacking in North America.

4.2 Methodology

We reviewed the literature for papers that compared biodiversity responses between plantations of short-rotation woody species (e.g., cottonwood, poplar) and other forest types. Response variables included diversity metrics (i.e., species richness, diversity or evenness), total abundance, abundance of taxa or groups of species (guilds), and abundance of individual species. We used Wildlife & Ecology Worldwide, USDA Forest Service TreeSearch and Google Scholar databases to search for relevant studies. We searched for the following forestry and biodiversity terms in article abstracts: willow, eucalyptus, poplar, cottonwood, short-rotation woody crops, biofuel plantation, biodiversity, diversity, richness, wildlife, birds, avian, amphibians, reptiles, invertebrates, insects, and mammals. We supplemented searches by examining bibliographies of articles for additional references.

Because responses can vary greatly among taxa and among species within taxa, we considered different biodiversity measures (e.g., diversity, abundance, richness) from the same study to be independent effects (Bender, Contreras, and Fahrig 1998). For birds, we also considered studies that presented analysis of breeding and winter bird response as separate "studies" because behavior, habitat use, and composition of bird communities can be very different during those two seasons. When studies presented comparisons for a metric in consecutive years, we calculated overall mean effect and standard deviation using pooled variance.

We conducted all meta-analyses using Meta-Win software (Rosenberg, Adams, and Gurevitch 2000). For each response, we calculated a response ratio which is the ratio of the experimental to control groups (Hedges, Gurevitch, and Curtis 1999). For each response, we coded the SRWC plantation mean treatment as the experimental group. For example, if richness was higher on plantations compared to the "control" comparison (e.g., bottomland hardwoods), response ratios would be > 1.00. Ratios < 1.00 indicate a negative response to SRWCs. Because some species means were zero, we added 1 to all species means before calculating effect sizes. We used bootstrap confidence intervals and considered a combined effect to be significant if the confidence interval did not include 1.00.

Because each study compared different-aged plantations to different forest types (sometimes very different), each comparison represented a unique set of conditions. Thus, we did not want to merely calculate overall effect sizes. Also, some studies compared plantation metrics to more than one additional forest type, so these effect sizes would not be independent because they included data from the same control sites. Thus, we calculated separate effect sizes for each unique plantation (age) vs. reference forest type comparisons for bird metrics only (only one study produced mammal effect sizes). We used fixed effects models for these subgroups because they were comprised of effect sizes from a single study. Also, some studies produced estimates of species richness that were corrected for area sampled (e.g., rarefaction; Christian et al. 1997) while others did not provide information about area sampled. Thus, overall effects sizes likely include much variation and more focused meta-analyses are warranted. Although some of these sub-groups have small sample sizes (and come from one paper), summary effect sizes are still better than vote counting or other forms of summary methods (Borenstein et al. 2009).

4.3 Response of Birds to Short-Rotation Woody Crops

We found 243 effect sizes for birds from seven studies (Figure 4.1, Table 4.1). Cumulative effect size estimates indicated diversity and abundance were lower on SRWC plantations compared to reference woodlands for birds, while abundance of individual species was more variable and not consistently higher or lower on SRWC plantations. Unfortunately, none of these studies directly compared SRWCs to traditional forest plantations, so we cannot predict whether increasing SRWC in these systems will increase or decrease overall forest biodiversity. However, we analyzed comparisons of SRWCs with other forest types, and this provided some evidence for preliminary predictions and guidance for future research. Although these seven studies spanned 26 years, they are still relevant because density and rotation recommendations for biomass applications of SRWC have not substantially changed (Dickmann 2006).

Species responses were highly variable, and confidence intervals for response ratios of species abundances always included 1.00 with one exception (6–9 year cottonwood plantations compared to bottomland hardwood; Table 4.2, Figure 4.2). Abundance of individual species as a group was not different from 1.00, because some species were less abundant on SRWC plantations (compared to forested lands), while others were more abundant on plantations. We further investigated species responses by calculating effect sizes for all species with ≥ 2 individual effect sizes. Seven species were statistically more abundant on SRWC plantations (95% confidence interval did not include 1.00), and seven species were statistically more abundant on reference forests (Table 4.3). Species that were more abundant on SRWC plantations were species commonly associated with dense, shrubby habitat structure

(ruby-throated hummingbird [*Archilochus colubris*], warbling vireo [*Vireo gilvus*], yellow-breasted chat [*Icteria virens*], eastern towhee [*Pipilo erythrophthalmus*], indigo bunting [*Passerina cyanea*], orchard oriole [*Icterus spurius*] and American goldfinch [*Spinus tristis*]). Species associated with reference forests tended to be mature forest associates and/or cavity-nesters (red-headed woodpecker [*Melanerpes erythrocephalus*], red-bellied woodpecker [*Melanerpes carolinus*], pileated woodpecker [*Dryocopus pileatus*], white-eyed vireo [*Vireo griseus*], American robin [*Turdus migratorius*], brown thrasher [*Toxostoma rufum*] and northern parula [*Parula americana*]). They likely responded to lack of vertical structure and canopy in SRWC plantations, especially at earlier stages of growth. Because SRWC plantations are typically harvested at early ages, they lack large stems that provide cavities for hole-nesting birds. Thus, it is not surprising that nearly 50% of species associated with reference woodlands were cavity-nesters. Because cavity-nesting birds may nest only at very low densities in SRWC plantations (Twedt et al. 2001), widespread adoption of SRWC could reduce stand-level density of cavity-nesting species. However, nest site availability could be increased by using supplemental nest boxes (Twedt and Henne-Kerr 2001), retaining some large stems in SRWC harvest units, or emphasizing nest sites in less intensively managed portions of the landscape.

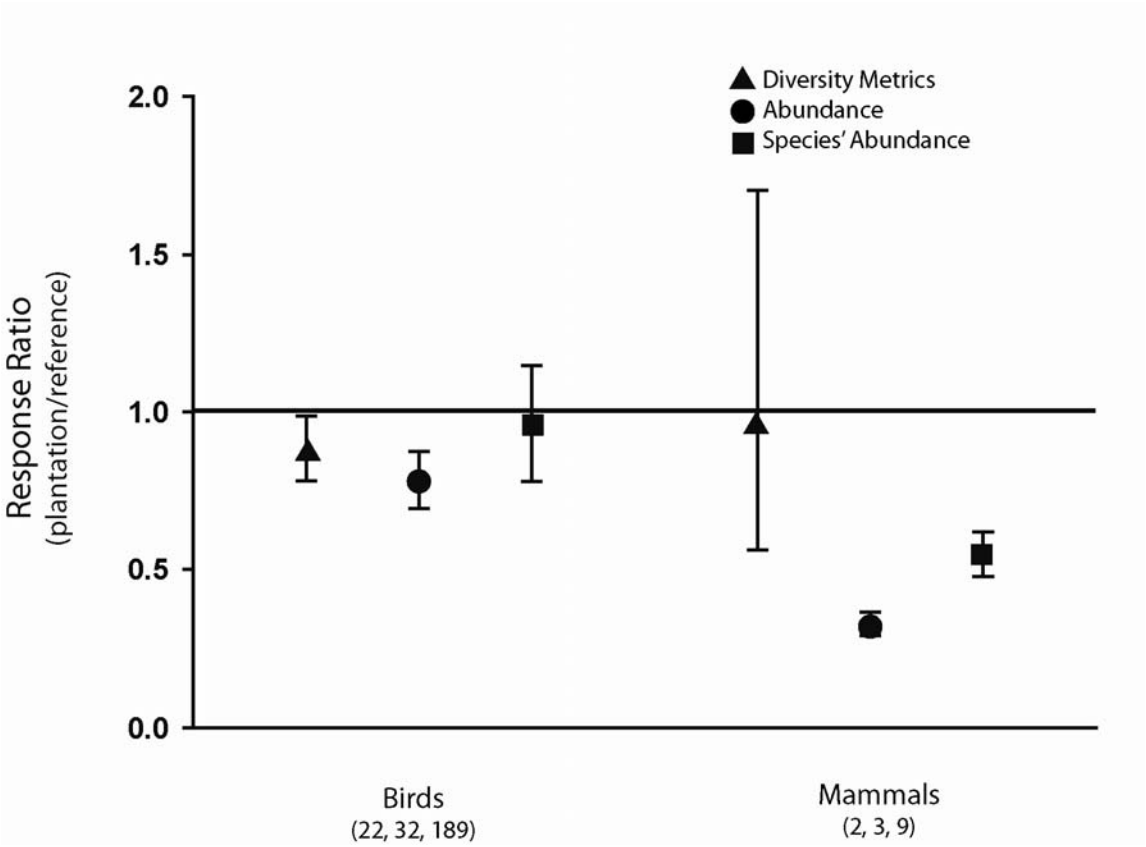

Figure 4.1 Summary Effect Sizes for Birds and Mammals across All Comparisons of Poplar and Cottonwood Plantations to Reference Forests

Table 4.1 Summary of Manipulative Studies Used in Meta-Analysis

Study	Location	SRWC	Reference Forest Habitat	Taxa	Effect Sizes[a]
Hanowski et al. 1997	MN, WI, SD	1–8 yr-old Hybrid Poplar	Surrounding Forestlands	Birds	1, 7, 7
Christian et al. 1997	MN, WI, SD	1–8 yr-old Hybrid Poplar	Surrounding Forestlands	Birds Mammals	4, 16, 0 1, 3, 8
Twedt et al. 2002	MS/LA	2–10 yr-old Cottonwood	2 – 10 yr old oak reforestation	Birds	2, 1, 40
Tomlinson 1977	MS	3–6 yr-old Cottonwood	Bottomland Hardwood	Birds	2, 1, 68
Staten 1977	MS	3–6 yr-old Cottonwood	Bottomland Hardwood	Mammals	1, 0, 1
Twedt et al. 1999	MS/LA	6–9 yr-old Cottonwood	Bottomland Hardwood	Birds	2, 1, 0
Wilson & Twedt 2003	MS/LA	6–9 yr-old Cottonwood	Bottomland Hardwood	Birds	1, 1, 0
McComb & Noble 1980	MS/LA	14 yr-old Cottonwood	Bottomland Hardwood-thinned Bottomland Hardwood-unthinned Mixed Hardwood–unmanaged Upland Hardwood-thinned Upland Hardwood-thinned	Birds Birds Birds Birds Birds	2, 1, 15 2, 1, 15 2, 1, 15 2, 1, 15 2, 1, 14

[a] Numbers indicate effect sizes for diversity, abundance, and species' abundance, respectively.

Table 4.2 Comparisons of Poplar/Cottonwood Plantations to Reference Forests

Comparison	Birds	Mammals	All Taxa
1 – 8 yr Poplar vs. Surrounding Forest (Hanowski et al 1997, Christian et al. 1997)			
Diversity	0.62 (n=5) **	0.56 (n=1)	0.64 (n=6) **
Abundance	0.90(n=23) **	0.32 (n=3) **	0.75(n=26) **
Species	0.96 (n=7)	0.53 (n=8)**	0.82 (n=15) **
2 – 10 yr Cottonwood vs. 2 – 10 yr Oak Reforestation (Twedt et al. 2002)			
Diversity	1.39 (n=2) **	--	1.39 (n=2) **
Abundance	1.60 (n=1)	--	1.60 (n=1)
Species	1.79 (n=40)	--	1.79 (n=40)
3 – 6 yr Cottonwood vs. Bottomland Hardwood (Tomlinson 1977)			
Diversity	0.79 (n=2) **	--	0.79 (n=2) **
Abundance	0.69 (n=1)	--	0.69 (n=1)
Species	0.60 (n=68)	--	0.60 (n=68)
6 – 9 yr Cottonwood vs. Bottomland Hardwood (Twedt et al. 1999, Wilson & Twedt 2003)			
Diversity	0.92 (n=3) **	--	0.92 (n=3) **
Abundance	0.76 (n=2) **	--	0.76 (n=2) **
Species	--	--	--
14 yr Cottonwood vs. Unthinned Bottomland Hardwood (McComb & Noble 1980)			
Diversity	0.66 (n=2) **	--	0.66 (n=2) **
Abundance	0.63 (n=1)	--	0.63 (n=1)
Species	0.83 (n=15) **	--	0.83 (n=15)
14 yr Cottonwood vs. Thinned Bottomland Hardwood (McComb & Noble 1980)			
Diversity	0.77 (n=2) **	--	0.77 (n=2) **
Abundance	0.84 (n=1)	--	0.84 (n=1)
Species	1.27 (n=15)	--	1.27 (n=15)
14 yr Cottonwood vs. Mixed Hardwood (McComb & Noble 1980)			
Diversity	0.93 (n=2)	--	0.93 (n=2)
Abundance	0.93 (n=1)	--	0.93 (n=1)
Species	0.92 (n=15)	--	0.92 (n=15)
14 yr Cottonwood vs. Unthinned Upland Hardwood (McComb & Noble 1980)			
Diversity	1.06 (n=2) **	--	1.06 (n=2) **
Abundance	1.23 (n=1)	--	1.23 (n=1)
Species	1.34 (n=15)	--	1.34 (n=15)
14 yr Cottonwood vs. Thinned Upland Hardwood (McComb & Noble 1980)			
Diversity	1.16 (n=2) **	--	1.16 (n=2) **
Abundance	1.42 (n=1)	--	1.42 (n=1)
Species	1.23 (n=14)	--	1.23 (n=15)
All Taxa Combined			
Diversity	0.88 (n=22) **	0.56 (n=1)	0.86 (n=23) **
Abundance	0.78 (n=32 **	0.32 (n=3) **	0.72 (n=35) **
Species	0.95 (n=189)	0.53 (n=8)**	0.93 (n=197)

** Indicates bootstrap confidence intervals did not include 1.00; k = # of papers, n = # of effect sizes.

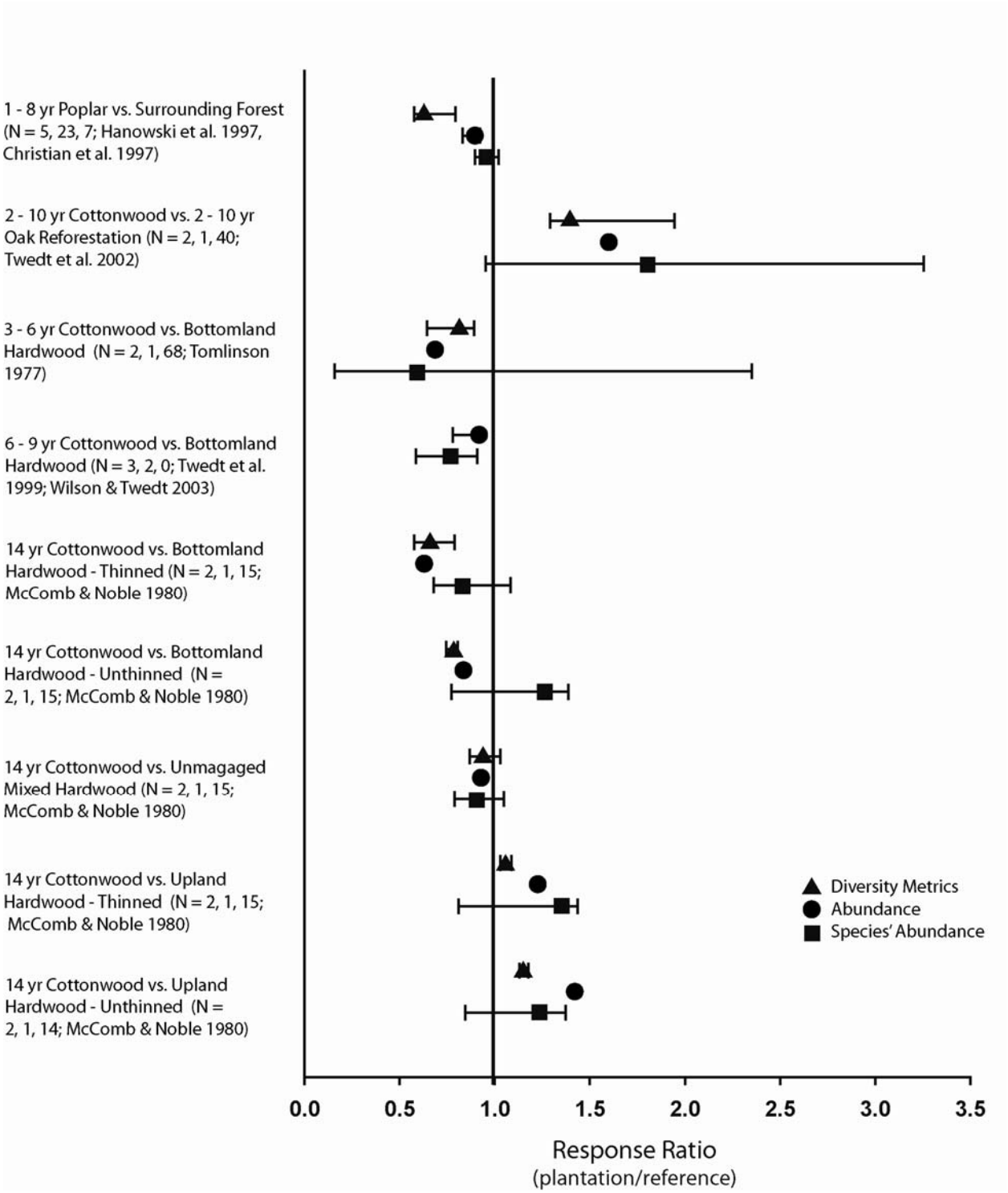

Figure 4.2. Summary Effect Sizes for Birds for Each Unique Comparison of Short-Rotation Woody Crop Plantation vs. Reference Forests

Table 4.3 Summary Effect Sizes for Species with ≥ 2 Effect Sizes
[Confidence intervals that do not overlap 1.0 are in **boldface**.]

Species	Scientific name	RR	df	95% Bootstrap CI
Mourning dove	*Zenaida macroura*	1.012	2	0.193 - 1.018
Yellow-billed cuckoo	*Coccyzus americanus*	0.516	2	0.417 - 3.192
Ruby-throated hummingbird	***Archilochus colubris***	**2.452**	**2**	**1.500 - 2.879**
Red-headed woodpecker	***Melanerpes erythrocephalus***	**0.349**	**2**	**0.111 - 0.500**
Red-bellied woodpecker	***Melanerpes carolinus***	**0.556**	**8**	**0.263 - 0.801**
Downy woodpecker	*Picoides pubescens*	3.626	3	0.333 - 7.600
Hairy woodpecker	*Picoides villosus*	0.337	2	0.167 - 1.300
Pileated woodpecker	***Dryocopus pileatus***	**0.143**	**2**	**0.143 - 0.143**
Acadian flycatcher	*Empidonax virescens*	1.991	2	0.350 - 12.30
Great-crested flycatcher	*Myiarchus crinitus*	0.192	2	0.100 - 1.200
White-eyed vireo	***Vireo griseus***	**0.784**	**6**	**0.669 - 0.937**
Warbling vireo	***Vireo gilvus***	**15.846**	**2**	**6.330 - 23.201**
Red-eyed vireo	*Vireo olivaceus*	1.000	7	0.750 - 1.132
Blue jay	*Cyanocitta cristata*	0.938	2	0.714 - 1.165
Carolina chickadee	*Poecile carolinensis*	0.743	8	0.508 - 1.891
Eastern tufted titmouse	*Baeolophus bicolor*	1.095	8	0.841 - 1.627
Carolina wren	*Thryothorus ludovicianus*	1.134	7	0.806 - 2.044
Blue-gray gnatcatcher	*Polioptila caerulea*	2.571	2	0.536 - 15.201
Eastern bluebird	*Sialia sialis*	1.110	2	0.143 - 1.200
Wood thrush	*Hylocichla mustelina*	1.539	2	0.709 - 1.600
American robin	***Turdus migratorius***	**0.797**	**7**	**0.591 - 0.828**
Brown thrasher	***Toxostoma rufum***	**0.955**	**4**	**0.765 - 0.969**
Northern parula	***Parula Americana***	**0.387**	**6**	**0.153 - 0.784**
Yellow-rumped warbler	*Dendroica coronate*	1.392	6	0.309 - 2.412
Prothonotary warbler	*Protonotaria citrea*	1.533	2	0.361 - 6.600
Kentucky warbler	*Oporornis formosus*	1.349	2	0.500 - 2.100
Common yellowthroat	*Geothlypis trichas*	0.887	3	0.551 - 3.280
Hooded warbler	*Wilsonia citrina*	0.972	6	0.863 - 1.019
Yellow-breasted chat	***Icteria virens***	**5.657**	**2**	**2.765 - 11.670**
Eastern towhee	***Pipilo erythrophthalmus***	**11.668**	**2**	**11.455 - 11.670**
Song sparrow	*Melospiza melodia*	1.605	2	0.526 - 15.669
White-throated sparrow	*Zonotrichia albicollis*	1.359	6	0.699 - 2.021
Summer tanager	*Piranga rubra*	1.481	2	0.625 - 1.600
Northern cardinal	*Cardinalis cardinalis*	1.359	8	0.977 - 2.081
Indigo bunting	***Passerina cyanea***	**1.425**	**7**	**1.328 - 1.457**
Red-winged blackbird	*Agelaius phoeniceus*	0.544	3	0.051 - 49.461

(Continued on next page.)

Table 4.3 Continued

Species	Scientific name	RR	df	95% Bootstrap CI
Common grackle	*Quiscalus quiscula*	0.830	8	0.452 - 1.045
Brown-headed cowbird	*Molothrus ater*	0.923	4	0.909 - 9.041
Orchard oriole	***Icterus spurius***	**1.526**	**2**	**1.476 - 1.554**
American goldfinch	***Carduelis tristis***	**1.047**	**2**	**1.045 - 2.757**

4.3.1 *Effect of Reference Forest Habitat*

Biodiversity response varied with the type of reference forest used (Table 4.2, Figure 4.2). Poplar (1-8 years old) and cottonwood (3-9 years old) plantations were less diverse and had lower overall bird abundance than reference forests (Hanowski, Niemi, and Christian 1997; Christian et al. 1997; Tomlinson 1977; Twedt et al. 1999; Wilson and Twedt 2003) (Table 4.2, Figure 4.2). Fourteen-year-old cottonwood plantations were also less diverse than bottomland hardwood reference forests (both thinned and unthinned; McComb and Noble 1980). In contrast, 14-year-old cottonwood forests had similar diversity and abundance compared to unmanaged mixed hardwoods, but higher diversity and abundance than upland hardwood reference forests (both thinned and unthinned; McComb and Noble 1980). Two- to 10-year-old cottonwoods were more diverse than similarly-aged oak reforestation plantations (Twedt et al. 2002).

Cottonwood plantations may be less diverse than bottomland hardwoods (yet, more diverse than upland hardwoods) in these studies because bottomland hardwoods are located in wetter and more heterogeneous conditions which may result in more diverse communities than upland forests (Drumtra and Cooper 1999; Heitmeyer et al. 1999). Based on comparisons that were available for our meta-analysis, short-rotation poplar and cottonwood plantations seem to be of intermediate diversity among other forest types, although comparisons to plantation forests (and many other forest types) are still lacking.

4.3.2 *Age of Short-Rotation Plantations*

Younger plantations tended to be less diverse than reference forests (response ratios < 1.00), but older plantations were relatively more diverse (higher response ratios). We caution that in our data set, ages are confounded with different reference forest types (and, in some cases, original land cover). However, increasing diversity with age of plantation has been observed for willow and poplar plantations in Europe (e.g., Sage and Robertson 1996, Berg 2002; but see Moser et al. 2002 for opposite response in mammals). As plantations age, they grow in height, which tends to increase both vertical structure and heterogeneity of the stands, increasing number of different nesting and foraging substrates. Moser and Hilpp (2004) found that wintering owls were most commonly found within interiors of relatively older poplar plantations. While this relationship makes ecological sense, more research is warranted about how age (i.e., height and heterogeneity) of SRWCs and characteristics of the land they are planted on influences biodiversity.

4.3.3 *Diversity of Birds during Winter and Migration*

The extent to which SRWC plantations provide migration habitat for birds is unknown. Two studies suggest that cottonwood (Wilson and Twedt 2003) and poplar plantations (Christian et al. 1997) may have lower species richness during migration compared to reference habitat types (bottomland hardwoods and surrounding woodlands, respectively), although differences were not large (Wilson and Twedt 2003). Additionally, cottonwood plantations supported a distinctly different bird community than bottomland hardwoods, suggesting that some cottonwood plantations in an otherwise forested matrix may supplement overall landscape diversity of birds during migration.

It is also unknown how SRWC plantations influence diversity of winter bird communities. Hamel et al. (2002) compared winter bird communities in former agricultural sites where young forests were established using four different afforestation techniques, including planting of cottonwood stem cuttings followed by underplanting of Nuttall oak (*Quercus nuttallii*) seedlings two years later. Cottonwood plantations had higher species richness than sites afforested with different methods (natural succession with regeneration control, sown Nuttall oak acorns, planted Nuttall oak seedlings), but similar total number of birds. However, they did not compare cottonwood to other types of forest (e.g., pine plantations, bottomland hardwoods). In Oregon, hybrid poplar plantations in a heavily agricultural matrix may provide suitable winter habitat for owls because their structure approximates that of native forests and they are often proximate to abundant prey in agricultural areas (Moser and Hilpp 2004). Because of their dense vegetation and sometimes high abundance of small mammals and other prey species, SRWC habitats have potential to provide suitable winter habitat for some birds that prefer dense, shrubby habitats.

4.3.4 *Effects of Landscape Context*

In addition to plantation type and age/structure, and the forest type being replaced, landscape context may contribute to patterns of richness and abundance. Poplar plantations in the Upper Midwest had higher diversity compared to cropland and pasture habitat types and this difference was greater in landscapes dominated by agricultural, urban or grassland land covers compared to forested landscapes (Christian et al. 1997). But poplar plantations had lower diversity than surrounding wooded habitat types in both landscapes. Nevertheless, landscape context could potentially mediate biodiversity response to SRWCs in other areas. Although this has not yet been demonstrated for comparisons between SRWCs and other forest types, we expect landscape context to be influential in some regions. Further research is warranted to establish if, and to what extent, landscape context might mediate effects of converting production forests to SRWCs.

4.3.5 *Bird Population Responses to Short-Rotation Woody Crops*

Higher densities, abundance, or diversity of birds do not necessarily indicate superior (or high quality) habitat because densities can often be higher in sub-optimal or sink habitat types (e.g. Van Horne 1983; Battin 2004). Measuring nesting success, fecundity, and survivorship in SRWC compared to other forest types is necessary to accurately predict effects of expanding SRWCs in forest systems on bird populations and diversity. Only one study has examined nest success in cottonwood plantations (Twedt et al. 2001). For all species and nests combined, nest success was lower on cottonwood plantations compared to bottomland hardwood forests primarily due to higher nest predation and parasitism rates. However, nest success of five species that were abundant in both cottonwood and bottomland hardwoods did not differ between the two forest types. Thus, lower nest success in cottonwoods may be a function of species-specific differences in nesting habitat and life history, rather than a negative effect of SRWC forestry *per se* (Twedt et al. 2001). Twedt et al. (2001) did note, however, that fire ants were more common in cottonwood plantations and were implicated in > 11% of nest failures (unlike reference bottomland hardwoods). Nesting success in SRWC plantations relative to other forest habitat types are generally unknown and poorly understood.

4.4 Response of Mammals to Short-Rotation Woody Crops

Only two studies (Staten 1977; Christian et al. 1997) provided effect sizes for mammals. Diversity was higher on cottonwood plantations compared to bottomland hardwoods in Mississippi (Staten 1977), but was lower on cottonwood plantations relative to surrounding wooded habitat types in the US Upper Midwest (Christian et al. 1997). The meta-analysis response ratio for diversity was close to 1.00 but with wide confidence intervals. Total abundance, guild abundances and species abundances were consistently lower in SWRC (Figure 4.1). Diversity and abundance of mice (Staten 1977; Christian et al. 1997) and shrews (Christian et al. 1997) were consistently lower on poplar plantations compared to surrounding woodlands. As with birds, these did not compare SRWCs to intensively managed forests, so predicting effects of converting intensively managed forests to SRWC plantations is difficult.

For small mammals, other studies provide additional but contrasting information that does little to clarify the situation. Poplar plantations in the midwestern US had lower winter densities of lagomorphs compared to surrounding forest, and other forest mammals were rarely detected using poplar plantations (Christian 1997). In contrast, rabbit (*Silvilagus* spp.) densities were greater on cottonwood plantations compared to surrounding forests in the Mississippi Alluvial Valley (MAV) (Staten 1977; Wesley, Perkins, and Sullivan 1981). Cottonwood plantations elsewhere in the MAV contained a similar number of species to reference forests, but total abundance was intermediate between bottomland hardwoods (lower than cottonwood) and upland hardwoods (higher than cottonwood; McComb and Noble 1980). Additionally, diversity and abundance of small mammals in SRWC plantations may decline with stand age (Moser et al. 2002), further complicating the task of understanding biodiversity response.

Little is known about how large mammals are affected by SRWC plantations. White-tailed deer (*Odocoileus virginianus*) appeared to use poplar plantations no differently than other forested habitat types in the midwestern US (Christian 1997), but were reported to favor cottonwood plantations during parturition in the southern US (Wigley et al. 1980; Wesley, Perkins, and Sullivan 1981). Browse damage to SRWC plantations by ungulates has been reported (Christian 1997) indicating that SRWC trees and other vegetation may be suitable forage, but this has not been quantified. Clearly, additional research is needed to guide management.

4.5 Summary of Biodiversity Responses

Because we found only eight studies that represented nine comparisons of different plantation types (different ages of cottonwood or poplar) to very different types of reference forest, our conclusions are tentative. Diversity and abundance of birds and mammals were lower on SRWC plantations compared to reference forests. Species responses were mixed; SRWC plantations favored shrub-associated birds while canopy/mature forest-associated birds and cavity-nesters typically favored reference forests. SRWC plantations could likely contribute to overall landscape diversity in forest-dominated landscapes by providing shrubby habitat structure for non-forest species. For example, recommendations for maximizing wildlife habitat in the MAV call for ≤ 5% of the landscape to be comprised of shrub/scrub habitats, which SRWC plantations could help provide (Wilson et al. 2007). However, more extensive conversion of intensively managed forests to SRWC would likely decrease overall diversity, especially if they replace high conservation value habitat types (Archaux and Martin 2009).

4.6 Geographic Limitations, Empirical Knowledge Gaps and Research Needs

Burger (2002) considered SRWC systems to be intermediate to forest plantations and agrosystems in terms of management inputs and implications for biodiversity. It is unknown how SRWCs differ from traditional forest plantations because existing studies did not use intensively managed forests as reference. Such comparisons need to be made because intensively managed forests cover increasingly large areas in many regions, provide habitat for species of concern, and support major components of biodiversity (e.g., Carnus et al. 2006; Stephens and Wagner 2007; Brockerhoff et al. 2008). Future studies should also compare SRWC plantations to reference forests in a similar seral stage (e.g., Twedt et al. 2002) in addition to more mature reference forests (the other studies we reviewed). These studies should be continued over the length of the rotation. Present information about diversity in SRWC plantations in the US is limited to the Upper Midwest and the Mississippi Alluvial Valley, and few studies exist for taxa other than birds. Rectifying this lack of basic information should be the top priority for research concerning diversity and SRWCs.

5.0 INTERCROPPING BIOMASS CROPS ON FOREST LANDS

5.1 Background and Definitions

Agroforestry typically emphasizes incorporating woody crops (e.g., hardwoods, softwoods, or nut and fruit trees) into traditional agricultural systems (Garrett and McGraw 2000; Kort et al. 2009). Similarly, herbaceous or biofuel crops could potentially be intercropped in intensive forestry systems and used for co-firing in local energy plants or potentially as cellulosic feedstocks. In these systems, herbaceous crops are seeded between rows of planted trees and then grown and harvested annually (or semi-annually) until shade from crop trees excludes the herbaceous crops. Intercropping will likely use wider tree spacings (often 6.1 m) to provide sunlight to biofuel crops and facilitate access by harvest machinery. Although some forest companies are exploring biofuel intercropping in intensive forest systems, research investigating biodiversity response to these practices is only now being installed.

Switchgrass (*Panicum virgatum*) and other native warm-season grasses (NWSG) such as little bluestem (*Andropogon gerardii*) and big bluestem (*Schizachyrium scoparium*) are currently being evaluated and developed as potential biofuel crops. Native warm-season grasses are C4 plants that historically occurred in North American grasslands and are well-suited to the climate and soils of croplands. Switchgrass has received the most emphasis as a potential intercropped biofuel feedstock because of its ease of establishment, availability of cultivars adapted to much of North America, rapid growth, and high biomass yield (e.g., Schmer et al. 2008; Mitchell, Vogel, and Sarath 2008; Keshwani and Cheng 2009; and many others). Switchgrass has potential for direct co-firing and as a cellulosic feedstock (Schmer et al. 2008), as does a suite of native prairie grasses (Tilman, Hill, and Lehman 2006; Nash 2007). Other potential biofuel species include reed canary grass (*Phalaris arundinacea*; Casler, Cherney, and Brummer 2009) and miscanthus (*Miscanthus x giganteus*; Bellamy et al. 2009). Hereafter, we will distinguish switchgrass from other NWSG. Cool-season grasses include tall fescue (*Festuca arundinacea*) and other C3 grasses that produce forage predominantly in the spring and fall.

Intensively managed forests can provide habitat for many species of conservation concern, support major components of regional biodiversity, and comprise a large proportion of forest land cover in many regions (e.g., Carnus et al. 2006; Stephens and Wagner 2007; Brockerhoff et al. 2008). Thus, adoption of intercropping has the potential to affect forest biodiversity. To examine possible effects of intercropping on biodiversity within intensively managed forest landscapes, we used a systematic literature review to summarize knowledge about how intercropping might affect forest biodiversity and identify knowledge gaps.

5.2 Methodology

We used Wildlife & Ecology Worldwide, USDA Forest Service TreeSearch and Google Scholar databases to search for relevant studies. We searched for the following forestry and biodiversity terms in article abstracts: native warm-season grasses, switchgrass, miscanthus, reed canary grass, pine, dual cropping, intercropping, biodiversity, diversity, richness, wildlife, birds, avian, amphibians, reptiles, invertebrates, insects, and mammals. We supplemented searches by examining bibliographies of articles for additional references.

We found no publications directly evaluating wildlife response of intercropping systems in intensive forest systems in North American. Thus, it is unclear how biodiversity in intensively managed forests with biofuel crops in the rows (i.e., understory) compares to similar forests with traditional management solely for timber production. Additionally, we did not find any articles designed to explicitly compare switchgrass cover to other land covers related to forestry. We did find several studies that compared switchgrass (or native grass plantings) to croplands or other grassland habitat types in agricultural settings (as did Fletcher et al. 2011). These papers primarily addressed bird response and often had small sample sizes, so they may be of limited applicability to forested landscapes. As a result, we have little knowledge about how other taxa might respond to switchgrass intercropping. Thus, we restricted our review to generalizations about wildlife habitat that we could derive from research in row crop settings.

5.3 Responses of Diversity and Wildlife to Intercropping Grasses

Biodiversity response to switchgrass management will likely depend on what cover type or vegetation association is being managed for switchgrass, extent of switchgrass management on the landscape, and other factors. For example, managing low diversity croplands or degraded lands for switchgrass will have different stand-level biodiversity implications than cropping switchgrass in native prairie or forests (Fletcher et al. 2011). However, our review focuses on potential consequences of incorporating a crop of switchgrass or other NWSG within existing forests managed intensively for timber production. We focus primarily on switchgrass and other native prairie grasses because little is known about wildlife responses to miscanthus and reed canary grass (and much less is known about their management and biofuel potential). Because landscape-level implications for biodiversity will likely vary for switchgrass in different spatial contexts and under different cultural practices, it is difficult to predict how widespread adoption would (or would not) influence landscape heterogeneity (and hence landscape-level biodiversity).

5.3.1 *Response to Switchgrass (and NWSG) Management*

Cool-season grass areas typically have lower diversity and less wildlife habitat value because they are denser and less heterogeneous than native grasslands (Washburn, Barnes, and Sole 2000; Johnson 2005; Burger 2005). However, Conservation Reserve Program lands planted to switchgrass (usually through CP2) typically have lower (McCoy et al. 2001; Dinsmore, Burger, and Conover 2009) or similar (King and Savidge 1995; Henningsen and Best 2005) bird diversity, abundance and nest density compared to CRP lands planted to cool-season forages (McCoy et al. 2001). McCoy et al. (2001) found that nest success differed between switchgrass CRP and cool-season CRP, but the direction of this difference changed from year to year.

Native warm-season grasses (e.g., little bluestem, big bluestem) are receiving increasing attention as biofuel (Nash 2007; Tilman, Hill, and Lehman 2006; Lee et al. 2009; and many others). Because other NWSG species are not as dense or homogeneous as switchgrass plantings (Dinsmore, Burger, and Conover 2009; Thompson, Boal, and Lucia 2009), other warm-season grasses, especially in mixed stands, may provide better stand-level habitat quality for grassland birds, and hence, greater diversity. Plantings of switchgrass or other species often may have high density of birds, but mostly

because of high numbers of a few species (e.g., red-winged blackbird [*Agelaius phoeniceus*]) that favor the tall, dense structure provided by switchgrass (Dinsmore, Burger, and Conover 2009; Thompson, Boal, and Lucia 2009). Thus, while switchgrass provides habitat for some bird species, it may not support the full complement of species present in native grasslands.

Intercropping switchgrass would likely replace or alter commonly diverse ground layer in young, regenerating pine stands, but these stands are typically colonized by grassland and early successional species (Burger 2005), at least in the southeastern United States where intercropping is most likely to occur. And, at least structurally, pine-switchgrass systems could be similar to pine-grassland systems that historically dominated large areas of the US South. Intercropped switchgrass would still represent a net addition of grasslands even though switchgrass plantings may not be as diverse as native grasslands. However, net effect on avian diversity is impossible to predict because species that favor recently clearcut and regenerating forests may respond negatively. Of course, extent of this potential response is unknown and effects on other taxa have not been studied.

5.3.2 Response to Forest Structure

There is growing evidence that grassland birds have lower success near wooded edges or avoid wooded edges altogether (e.g., Bollinger and Gavin 2004; Patten et al. 2006; and many others). Edge effects in intercropping systems may occur in two ways. First, surrounding forest types may influence diversity in intercropped stands. Second, young trees in rows will eventually reach heights that may separate the intercropped stand into narrow strips of switchgrass, potentially creating functional edges. Understanding how this arrangement influences birds (and other organisms) is essential to understanding how wildlife may respond to intercropping.

Henningsen and Best (2005) studied birds nesting in switchgrass filter strips that averaged 8–36 m wide. The lower end of this range nearly overlaps with the maximum planted row width currently being considered for intercropping forestry systems in the southeastern US (6.1 m; D.A. Miller, Weyerhaeuser Company, pers. comm.), so this is the best comparison available. Red-winged blackbird nests were much less common in filter strips bordered by woody vegetation, and dickcissels (*Spiza americana*, a grassland bird) avoided nesting in filter strips bordered by woody vegetation altogether. Nesting success in filter strips bordered by woody vegetation was lower compared to non-wooded filter strips through increased predation rates near woody vegetation (Henningsen and Best 2005).

Surrounding landscape structure can mediate nature and extent of edge effects (e.g. Chalfoun, Thompson, and Ratnaswamy 2002). Henningsen and Best (2005) worked in an agriculture-dominated landscape; predator (and parasite) communities and edge effects might be very different in forest-dominated landscapes where switchgrass intercropping is likely to occur. Also, early in the rotation, planted trees may be short enough and spaced far enough apart that intercropped stands function as large grassland tracts with much interior area, thereby initially providing grassland areas in forest-dominated landscapes. However, growing trees will eventually create strips of grassland bordered by trees. Biodiversity implications of this changing structure will depend upon which bird species begin, continue, or stop using these areas and changes in predation and nest parasitism. Most likely, growing trees would gradually shift the bird community from mostly grassland species to shrubland species, and eventually to forest species (Dickson, Conner, and Williamson 1993; Wilson and Watts 2000; Coppedge et al. 2001). Research about when and to what extent growing trees influence nesting birds and/or nest success is needed. Impacts of woody edges on mammals and other taxa in intercropped switchgrass are unknown.

5.3.3 *Response to Timing of Harvest*

Timing of harvest is a critical factor in determining how biodiversity might respond to management of switchgrass. When haying or mowing grassland during breeding season, nesting bird and wildlife mortality can be high (e.g. Bollinger, Bollinger, and Gavin 1990), but this may not be the case with biofuel harvests. Various recommendations for harvest schedules include a) dual harvests in July and September (Roth et al. 2005); b) a single October harvest after senescence (Harper et al. 2007); c) a single harvest in September to allow some re-growth for winter wildlife cover; and d) spring harvest allowing biomass to remain over winter (Lee et al. 2009). There is a growing consensus that a single, post-senescence harvest (e.g. September-November) maximizes total biofuel potential (Harper et al. 2007; Mitchell, Vogel, and Sarath 2008; and others). Following these recommendations, switchgrass grown for biofuel would not be harvested during breeding season for most birds and other wildlife species. Although total biomass harvested may be ≥ 20% less in spring compared to fall (Mitchell, Vogel, and Sarath 2008), total biofuel production is often only slightly reduced (Mitchell, Vogel, and Sarath 2008; Lee et al. 2009). Some other NWSGs such as big bluestem (Lee et al. 2009) may lose little, if any, biomass yield overwinter compared to fall harvests, so NWSGs could provide increased benefits to biodiversity (by providing winter cover) by delaying harvest until spring.

5.3.4 *Response to Harvest-Related Changes in Vegetation*

Although fall harvesting of biofuel may minimize direct effects on reproduction of many wildlife species and diversity, changes in vegetation could persist into subsequent breeding seasons. Biofuel harvest may reduce accumulation of dead vegetation and mimic some aspects of natural disturbances (e.g., fire), thereby ameliorating development of dense vegetation that renders switchgrass less suitable than mixed stands of native grasses for many birds. Harvest may not influence overall bird diversity (e.g., number of nesting bird species [Henningsen and Best 2005]), but may shift community structure towards species that prefer very short grass heights (e.g., grasshopper sparrow [*Ammodramus savannarum*]), while species that prefer taller, denser grasslands (e.g., sedge wren [*Cistothorus platensis*]) might become less common (Murray and Best 2003; Roth et al. 2005). Murray and Best (2003) compared unharvested, strip-harvested (only partially harvested stands), and total harvest regimes in switchgrass. Short-grass species like grasshopper sparrow were more common in total harvests; upland sandpiper (*Bartramia longicauda*) and bobolink (*Dolichonyx oryzivorus*) were most abundant in strip harvests; and sedge wren and ring-necked pheasant (*Phasianus colchicus*) most abundant on non-harvested switchgrass stands (see also George et al. 1979). Red-winged blackbird nest survival rates for both total and strip harvest, however, were lower post-harvest compared to unharvested controls (Murray and Best 2003; see also George et al. 1979). Bird diversity in other NWSG fields (mixed or monocultures) will likely respond to harvest in similar ways as in switchgrass. Bird communities will likely shift to favor short grass species post harvest, and maximum diversity probably would be achieved when harvest regimes are designed to provide a mosaic of different grass heights. Research is needed, however, to confirm these hypotheses.

Maximum biodiversity would likely be achieved using rotational harvest regimes to create a mosaic of grass heights across the landscape (Flaspohler, Webster, and Froese 2009). A mosaic of regimes would provide both short (post-harvest) and tall (1+ years post-harvest) grassland habitat for a wide suite of grassland bird species. Regimes could also include harvesting only a portion (1/2 or 1/3) of each intercropped stand in a given year, or strip harvesting within each stand. Alternatively, if many intercropped stands are located in close proximity, individual stands could be rotationally harvested. However, these regimes would impact economic returns (at least in the short term), so harvest planning should consider tradeoffs between biodiversity goals and economic objectives.

Little is known about how changes in vegetative structure related to harvest might influence taxa other than birds. In mixed native grasslands, mammal diversity and abundance of most small

mammals were lower in hayed vs. unharvested treatments (Kaufman and Kaufman 2008), but some species responded positively to haying. These responses were attributed to changes in vegetation over the winter following harvest. It is possible that mammals in switchgrass stands would respond similarly, but this needs to be confirmed through field experiments.

5.4 Responses of Diversity and Wildlife to Other Herbaceous Species

Other grass species (miscanthus, reed canary grass, etc.) are being investigated for use as biofuel. In Europe, farmland birds were more abundant in conventional fields compared to reed canary grass fields (Vepsäläinen 2010), while miscanthus may benefit some species of birds but not others (Bellamy et al. 2008; Sage et al. 2010). For both miscanthus and reed canary grass fields, bird and mammal communities were more diverse in more natural field margins than in the crop fields themselves (Semere and Slater 2007). However, little is known about how (or if) these grasses might grow in a forestry intercropping system or about how wildlife in North America might use (or not use) these habitat types or respond in intercropping systems involving these grasses.

5.5 Response of Biodiversity to Intercropping-Related Spacing Changes

How different row spacings (for planted crop trees) would potentially impact tree growth, biofuel crop production, and biodiversity of intercropped systems is unknown. Intercropping will likely require use of a wider tree spacing (e.g., 6.1 m) to provide sunlight to biofuel crops and facilitate access by harvest machinery. Lower overstory density resulting from wider spacing would likely increase within-stand heterogeneity – and hence biodiversity – by allowing a more open canopy and more structurally diverse understory (Melchiors 1991; Andreu, Zobrist, and Hinkley 2008). This also may extend the time until canopy closure and have a positive impact on biodiversity because the time period between canopy closure and first thin is typically a low diversity period, especially in southeastern pine forests (Burger 2005).

Research about different row spacings, however, is lacking. In Louisiana, there were no differences in bird communities between stands planted on 4.3-m spacing vs. 6.1-m spacing (Taylor 2008) during the first two years after planting. Some mammals (cotton rat [*Sigmodon hispidus*] and house mouse [*Mus musculus*]) were more abundant on 4.3-m spacing, but distribution of DCWD was a much stronger predictor of mammal abundance than row spacing (Bechard 2008). In North Carolina, bird abundance and species richness was greater for six years after planting in plantations with 6.1-m spacing compared to plantations with 3-m spacing (Lane 2010). Small mammal capture rates, richness, and diversity were also greater in 6.1-m spacing for the first two years following site preparation (Lane 2010). On these same sites, Marsh (2011) found that 3-m spacings had greater cover of woody and vine food plants for white-tailed deer (*Odocoileus virginianus*) compared to 6.1-m spacings in Years 1 and 8 after site preparation. In Years 2 through 7, cover of these plants did not differ between wide and narrow spacings. However, spacing in the studies by Lane (2010) and Marsh (2011) was confounded with site preparation method (shear blade in 6.1-m, chop in 3-m), so these differences cannot be solely attributed to spacing width.

5.6 Geographic Limitations, Empirical Knowledge Gaps and Research Needs

No published findings specific to biodiversity or wildlife in these kinds of intercropping systems exist. Knowledge about managing switchgrass (and other native grasses) for birds comes primarily from the midwestern and southeastern US. We list several important questions for future research below.

1. *What are the basic biodiversity responses to intercropping in forest systems?* Evaluations of wildlife response to pilot intercropping projects are needed to guide and facilitate additional research. Even basic information about biodiversity response is lacking, and our recommendations are largely extrapolated from research in agriculture-dominated regions, not from extensively forested regions where intercropping is likely to occur.

2. *How do stand and landscape characteristics influence biodiversity in intercropped forest systems?* Research about how edge effects influence biodiversity response and reproductive success in switchgrass intercropping systems (and other herbaceous biofuel crops) and dual-cropping pine systems is needed. Research should describe how fecundity, survival, predation rates, and other measures change as planted crop trees grow and introduce woody structure into grass stands. If faunal species prefer or avoid reproducing in intercropping systems when woody crop trees develop, if growing trees influence predation/parasitism rates, or if biomass harvests affect reproduction, then population growth rates of those species may change with time. Information about intercropped stand characteristics like size, shape, and position in the landscape may influence colonization patterns, genetic diversity and metapopulation persistence in the landscape. Stand characteristics may also exacerbate or mediate edge effects.

3. *How do intercropping practices/systems influence biodiversity and economic return?* Research is needed to identify how row spacing influences tree growth, production by biofuel crop species (minimize shading), and biodiversity. Studies of different compositions of switchgrass and NWSG (including both plantings and mixtures of species, including forbs) are needed to identify the relationship between species mix and both biofuel yield and biodiversity. Studies of biodiversity response to different harvest strategies (e.g., timing and frequencies of harvests) are also needed. Without this information, intercropped forest biofuel systems cannot be optimized for either production or biodiversity goals.

6.0 SUMMARY OF RELATIONSHIPS BETWEEN INTENSIVE BIOMASS PRODUCTION AND BIODIVERSITY IN NORTH AMERICAN FORESTS

We used a literature review and meta-analyses of manipulative and observational studies to assess potential biodiversity responses to practices associated with intensive biomass production systems in North American forests. Biodiversity responses varied among taxa and among the different production systems were reviewed. A small number of studies or strong geographic bias in available studies for some practices increased uncertainty about consistency of observed responses across different landscapes of North America. Below (and in Figure 6.1), we summarize both the nature of biodiversity responses (e.g., positive or negative) to production systems and the degree of uncertainty about those responses.

1. *Most taxa responded positively to thinning treatments.* Most taxa responded positively to thinning treatments and there is more certainty about results from studies of thinning than for other production practices because we found more publications with broader geographic coverage. Strong, positive responses were observed for birds, mammals and reptiles. Although responses of reptiles were strongly positive, there is considerable uncertainty

because they were based on only one study. There was also some uncertainty about responses by invertebrates (positive response) and amphibians (neutral or mixed responses).

2. *Reducing CWD (similar to removing harvest residues) will likely decrease bird diversity, but other taxa may not respond strongly. If reductions in coarse woody debris from actual harvests are less than the 70–95% used in experimental studies, biodiversity responses may be minimal.* Bird diversity responded negatively to removal of CWD and a reasonably large number of studies provided some certainty about this response. Responses for invertebrates (negative response), reptiles (neutral/mixed), amphibians (neutral/mixed) and mammals (slight negative) were associated with moderate to high amounts of uncertainty.

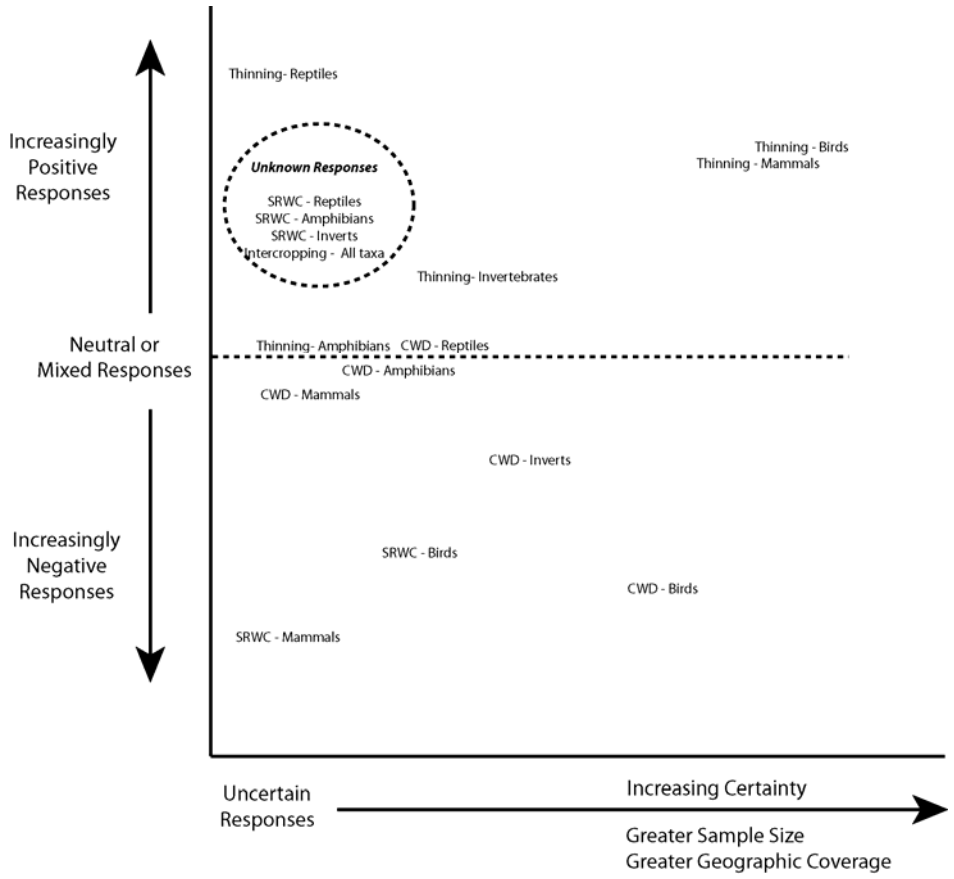

Figure 6.1 Summary of Biodiversity Responses to Biomass Harvest Systems [Production system codes are as follows: thinning = thinning; CWD = removal of forest harvest residues; SRWC = short-rotation woody crops; intercropping = intercropped grasses.]

3. *Short-rotation woody crops may have lower diversity of birds and mammals than production forests, but there is considerable uncertainty.* Poor geographic distribution and a limited number of effect sizes from only *Populus* spp. systems makes it difficult to predict biodiversity response to these systems if applied in other forest types or regions. Further complicating this is the lack of temporally matched, appropriate reference forests in existing studies. Responses of reptiles, amphibians and invertebrates to SRWCs are totally unknown.

4. *Intercropping switchgrass or other grasses has unknown effects on biodiversity*. If intercropped grasses provide quality habitat for grassland species early in the rotation (when trees are short), this could increase overall landscape-scale diversity in forest-dominated landscapes. However, even basic knowledge about intercropping-wildlife relations is lacking.

We identified two additional information gaps related to intensive biomass harvesting. First, all biomass production systems we reviewed (with the exception of removing residues during harvest) potentially involve added forest operations beyond those normally required by traditional production forestry and/or increased frequency of operations. For example, intercropped switchgrass or other biofuel crops would require at least annual biomass harvesting during some portion of the forest rotation, thereby increasing potential for direct (e.g., disruption of nesting) and indirect (e.g., changes to vegetation structure or soils) effects on wildlife. We found no studies that characterized biodiversity response to added or increased frequency of harvesting operations. Second, biodiversity response to intensive biomass harvesting cannot be fully understood without considering spatial and temporal context. All of the studies we reviewed characterized biodiversity response either over a short time period (i.e., immediately after practice was implemented), at the stand-scale, or both. Thus, additional research about added or increased frequency of harvesting operations, research about biodiversity response at larger scales, and perhaps studies of biomass harvesting within different landscape contexts would greatly strengthen both our understanding of biodiversity response and the technical basis for harvesting guidelines.

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Note: Many of the items in the reference list contain digital object identifiers (DOIs). DOIs allow for persistent links for electronic objects. More information is available at www.doi.org.

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