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Estimates of waste rice, natural seeds, and wetland birds in Gulf Coast Prairie ricelands during fall–winter

By

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Estimates of waste rice, natural seeds, and wetland birds in Gulf Coast Prairie ricelands during fall–winter

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Hundreds of wetland bird species use ricelands annually in the Gulf Coast Prairie region of Louisiana and Texas. Much of the original ecosystem was transformed for rice and other crops, cattle ranching, flood control, and other human uses. Flooded production and idled ricelands provide critical foraging habitat for breeding, migrating, and wintering wetland birds. Ricelands in coastal Louisiana and Texas provide approximately 42% of the estimated habitat carrying capacity for wintering waterfowl in this region. In 2010, the Deepwater Horizon oil spill in the Gulf of Mexico prompted enactment of the Migratory Bird Habitat Initiative (MBHI) by USDA Natural Resources Conservation Service. The MBHI provided avian habitat, including flooded ricelands, inland from oil impacted areas. My objectives were to: 1) estimate and model variation in biomass of waste rice and natural seeds as potential waterfowl forage in Gulf Coast Prairie ricelands, 2) estimate and model variation in wetland bird use of ricelands, and 3) conduct sensitivity analyses of bioenergetics models by varying foraging thresholds and true metabolizable energy (TME) values. A growing season of ~270 days allows Gulf Coast rice producers to grow two rice crops (i.e., the second termed ratoon). Waste rice was greatest in production fields with harvested and standing ratoon crops, and natural seed biomass was greatest in idled fields with standing vegetation. Wetland bird species richness and waterbird abundance were greatest in shallowly flooded (1–15 cm) ricelands with sparse vertical vegetation (1–20 cm), and duck abundance was greatest in shallow–intermediately (1–30 cm) flooded ricelands with short vegetation (1–15 cm). Shallowly flooded rice fields containing harvested or standing ratoon crops, and shallowly flooded idled fields with standing vegetation provided abundant potential foods for waterfowl and waterbirds. Bioenergetics models indicated that planners in the Gulf Coast Prairie region may be underestimating riceland habitat requirements for waterfowl by 10,000 ha. Models were most sensitive to changes in seed biomass estimates, and less sensitive to changes in foraging thresholds and TME values. Collectively, these results will facilitate conservation partners to refine models for conserving habitats for waterfowl and other waterbirds in the Gulf Coast Prairie.

DEDICATION

I dedicate this dissertation to my wife, Deanna, who has showed endless support, encouragement, love, and care for me during my education at Mississippi State University. I am eternally grateful for all you have given me, and I look forward to our future. I also dedicate my dissertation to my father and mother, Paul and Sandra Marty. They were without doubt my greatest supporters who encouraged me to pursue my dreams and follow my passions.

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CHAPTER I

INTRODUCTION

Hundreds of wetland bird species that include waterfowl, wading birds, and shorebirds annually use nutrient rich interior and coastal wetlands in the Gulf Coast Prairie (GCP) region of Louisiana and Texas (Esslinger and Wilson 2001, Wilson and Esslinger 2002, Eadie et al. 2008, Marty et al. 2015). Pristine habitats of the GCP included extensive coastal marshes and prairies, freshwater wetlands, and savannas (Esslinger and Wilson 2001). However, much of this original ecosystem was lost or transformed for rice and other commercial crops, flood control, and other land uses (Dahl 2011). The region has an impervious clay pan, long growing season, mild climate, abundant rainfall (77–113 cm annually; Gosselink et al. 1979, Hobaugh et al. 1989), and an abundance of land that provides optimal rice producing conditions. The modern rice industry in the GCP began in the late 1800s. Rice agriculture continued to expand within the GCP through the 1900s to >400,000 ha, but has declined to approximately 140,000 ha in 2015 (USDA 2016). Despite recent declines, rice remains among the dominant crops in the GCP landscape.

Although ricelands are not considered as diverse and productive as natural wetlands, vegetation structure of rice is dense herbaceous, similar to some seasonal wetlands. Production and idled ricelands provide critical wetland habitat for breeding, migrating, and wintering wetland birds, and are an important source of dietary energy

(Meanley 1956, Remsen et al. 1991, Rettig 1994, Elphick and Oring 1998, Eadie et al. 2008, Stafford et al 2010). Flooded ricelands typically provide abundant energy through waste rice, natural seeds, tubers, and invertebrates (Low and Bellrose 1944, Fredrickson and Taylor 1982, Kross et al. 2008, Hagy and Kaminski 2012, Schummer et al. 2012, Marty et al. 2015). The Gulf Coast Joint Venture (GCJV), a partnership around which collaborative conservation for migratory bird habitats is based in the Western Gulf Coast, endeavors to provide habitat for millions of wetland birds annually during winter (U.S. Department of the Interior and Environmental Canada 1986, Esslinger and Wilson 2001, Wilson and Esslinger 2002, U.S. Department of the Interior et al. 2012). Ricelands in coastal Louisiana and Texas provide approximately 42% of the estimated carrying capacity for wintering waterfowl in the GCJV region (Petrie et al. 2014).

In 2010, the Deepwater Horizon oil spill in the Gulf of Mexico affected vast hectarage of coastal wetland bird habitats along the Gulf Coast, which prompted natural resource agencies to develop programs for enhancing inland habitats for migratory birds (Davis et al. 2014, Kaminski and Davis 2014). One of these programs was the Migratory Bird Habitat Initiative (MBHI), which was implemented through the Natural Resources Conservation Service (NRCS) and led to cooperative efforts among conservation organizations, agricultural producers, and a variety of other landowners to enhance habitat for migratory birds on private lands (Davis et al. 2014, Kaminski and Davis 2014). In the GCP, one of the primary objectives was to flood production and idled ricelands and other wetland habitats during autumn and winter.

Because ricelands are a major component of the carrying capacity for waterfowl wintering in the GCJV, precise contemporary estimates of waste-rice and natural seed biomass are necessary for effective habitat conservation planning and implementation. Current estimates of waste-rice and natural seed biomass used in GCJV carrying capacity models were derived from studies that were relatively limited in temporal and spatial replication, and therefore likely did not fully capture the variability in food resources across space and time within the GCJV region (T. C. Michot and W. Norling, U. S. Geological Survey, unpublished data).

The need for precise and contemporary food resource estimates, and the initiation of the MBHI both afforded me the opportunity to investigate waste-rice and natural seed biomass, and wetland bird use of GCP production and idled ricelands. The objectives for my dissertation research were to: (Chapter II; 1) estimate waste-rice and natural seed biomass in production, seed-rice, and idled rice fields with an acceptable level of precision (CV ≤15%; Stafford et al. 2006*a*,*b*); 2) model variation in field-level rice and natural seed biomass in production and idled rice fields in November, relative to weather, soil, and field classifications for comparison with similar research conducted in the MAV (Stafford et al. 2006*a*,*b*); and 3) estimate and compare November waste-rice and natural seed biomass by seed variety (i.e., Clearfield® and conventional); (Chapter III; 4) estimate and model variation in duck and other waterbird (i.e., waders, shorebird, rails, and other birds) species richness and abundance in relation to habitat characteristics and rice-seed varieties of production rice fields, and habitat characteristics of idled rice fields; (Chapter IV; 5) investigate effects that applying GUDs and FATs at different ecological scales, and using average versus species-specific TME values, have on available metabolizable energy (AME) estimates; and subsequent habitat requirements necessary to support LCP waterfowl populations from August–March; and 6) compare estimates of habitat requirements from my study to current GCJV estimates. Previously published data collected in 2010 as a part of my Masters pilot study are included in Chapter II, III, and IV (Marty 2013, Marty et al. 2015)

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CHAPTER II

SPATIO-TEMPORAL EVALUATION OF WASTE-RICE AND NATURAL SEED BIOMASS IN PRODUCTION AND IDLED RICE FIELDS IN THE GULF COAST PRAIRIES OF LOUISIANA AND TEXAS

The North American Waterfowl Management Plan (NAWMP) was implemented in 1986 to increase continental waterfowl populations that declined during the early 1980s from widespread drought and anthropogenic land uses in Prairie Pothole and Parkland Regions of the northern United States and southern Canada (U.S. Department of Interior and Environment Canada 1986). The NAWMP established habitat and population goals for species in North America, and charged Joint Ventures (JV) with implementing NAWMP recommendations at regional scales. Since inception, the NAWMP has adapted and evolved as new scientific information has become available to support waterfowl conservation decisions (Humburg and Anderson 2014).

The food-limitation hypothesis has been conceived and supported by scientific literature indicating that habitat conditions (i.e., food biomass, habitat and food availability, cover, etc.) and diet quality (i.e., seeds, vegetation, and invertebrates) can influence body condition, survival, migration phenology, and clutch size in birds (Lack 1947; Heitmeyer 1988, 2006; Loesch and Kaminski 1989; Moon et al. 2007; Osnas et al. 2016). During non-breeding periods, habitat and resource use is essential for successful completion of spring migration and subsequent breeding (Paulus 1988, Davis et al. 2014).

Diet composition and access to high quality foraging habitats influence body condition, including nutrient reserves for migrating and wintering waterfowl (Delnicki and Reinecke 1986, Jorde et al. 1995, Ballard et al. 2006). Wetland habitat conditions encountered by waterfowl, particularly in late winter and early spring, may influence subsequent reproduction and recruitment (Heitmeyer and Fredrickson 1981, Kaminski and Gluesing 1987, Raveling and Heitmeyer 1989, Osnas et al. 2016). Williams et al. (1999) suggested availability of food resources as a factor with potential to influence survival of waterfowl populations during non-breeding periods. Bioenergetics models, which represent a class of resource depletion models, are used to integrate the food-limitation hypothesis with conservation plans for migrating and wintering waterfowl. Generally, JVs in non-breeding areas for waterfowl presume that food abundance and availability during non-reproductive seasons can influence waterfowl body condition and survival (i.e., the food-limitation hypothesis; Williams et al. 2014). The Gulf Coast Joint Venture (GCJV) of NAWMP endeavors to provide foraging habitat for approximately 14 million ducks and 1.6 million geese (*Anatidae*) annually during winter, emphasizing the importance of the Gulf Coast Prairies (GCP) ecoregion of Louisiana and Texas for sustainment of North American waterfowl and waterbird populations (U.S. Department of the Interior and Environmental Canada 1986, Esslinger and Wilson 2001, U.S. Department of the Interior et al. 2012). Given the need to support 17–19% of the continental waterfowl population during winter, the GCJV prioritizes conservation actions that enhance food availability within the GCP ecoregion.

The Chenier Plain (CP) of Louisiana (LCP) and Texas (TCP) and Texas Mid-Coast (TMC) are GCJV initiative areas (i.e., conservation planning regions) where migrating and wintering waterfowl commonly consume energy rich food resources such as waste rice (3.34 kcal/g; Reinecke et al. 1989) and natural seeds (2.47 kcal/g; Kaminski et al. 2003). Rice is an important agricultural seed used by wetland birds and is often grown in areas where natural wetlands previously occurred because of hydric soils (Eadie et al. 2008). Most rice grown in the United States is produced in the Central Valley of California, the Mississippi Alluvial Valley (MAV), and the GCP (Petrie et al. 2014). Although commercial agriculture and other anthropogenic activities have altered the natural biotic communities of these landscapes, rice and other croplands provide important habitats for diverse guilds of waterfowl and waterbirds (Hobaugh et al. 1989, Reinecke et al. 1989, Elphick et al. 2010, Dahl 2011, Marty et al. 2015). In the southeastern United States, rice agriculture extends across southeastern Missouri, eastern Arkansas, western Mississippi, and northern Louisiana, south and westward into the CP and TMC; these latter two regions formed the core areas of my research.

The rice growing region is one of the most important waterfowl habitats in the GCP ecoregion, yet numbers have declined since the 1980's. In the late 1960's, producers planted approximately 429,993 ha of rice in southwest Louisiana and along the upper coast of Texas (U.S. Department of Agriculture [USDA] 2017). However, low commodity prices, high productions costs, farm policy, urban development, and restrictions on water supplies have reduced rice production in the GCP, with only 140,000 ha of rice planted in 2015 (USDA 2017). In addition to the aforementioned commercial rice production, seed rice (i.e., rice grown and harvested for subsequent planting) is produced in the GCP ecoregion. Seed-rice production in Louisiana decreased 63% from 6,074 ha in 2005 to 2,221 ha in 2015. Conversely, during 2008–2011, seed-rice production in Texas increased 109% from 8,036 ha to 16,796 ha. However, seed-rice production in Texas has steadily declined since 2011 with only 1,171 ha planted in 2015. Seed-rice production has declined of late because, 1) the commercial rice price has not followed other commodities and has been suppressed for a long period of time causing growers to rotate to other crops, 2) the seeding rate of rice has been significantly reduced, and 3) yields of the newly developed rice varieties is increasing (L. Cannon, Louisiana Department of Agriculture and Forestry, personal communication). Thus, a more complete understanding of food resource dynamics provided by riceland systems is needed to support programs, policies, and management actions that encourage the sustainability of rice production with respect to waterfowl and other wetland birds.

Spatio-temporal dynamics of rice lost during harvest (i.e., waste rice) and natural seeds for foraging waterfowl have been studied extensively in some of the rice growing regions of the United States (Miller et al. 1989; Reinecke et al. 1989; Loughman and Batzer 1992; Manley et al. 2004; Stafford et al. 2006*a*,*b*; Kross et al. 2008*a*,*b*; Greer et al. 2009; Havens et al. 2009; Hagy and Kaminski 2012). Abundance of waste rice is variable and influenced by region, sampling month, harvester type (i.e., conventional or stripper header), and post-harvest farming practices including, burning, disking, rolling, flooding, or those remaining dry and with standing stubble (Stafford et al. 2006*b*, Kross et al. 2008*a*, Havens et al. 2009). Increased harvester efficiency and early planting and harvest result in marked decomposition of waste rice during fall (Manley et al. 2004, Stafford et al. 2006*b*). Over a 71% decline in waste-rice biomass from time of harvest (271 kg [dry]/ha; mid-late September) through late fall (78.4 kg [dry]/ha; late November-early December) has been documented in the MAV (Stafford et al. 2006*b*). By comparison, managed emergent wetlands in the MAV may contain natural seed biomass of 556 kg [dry]/ha (Kross et al. 2008*b*). These results have important implications for waterfowl habitat conservation planning and implementation, because rice fields that are winter flooded account for 11% and 44% of the estimated habitat carrying capacity for wintering ducks in the MAV and Central Valley of California, respectively (Petrie et al. 2014). Production and idled ricelands in coastal Louisiana and Texas provide approximately 42% of the estimated carrying capacity for wintering waterfowl in the GCJV region (Petrie et al. 2014). Thus, precise and contemporary estimates of waste-rice and natural seed biomass, and an understanding of their temporal dynamics, are necessary for effective habitat conservation planning and implementation.

Agricultural practices for rice production differ among the three primary growing regions of the United States and are influenced by climate (i.e., length of the growing season), economics, water resources, soil characteristics, and other factors (Manley et al. 2004, Eadie et al. 2008, Stafford et al. 2010, Marty et al. 2015). In the CP and TMC, producers regularly grow and harvest a second rice crop (i.e., ratoon [Spanish origin from the word *retono*, meaning a sprout]) in November from the first rice crop that is harvested in July–August. This practice is generally not possible in the MAV or Central Valley of California because of shortened growing seasons (Bollich and Turner 1988, Hobaugh et al. 1989, Eadie et al. 2008, Havens et al. 2009, Stafford et al. 2010). Planting and harvest practices for seed rice include a single harvest in autumn, followed by an idle period through winter and spring. Additionally, crop rotation strategies differ among rice growing regions of the United States. Growing rice in the same field during consecutive years would increase disease and weed prevalence, and decrease soil fertility resulting in reduced yields (Hohman et al. 2014). In the Central Valley of California and the MAV, rice fields are commonly rotated between years with other crops such as soybean, wheat, or corn. However, in the GCP, rotational options are limited, and producers typically idle rice fields for 1–2 years. In idle rice fields, natural vegetation (i.e., moist-soil plants) such as grasses (*Poaceae*), sedges (*Cyperaceae*), rushes (*Juncaceae*), and forbs (*Polygonaceae*) will typically grow and produce abundant seeds and tubers, as well as provide substrates for aquatic invertebrates (Low and Bellrose 1944, Fredrickson and Taylor 1982, Kross et al. 2008*b*, Hagy and Kaminski 2012, Schummer et al. 2012, Marty et al. 2015). During idle years, some producers will graze cattle to aid in the reduction of prevalent weeds and grasses, which provides an economic return from idled land (Craigmiles 1975; Hobaugh et al. 1989).

Rice producers plant different rice varieties. Clearfield® rice varieties provide superior weed control compared to traditional rice varieties, and hence have become increasingly popular for agriculture since 2002 (Wilson et al. 2010). More than 60% of all rice hectares in the United States are now planted in Clearfield® varieties (Wilson et al. 2010). Clearfield® is a non-genetically modified crop technology that provides selective herbicide resistance to rice plants, thereby enabling increased control of broadleaf and grass plants in rice fields (Croughan 2003). Despite apparent advantages for producers, there is growing speculation among waterfowl hunters that traits related to Clearfield® rice (e.g., more effective weed control) are leading to decreased use, or avoidance, of fields by waterfowl. If waterfowl and other granivorous waterbirds avoid fields planted with Clearfield® rice varieties, there could be implications for waterfowl and other avian foraging strategies, body condition, and subsequent survival of waterfowl.

The GCJV region provides essential habitat to large concentrations of wintering waterfowl and other wetland birds. However, our lack of a current understanding of spatial and temporal dynamics of waste-rice and natural seed biomass justifies need for a region-wide study to estimate abundances of these important foods. Previous biomass estimates of waste rice and natural seeds in the GCJV region were derived from studies with limited temporal and spatial replication, and existing information is outdated (T. C. Michot and W. Norling, U.S. Geological Survey, unpublished data).

My objectives were to: 1) estimate waste-rice and natural seed biomass in production, seed-rice, and idled rice fields with an acceptable level of precision (CV ≤15%; Stafford et al. 2006*a*,*b*), 2) model variation in field-level rice and natural seed biomass in production and idled rice fields in November, relative to weather, soil, and field classifications for comparison with similar research conducted in the MAV (Stafford et al. 2006*a*,*b*), and 3) estimate and compare November waste-rice and natural seed biomass by seed variety (i.e., Clearfield® and conventional). I hypothesized that waste-rice and natural seed biomass in production, seed-, and idled rice fields would vary temporally and among field classifications. I predicted that waste-rice biomass would increase from August–November in fields with harvested and standing ratoon crops, because of the increase of waste rice resulting from the ratoon cropping practices in the GCP in contrast to the MAV. Additionally, I predicted that waste-rice and natural seed biomass would decline from August–November in seed-rice fields, similar to MAV patterns (Manley et al. 2004 Stafford et al. 2006*b*), because no ratoon crops are produced in seed-rice fields. Finally, I hypothesized that biomass of natural seeds would be lower in Clearfield® fields due to superior weed control, and there would be no discernable difference in rice between rice varieties. My study provides needed contemporary spatial and temporal estimates of waste-rice and natural seed biomass for GCJV waterfowl conservation planning and is an important contribution for use in bioenergetics models to refine potential carrying capacity estimates for the regions.

Study Area

I conducted my study in agricultural landscapes of the CP of Louisiana and Texas and the TMC. The CP encompasses areas of southwest Louisiana and southeast Texas, roughly spanning from Lafayette, Louisiana, westward to Houston, Texas, and extending inland 130–160 km from the coastline (Figure 2.1). The TMC extends from Galveston Bay to Corpus Christi, Texas, and inland from the coastline approximately 170 km (Figure 2.1). My specific study area included the Louisiana parishes of Acadia, Allen, Calcasieu, Cameron, Evangeline, Jefferson Davis, St. Landry, and Vermilion, and the Texas counties of Brazoria, Chambers, Colorado, Jackson, Jefferson, Liberty, Matagorda, and Wharton. These counties aligned closely with the GCJV's Chenier Plain and Texas Mid-Coast Initiative Areas.

Historically, these regions contained extensive coastal marshes and prairies, freshwater wetlands, and savannahs. Today, the CP and TMC contain coastal marshes along the Gulf of Mexico, but many of the historic coastal prairies and savannas have been converted for cultivation of rice and other crops (Esslinger and Wilson 2001). The climate is sub-tropical and humid with an average growing season of 270 days, 13 freeze-days per year, and temperatures ranging from 14° C in December–January to 30° C July–

August (Chabreck et al. 1989). Average annual precipitation decreases east to west in the CP from 144 cm near Lafayette, Louisiana, to 113 cm per year near Houston, Texas and 77 cm per year near Corpus Christi, Texas (Gosselink et al. 1979, Hobaugh et al. 1989). The CP and TMC regions are subject to frequent and sometimes intense weather disturbances; on average, tropical storms make landfall approximately once every 1.6 years and hurricanes every 3.3 years (Roth 1999).

Methods

Sampling Design

I randomly collected soil cores based on a stratified, multi-stage survey design with primary, secondary, and tertiary sampling strata: 1) rice farms; 2) production and idled rice fields within farms; and 3) soil cores collected within fields (Stafford et al. 2006*a,b*; Marty et al. 2015). I derived my sampling universe of GCP farmers from Louisiana rice producers who enrolled in the USDA Natural Resources Conservation Service Migratory Bird Habitat Initiative (MBHI; Kaminski and Davis 2014) and Texas producers who cooperated with Ducks Unlimited, Inc. through private land wetland restoration programs (i.e., the Texas Prairie Wetlands Project). The MBHI and Texas Prairie Wetlands Project are incentive-based habitat management programs which promote conservation and flooding of natural and agricultural habitats for waterfowl. I considered these databases representative of ricelands and producers within my study region, because local agronomists advised that agricultural practices employed by producers were typical of the population of commercial rice producers within my study area (S. D. Linscombe, Louisiana State University Agricultural Center [LSUAC], personal communication). I selected producers randomly, and stratified samples into

LCP, TCP, and TMC regions. In 2010, I selected a total of 25 producers and allocated selections between regions roughly in proportion to average area planted to rice in each region (LCP [60%], *n* = 15; TMC [40%], *n* = 10). During each year 2011–2013, I randomly selected 25 producers across the LCP, TCP, and TMC regions, and I allocated my selection among regions in proportion to planted rice acreage as measured in 2011 (LCP [64%], *n* = 16 producers; TCP [12%], *n* = 3 producers; TMC [24%], *n* = 6 producers). I randomly selected and sampled two production and two idled rice fields for each producer. I defined fields as areas of varying size surrounded by exterior levees that contained rice or were temporarily idled. I sampled seed-rice fields only during 2012–2013 in the TMC and TCP due to limited access of these producers. In the TMC in 2012, I selected four seed-rice producers. From these I sampled three fields each from two producers and two fields each from the other two producers (*n* = 10 fields). In 2013, I sampled three producers from the TMC and one from the TCP (*n* = 9 fields, *n* = 3 fields, respectively).

Field Sampling

Field classifications of production and idled rice fields included: 1) July–August harvest only (first harvest, FH); 2) fields harvested in August and again in November for a ratoon crop (harvested ratoon, HR); 3) fields in which a second crop was grown but not harvested and left standing, generally for crawfish aquaculture or waterfowl habitat (standing ratoon, SR); 4) fields harvested in July–August but with no ratoon crop grown (no ratoon, NR); 5) idle fields with standing natural vegetation (standing idle, SI); and 6) disked idled fields (disked idle, DI). Application of these field classifications was not mutually exclusive. For example, all production rice fields were harvested July–August, but each was subject to one of several unique practices (e.g., classifications 2–4) that affected field dynamics (e.g., food dynamics, water depth, vegetation conditions) during autumn. Thus, some of my identified field classifications are best viewed as a combination of farming activity and sampling period.

During the 2010 and 2011 field seasons, I established in each selected field a single random directional (1–180°) transect and extracted 10 soil cores (10 cm diameter and depth), each spaced ~25 m apart (Manley et al. 2004, Stafford et al. 2006*b*). I used data from 2010 and 2011 to calculate optimal sample sizes by field classification for the remainder of my study. Therefore, during 2012 and 2013 field seasons, I collected 10 soil cores from FH, SI, and DI fields, 15 cores in SR and NR fields, and 6 cores in HR fields (J.R. Marty, unpublished data). Additionally, during August and November 2012–2013, I collected 15 soil cores from each randomly selected seed-rice field, using the same sampling protocol as for conventional production and idled fields. I selected 15 August and 1 November as my target sampling dates because these corresponded to the beginning of the early and late conservation planning periods identified by the GCJV (Esslinger and Wilson 2001). For both sampling periods, I collected soil cores from production rice fields 1–7 days after harvest or, for the November sampling period, immediately after farmers indicated the ratoon crop would not be harvested and left as a forage base for crayfish (*Procambarus* spp.), or as waterfowl habitat. In addition to August and November collection periods, I collected soil cores from idled rice fields in early October 2010 (n = 10 cores/field) and in early October 2012 from SI (n = 6 cores) and DI fields (n = 10 cores), because seeds of many moist-soil plants had not matured and dehisced by mid-August. This allowed me to examine temporal dynamics of moist-soil seed biomass in idled fields on a finer scale (2010, *n* = 15 fields, *n* = 250 cores; 2012, *n* = 50 fields, *n* = 378 soil cores).

Laboratory Procedures

I replicated core sample processing procedures from previous and related studies (Manley et al. 2004; Stafford et al. 2006*a*,*b*; Kross et al. 2008*a,b*; Hagy et al. 2011; Hagy and Kaminski 2012). I stored soil cores at -13° C to preserve seed biomass and deter germination and decomposition (Murkin et al. 1994, Stenroth and Nyström 2003). I randomly selected soil cores for processing from the freezer regardless of collection date to minimize bias resulting from potential storage degradation. Once thawed, I used a mixture of 3% solution of hydrogen peroxide (H_2O_2), a mixture of ≤ 250 cm^3 baking soda and approximately 1L water, or a combination of these to oxidize clay and facilitate sediment transport through sieves (Bohm 1979; Kross et al. 2008*a,b*). I washed the cores through a series of sieves containing mesh sizes 4 (4.75 mm), 10 (2.0 mm), and 50 (300 µm) to remove rice and natural seeds containing whole or partially intact endosperm (i.e., ≥50% of seed remaining; Stafford et al. 2006*b*). I allowed samples to air dry before sorting. When dry, I extracted rice and natural seeds with endosperm (i.e., ≥50% of seed remaining). I considered germinated seeds to be potential waterfowl food if the primary shoot was less than or equal to the length of the seed, and if the endosperm was firm (Stafford et al. 2006*b,* Marty et al. 2015). I dried seed samples to constant mass (± 0.5 mg) at 87ºC before weighing to the nearest 0.0001g (Manley et al. 2004, Stafford et al. 2006*b,* Marty et al. 2015).

Statistical Analysis

Seed Biomass in Production, Seed- and Idled Rice Fields

Using data collected during August–November, 2010–2013, I used PROC SURVEYMEANS in SAS v9.4 (SAS Institute 2015) to estimate means for waste-rice and natural seed biomass for each field classification within GCP production, seed-rice, and idled fields. Additionally, I used SURVEYMEANS to estimate mean waste-rice and natural seed biomass for each field classification and within each ecoregion in production and idled rice fields (i.e., LCP, TCP, TMC; Stafford et al. 2006*b*; Marty et al. 2015). I analyzed and reported natural seed biomass using only seeds considered waterfowl foods (Hagy and Kaminski 2012; Table 2.1). Furthermore, I estimated means for waste-rice and natural seed biomass for Clearfield® and conventional seed varieties and field classification within the GCP. I tested for differences in waste-rice and natural seed biomass between rice seed varieties using PROC TTEST in SAS v9.4 (SAS Institute 2015).

I applied size-specific seed bias correction factors to account for rice and natural seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011; Hagy and Kaminski 2012). I partitioned seeds into small (<18 mm^3), medium (18–40 mm^3), and large (>40 mm^3) size classes and applied correction factors of 1.35, 1.10, and 1.07, respectively (Hagy et al. 2011, Marty et al. 2015). I applied correction factors at the core sample level, because it was the level at which most bias was generated (Hagy et al. 2011, Marty et al. 2015). I analyzed data collected under the multi-stage survey design by incorporating appropriate weights and selection probabilities corresponding to the three levels of sampling (Stafford et al. 2006*b*, Marty et al. 2015). The probability of selecting a producer was p_i/P_i, where p_i and P_i were numbers of producers selected and enrolled each year in each stratum i (i.e., GCJV initiative area), respectively. The probability of selecting a field was f_i/F_i, where f_i was the number of fields (1–3) randomly selected among F_i fields farmed by producer i. Finally, the probability of selecting a soil core within a field was (n $cores$)/($C_{ij}/8.107 \times 10^{-7}$), where *n cores* was the number of cores collected in each field and the potential number of cores was the area (C_{ij}; ha) of field$_j$ within a producer$_i$ divided by the area of a core sample (8.107×10^{-7}ha; Stafford et al. 2006*b*). The inverse of the product of the three selection probabilities is the sampling weight used in the SURVEYMEANS procedure. The SURVEYMEANS procedure uses Taylor series linearization to estimate variances for multi-stage surveys (SAS Institute 2015).

Modeling Variation in November Seed Biomass

I evaluated for differences in November waste-rice and natural seed biomass in production rice fields and total seed biomass (i.e., waste rice and natural seed combined) in idled rice fields in relation to various explanatory variables. The GCJV identified 1 November as the starting date of their late planning period, which generally coincides with large increases in waterfowl abundance in the GCP region. Although August estimates of seed biomass are important, November estimates are the primary estimates used by the GCJV to estimate winter carrying capacity. Therefore, I identified potential fixed effect influences on November seed biomass as 1) field classification (FC), 2) soil type (SOIL), 3) precipitation (PRECIP), and 4) seed variety (VAR). I did not include a year effect to avoid potential cofounding with precipitation, and as my goal was to evaluate models representative of all years of my study.

I obtained precipitation data from the National Oceanic and Atmospheric Administration's Applied Climate Information System (NOAA 2016). Precipitation likely created favorable germination conditions for rice and moist-soil seeds in idled rice fields. I did not include a precipitation variable for production rice fields because fields were already flooded. I calculated precipitation as cumulative precipitation from time of first sampling in August until the time of second sampling in November. I did not include a temperature variable because temperature was above 10° C ≥99% of days following the first sampling period through the ratoon harvest, which is the threshold temperature for rice seed germination (Yoshida 1981, Miller and Street 2000). Using USDA's Web Soil Survey, I categorized soil as either clay or loam (USDA 2016). I surveyed rice producers and categorized rice seed variety as either conventional rice or Clearfield®. I developed a set of a priori candidate models, each representing a possible biological scenario for waste-rice and natural seed biomass in production rice fields and total seed biomass (i.e., waste rice and natural seeds) in idled rice fields.

In evaluating November, 2010–2013 waste-rice and natural seed biomass in production and idled rice fields, I used linear mixed models in R (lme4; Bates and Maechler 2016; R Development Core Team 2016). I used mixed effects models because models included the aforementioned fixed effects in addition to a random effect of landowner. I included landowner as a random effect because I sampled only a subset of landowners from a much larger population of GCP rice producing landowners. Inspection of residual plots and histograms indicated that seed biomass were not normally distributed. Subsequently, I natural log transformed seed biomass prior to analysis. I compared models which were selected subsets of the global model using Akaike's

Information Criterion adjusted for small sample size (AICc; Burnham and Anderson 2002), and considered models with ΔAICc ≤ 2 units from the top model as competitive (Burnham and Anderson 2002). I developed models using plausible combinations rice production effects (FC, VAR) and ecological effects (PRECIP, SOIL). When calculating K, I considered fixed and random effects as parameters. I calculated marginal and conditional R^2 statistics as a means to assess the fit of each candidate model (Nakagawa and Schielzeth 2013). I back-transformed estimates from only the most explanatory model. For my top model, I performed pair-wise comparisons of least-squared means (*lsmeans*, Lenth 2016) to test for differences in seed biomass among fixed effects. I considered results statistically significant at $\alpha \leq 0.05$. I did not model average because my goal was to investigate parameter estimates from each supported model, and models contained a random variable of landowner.

Results

Soil Core Sampling Summary Statistics

From August–November, 2010–2013, I analyzed 8,896 soil cores from 196 production, 22 seed-rice, and 200 idled rice fields within the GCP of Louisiana and Texas. I analyzed 5,183, 749, and 2,331 soil cores from production and idled ricelands in the LCP, TCP, and TMC, respectively, during this same time. I also analyzed 633 soil cores from seed-rice fields within the TCP and TMC regions.

Gulf Coast Prairie Seed Biomass Estimates

Production Rice Fields

In 2010–2013 first harvest (FH) production rice fields, waste-rice and natural seed biomass in August were 252.8 kg[dry]/ha (CV = 11%) and 140.0 kg/ha (CV = 13%), respectively (Table 2.2; Figure 2.2). After first harvest, some producers elected to grow and harvest a ratoon crop in November (HR), while others left the ratoon crop standing for subsequent crawfish production (SR), or they did not grow a ratoon crop (NR). In November, 2010–2013 HR production rice fields, waste-rice biomass was 212.2 kg/ha (CV = 21%; 16% decrease), and natural seed biomass increased 31% to 183.5 kg/ha (CV = 16%; Table 2.2; Figure 2.2). In November, 2010–2013 SR production rice fields, waste-rice biomass increased 231% to 837.7 kg/ha (CV = 17%), and natural seed biomass increased 78% to 249.0 kg/ha (CV = 28%; Table 2.2; Figure 2.2). In November, 2010–2013 NR production rice fields, waste-rice and natural seed biomass was 119.3 kg/ha (CV = 19%, i.e., 53% decline) and 103.6 kg/ha (CV = 18%; i.e., 26% decline; Table 2.2; Figure 2.2), respectively. Among ecoregions (i.e., LCP, TCP, TMC) from 2010–2013, waste-rice and natural seed biomass in production rice fields ranged from 32.5 kg/ha (CV = 84%) to 1,022.5 kg/ha (CV = 76%), and 54.9 kg/ha (CV = 49%) to 260.0 kg/ha (CV = 28%), respectively (Appendix A).

Idled Rice Fields

From August–October, 2010–2013, rice biomass in SI fields declined from 15.5 kg/ha (CV = 80%) to 0.3 kg/ha (CV = 97%) and remained negligible from October–November (9.0 kg/ha; CV = 41%; Table 2.2; Figure 2.3). In SI fields, natural seed biomass was 187.2 kg/ha (CV = 12%) in August, 268.9 kg/ha (CV = 24%; i.e., 44%

increase) in October, and 304.8 kg/ha in November (CV = 17%; i.e., 13% increase; Table 2.2; Figure 2.3). In DI fields rice biomass was 3.4 kg/ha (CV = 49%) in August, 0.6 kg/ha (CV = 89%) in October, and 25.5 kg/ha (CV = 69%; Table 2.2; Figure 2.3) by November. In DI fields, natural seed biomass was 162.0 kg/ha (CV = 21%) in August, 477.3 kg/ha (CV = 25%; i.e., 195% increase) in October, and 210.9 kg/ha (CV = 21%; i.e., 56% decline; Table 2.2; Figure 2.3) in November. Among ecoregions from, rice and natural seed biomass in idled fields ranged from 0 kg/ha to 30.7 kg/ha (CV = 72%), and 129.6 kg/ha (CV = 7%) to 521.3 kg/ha (CV = 22%), respectively (Appendix A).

Seed-Rice Fields

In seed-rice fields in Texas (i.e., TMC and TCP) following the first and only harvest (i.e., late July–August, 2012–2013), waste-rice and natural seed biomass were 127.6 kg/ha (CV = 14%) and 45.9 kg/ha (CV = 33%), respectively (Table 2.3; Figure 2.4). In November, waste-rice biomass declined to 54.0 kg/ha (CV = 39%, i.e., 58% decline), and natural seed biomass increased 12% to 51.4 kg/ha, CV = 43%; Table 2.3; Figure 2.4).

Modeling Variation in November Rice and Natural Seed Biomass

Variation in waste-rice biomass in GCP production rice fields during November, 2010–2013 was best explained by the interaction of field classification and seed variety. This model had a w_i of 0.53, and there were no competing models (Table 2.4). The interaction of field classification and seed variety explained 27% of the variation in waste-rice biomass, while the combination of field classification and seed variety as random effects explained 43%. Waste-rice biomass for no ratoon (NR) did not differ between conventional (65.6 kg/ha, 95% CI = 34.7–124.0) and Clearfield® rice varieties (131.2 kg/ha, 95% CI = 89.1–193.1; *z* = 1.873, *P* = 0.419; Table 2.5; Figure 2.5). For harvested ratoon (HR), waste-rice biomass did not differ between conventional (189.8 kg/ha, 95% CI = 135.0–267.0) and Clearfield® rice varieties (116.3 kg/ha, 95% CI = 88.7–152.3; *z* = -2.312, *P* = 0.189; Table 2.5; Figure 2.5). Waste-rice biomass did not differ between conventional (708.4 kg/ha, 95% CI = 385.5–1,301.6) and Clearfield® rice varieties in standing ratoon (SR; 581.7 kg/ha, 95% CI = 351.2–963.7; *z* = 0.495, *P* = 0.996; Table 2.5; Figure 2.5). For Clearfield® varieties, waste-rice biomass in SR fields (581.7 kg/ha, 95% CI = 315.2–963.7) was 5.0 times greater than HR fields (116.3 kg/ha, 95% CI = 88.7–152.3; *z* = -5.717, *P* <0.001) and 4.4 times greater than NR fields (131.2 kg/ha, 95% CI = 89.1–193.1; *z* = -4.788, *P* <0.001; Table 2.5; Figure 2.5). Moreover, for Clearfield® varieties, waste-rice biomass did not differ between HR and NR fields (*z* = 0.522, *P* = 0.995). When producers planted conventional rice varieties, waste-rice biomass in SR fields (708.4 kg/ha, 95% CI = 385.5–1,301.6) was 3.7 times greater than HR fields (189.8 kg/ha, 95% CI = 135.0–267.0; *z* = -3.724, *P* <0.001), and 10.8 times greater than NR fields (65.6 kg/ha, 95% CI = 34.7–124.0; *z* = -5.496, *P* = <0.001; Table 2.5; Figure 2.5). Additionally, for conventional varieties, waste-rice biomass was 2.9 times greater in HR fields than NR fields (*z* = -2.886, *P* = 0.045; Table 2.5; Figure 2.5).

Variation in natural seed biomass in GCP production rice fields was best explained by soil type (Table 2.6). This model had a w_i of 0.23. Other competing models included the null model, an interaction model of field classification and precipitation, an additive model of field classification and soil type, and singular variable models including field classification and rice seed variety. Soil type only explained 2% of the variation in natural seed biomass, and fit for all models was poor ($R^2 \leq 0.10$).

Variation in total seed biomass (i.e., waste rice and natural seeds combined) in idled rice fields in the GCP was best explained by field classification. This model had a w_i of 0.65 (Table 2.7). Field classification explained only 6% of the variation in total seed biomass in idled rice fields, and fit for all models was poor ($R^2 \leq 0.10$). Total seed biomass was 2.2 times greater in SI fields (175.8 kg/ha, 95% CI = 117.6–262.8) than DI fields (78.9 kg/ha, 95% CI = 55.1–112.9; z = -3.583, P = <0.003; Table 2.8; Figure 2.6). Additionally, an additive model containing field classification and precipitation was considered a supporting model, and had a w_i of 0.27. I detected a negative relationship between total seed biomass and precipitation, where seed biomass declined 2% for every 2.54 cm of rainfall.

Gulf Coast Prairie Seed Biomass by Rice Seed Variety

Waste-rice biomass in production rice fields was 2.1 times greater when planted with conventional rice varieties (474.3 kg/ha; CV = 21%) than when planted with Clearfield® varieties (226.0 kg/ha; CV = 18%; $t_{1964} = -7.28$, P <0.001; Table 2.9; Figure 2.7). Moreover, I detected a significant difference in November natural seed biomass in production rice fields planted with conventional (221.9 kg/ha; CV = 20%) and Clearfield® rice varieties (154.3 kg/ha; CV = 14%; $t_{1964} = -5.59$, P = <0.001; Table 2.9; Figure 2.7).

Discussion

Seed Biomass: Gulf Coast Prairie

Production Rice Fields

Rice farming practices differ among regions of the United States, which subsequently influence dynamics of waste rice and natural seeds. In the MAV, growing seasons are shorter and generally one crop of rice is harvested per season (Manley et al. 2004, Stafford et al. 2006*b*). Stafford et al. (2006*b*) reported that waste-rice biomass declined >71% after harvest in July-August to mid-November in the MAV, mostly due to decomposition. Among GCP production rice fields harvested once and not managed to grow a ratoon crop during autumn after initial harvest, waste-rice and natural seed biomass declined 56% and 33%, respectively, from August–November, similar to trends for the MAV (Stafford et al. 2006*b*). However, a 270-day growing season in the GCP is a primary factor affording producers an opportunity to grow a ratoon crop. Both harvested and standing ratoon field classifications influenced November waste-rice and natural seed biomass. Because producers cannot grow a ratoon crop in the MAV, waste-rice biomass remaining in rice fields (78 kg/ha; Stafford et al. 2006*b*) is much lower when compared to harvested and standing ratoon crops in the GCP (212-838 kg/ha; this study).

McGinn and Glasgow (1963) investigated seed loss in rice fields in southwest Louisiana and reported that from mid-September to mid-November 69% and 98% of rice seeds decomposed in dry and flooded fields, respectively. In the MAV, Manley et al. (2004) suggested that earlier maturing rice varieties, resulting in earlier harvest, exposed rice seeds to the environment for longer periods of time in autumn, exacerbating decomposition, germination, and granivory. Stafford et al. (2006*b*) placed enclosures with rice seed into production rice fields in the MAV and found that 20% of the seeds remained intact, 8% germinated, and 14% were consumed. The remaining 58% was unaccounted for and assumed decomposed. Similarly, I placed 40 sealed packets made of window screen, each containing 20 whole rice seeds in GCP production rice fields (*n* = 2 packets per field) following first harvest in August 2013. I collected rice packets prior to the second harvest in early November. Results indicated that in fields which had not been flooded to produce a ratoon crop (i.e., dry, no ratoon fields; *n* = 3), 66% of rice seeds decomposed, 22% germinated, and 12% remained intact as potential waterfowl food. I did not observe any tears or openings in packets, which might have indicated granivory. In fields which had been flooded to produce a ratoon crop (i.e., harvested ratoon and standing ratoon, flooded fields; *n* = 17), 90% of rice seeds decomposed, 7% germinated, and 3% remained intact as potential waterfowl foods. Regardless of field classification, from August–November, in the GCP little seed remained intact and available for waterfowl, which was possibly attributable to decomposition and warmer ambient temperatures compared to that of the MAV. As previously noted, ambient temperatures were above 10° C ≥99 % of days following the first sampling period through the ratoon harvest, which is the threshold for rice seed germination (Yoshida 1981, Miller and Street 2000).

The extended growing period in the GCP, coupled with advancement of earlier maturing rice varieties that began in the mid-late 1960s, have allowed rice producers to successfully grow and harvest ratoon crops (Santos et al. 2003). Ratoon crops apparently mitigate much of the decline in waste-rice biomass that occurs from August–November through decomposition, germination, and granivory. When production fields are flooded, the waste rice that remains following harvest of the ratoon crop provides abundant high energy seed for migrating and wintering waterfowl.

Production rice fields classified as standing ratoon typically contained erect mature rice plants that resulted from fertilizing and irrigation following the first harvest in July–August. Producers typically leave ratoon crops standing if the yield was forecasted as unprofitable, or if they intend to produce crayfish. During the 2013–2014 season, production rice fields accounted for 69% of crayfish pond hectares in the CP (1,165 ha; Foley 2015). The stubble or stalk of rice provides the foundation for the detritus-based food web for crayfish (McClain and Romaire 2004). Production rice fields in the GCP with a standing ratoon crop contained 7 times more rice and 2.4 times more natural seed than fields with no ratoon, and 4 times more rice and 1.3 times more natural seed than fields with a harvested ratoon. Additionally, waste-rice biomass in GCP fields with a standing ratoon was nearly 11 times greater than single harvested rice fields in the MAV. Following the first harvest, if a rice producer elected to grow a ratoon crop, the waste rice remaining in fields would have been available to early migrating and resident waterbirds in fall (e.g., blue-winged teal [*Anas discors*]) during the growing period of the ratoon crop (McClain and Romaire 2004). As the ratoon crop grows and matures, fields are typically flooded to 20-40 cm during winter for crawfish production; these depths could render some waste grain inaccessible to waterfowl. However, rice panicles containing intact rice would likely be available to waterfowl, and conditions found in crayfish fields support aquatic invertebrate communities which are essential protein sources for many waterbird species (Delnicki and Reinecke 1986, Manley et al. 2004, Stafford et al. 2010). In CP rice fields, invertebrate density (40-63 inverts/m^2) can actually be greater than in natural wetlands (17-47 inverts/m^2; Kang 2011, Foley 2015).

Production rice fields are a valuable source of abundant natural seeds (i.e., moist-soil seeds) and tubers, despite significant efforts to control natural vegetation growth. Many natural seed species are consumed by waterfowl and are valuable sources of dietary energy and other nutrients during the non-breeding period. Seeds and tubers persist in the seed bank until germination conditions are favorable. Conditions are typically most favorable during idle (i.e., non-production years) periods, when soils are disked and precipitation creates moist-soil conditions.

Manley (2004) reported a natural seed biomass of 7 kg/ha in the Mississippi MAV, whereas in the previous studies in Louisiana reported variable seed biomass ranging from 42 kg/ha (McAbee 1994) to 973 kg/ha (Hohman et al. 1996). Results from my study indicated that natural seed biomass estimates fell within the aforementioned range among all survey periods and field classifications (104-249 kg/ha). Perhaps natural seed biomass estimates from McAbee (1994) were less than those from my study because of shorter growing seasons and different farming practices in northern Louisiana. Moreover, natural seed biomass estimates reported by Hohman et al. (1996) were likely greater than those derived from my study because of advancements in weed control (e.g., herbicides, rice varieties, water management techniques, etc.).

Idled Rice Fields

In the GCP, ricelands not in rice production during a given year are considered idled and are typically either disked (DI) or contain standing natural vegetation (SI). Rice seed biomass in idled fields was low (i.e., <30 kg/ha) among all field classifications and sampling periods. The presence of rice in idled fields likely originated from plants that grew from waste-rice seeds remaining from the previous production year. Natural seeds (i.e., moist-soil seeds) were the most common seeds observed in idled fields. In idled rice fields, Davis et al. (1961) reported a natural seed biomass of 364 kg/ha in southwest Louisiana, which was greater than most estimates in standing natural vegetation and disked fields among all time periods from my study. Reduction in natural seed biomass could have resulted through increased control efforts through the use of the Clearfield® rice system and other more effective herbicide treatments and weed control techniques than those employed >50 years ago. I observed a general increasing trend in natural seed biomass from August–November in both standing idled and disked idled fields, particularly as seeds matured and dehisced (Reinecke and Hartke 2005, Kross et al. 2008*b*). Many idled fields with standing natural vegetation were disked from August–October, which may have incorporated substantial amounts of natural seed shallowly into the seed bank (Hagy and Kaminski 2012). Rice producers actively disked idled fields to reduce growth of natural vegetation and future competition with subsequent plantings of rice. However, in some cases where farmers did not continue disking fields in fall and winter, disking in summer and early fall may have actually promoted growth of early successional natural plant communities where adequate soil moisture existed (Fredrickson and Taylor 1982, Gray et al. 1999, Kross et al. 2008*b*). In disked fields, natural seed biomass increased from August–October and then declined from October–November. The decline in natural seed biomass in disked fields in late fall was presumably a result of decomposition, granivory, and germination, similar to that which occurred in production rice fields (Stafford et al. 2006*b*).

Seed-Rice Fields

To my knowledge, no research of seed dynamics in seed-rice fields had been conducted in the GCP. In Texas, area of planted seed-rice increased during the early 2000s, peaked in 2011 at 16,796 ha, and has declined to <2,000 ha in 2015. In Louisiana, areas of planted seed-rice has been declined from 6,074 ha in 2005, to 2,221 ha in 2015. However, in the advent of an increase in seed rice production, my study will provide baseline results for conservation planners. Field classifications in seed-rice fields resemble those of a single harvest in the MAV and GCP, where no ratoon crop is grown and the field is idled following the first harvest in August. This contrasts with the more common practice for standard rice production in the GCP ecoregion of growing a ratoon crop following first harvest. I observed a 58% decline in waste-rice biomass and a 12% increase in natural seed biomass in seed-rice fields from August–November. Seed-rice fields were rarely flooded post-harvest, which mostly restricted waterfowl from accessing the limited food resources in these fields by November. Among field classifications and time periods, waste-rice and natural seed biomass in seed-rice fields were always less than in standard production rice fields. After subtracting a giving-up density of 50 kg/ha from biomass estimates in November seed-rice fields, approximately 4 kg/ha of seed biomass would remain as potential waterfowl foods. Thus, an increase in production of seed rice in the GCP would be a cause for concern among conservation planners, as these fields contain less seed biomass and are therefore presumably of less value to foraging waterfowl.

Estimates of Precision

My goal was to estimate waste-rice and natural seed biomass in production, seed-, and idled rice fields with an acceptable level of precision (i.e. CV = ≤15%). Generally, with the exception of waste-rice biomass estimates in FH production and seed-rice fields, and natural seed biomass estimates in FH and August SI fields, I did not achieve that goal. Perhaps lower than desired levels of precision can be attributed to the variability in farming methods within each field classification (i.e., FH, SR, HR, NR, SI, and DI). In other words, within a field classification, rice producers may plant different seed varieties, apply different herbicides or pesticides, use different farming machinery, or apply different levels of treatment intensity. For example, in DI fields, farmers may disk fields once or multiple times per season. Presumably those fields disked multiple times will contain less natural vegetation growth and seed production. Moreover, precision in seed-rice fields in August and November was lower than desired levels likely because of a small sample size (300 soil cores in August, 333 soil cores in November).

Implications for Waterfowl

The GCJV is tasked with providing foraging resources for 14 million ducks and >1.6 million geese annually during the non-breeding period (Esslinger and Wilson 2001. My results indicated that waste-rice and natural seed biomass was greater in production and idled rice fields in the GCP than the MAV (Stafford et al. 2006*b*). Waste-rice and natural seed biomass in GCP production rice fields are as much as 1.5–11 and 15–35 times greater than rice fields in the MAV, respectively (Stafford et al. 2006*b*, Manley et al. 2004). Thus, the normal agriculture practice of producing a ratoon crop in the GCP is a tremendous advantage, which provides abundant waste-rice seed for non-breeding waterfowl that is less attainable in the MAV given current field classifications there.

The GCJV currently estimates that production and idled ricelands account for 44% of the waterfowl carrying capacity in this region (Petrie et al. 2014). The potential to over- or under-estimate energetic carrying capacity is affected greatly by the precision of seed biomass estimates used in bioenergetics models. If current energetic carrying capacity estimates are substantially underestimated, conservation organizations could unnecessarily spend significant amounts of limited funds to meet waterfowl energetic needs. In contrast, if energetic carrying capacity is over-estimated, waterfowl habitat conservation activities may be inappropriately scaled back, leading to a landscape that is insufficient to satisfy the energetic needs of target waterfowl populations. My results indicated that waste-rice and natural seed biomass in FH fields was 85 kg/ha greater, and 18 kg/ha less than estimates currently used in GCJV bioenergetics models, respectively. For HR fields, my results indicated that waste-rice and natural seed biomass was 161 kg/ha less, and 59 kg/ha greater than current GCJV estimates, respectively. For SR fields, my results indicated that waste-rice biomass 644 kg/ha less than estimates used by the GCJV. Current GCJV bioenergetics models do not incorporate a natural seed biomass estimate for SR fields. Additionally, the GCJV aggregates all idled field types into one “idle” classification. My results indicated that August waste-rice and natural seed biomass in SI fields was 127 kg/ha and 115 kg/ha less than current GCJV idle field estimates, respectively. Finally, my results indicated that August waste-rice and natural seed biomass in DI fields was 139 kg/ha and 140 kg/ha less than current GCJV idle field estimates, respectively. Thus, I recommend the GCJV use estimates from this study in their bioenergetics models, as these more contemporary estimates differ from previous GCJV estimates, and are much more representative and precise than my 2010 pilot study (Marty et al. 2015).

Modeling Variation in November Seed Biomass

The interaction between field classification and seed variety best explained variation in November waste-rice biomass in production rice fields in Louisiana and Texas. Waste-rice biomass between rice varieties did not statistically differ within a field classification. Although I was unable to collect the information, harvester age, harvester-operator variation, harvester type, speed at which a field was harvested, field conditions and topography, grain moisture, or moisture on plant surface may have further influenced harvester efficiency and November waste-rice biomass (Wilson et al. 2001, Stafford et al. 2006*b*).

Models predicting natural seed biomass in production rice fields had little explanatory power (i.e., $R^2 \leq 0.10$). Therefore, I could not reconcile influences of measured variables on November variation in natural seed biomass. Poor model fit supported my findings of no detectible difference in November natural seed biomass between soil types (i.e., the best approximating model). Models presumably had poor fit because the selected variables (e.g., soil type, field classification, etc.) were not the dominant factors influencing variation in natural seed biomass. Other non-quantified variables likely influenced November natural seed biomass in production rice fields, including fertilization and herbicide treatments, rice seed varieties, and field planting techniques (i.e., aerial or drill). Development of better models to account for variation in natural seed biomass may potentially be achieved by intensively monitoring a sample of fields prior to the time of planting through November to attain information regarding prior field classifications implemented in the selected field, past and present fertilizer and herbicide treatments, more precise weather data, and any other field classifications which may be applied during the growing season.

The best approximating model predicting variation in November total seed biomass (i.e., waste rice and natural seed combined) in idled rice fields was field classification. Idled rice fields in the GCP which were not planted with row crops, such as soybean, typically contained standing vegetation or were actively disked throughout the year to inhibit natural vegetation. My top model predicted that if producers allowed natural vegetation to grow in idled fields, seed biomass would be significantly greater than in actively disked fields. The growth and development of seeds, and subsequent seeds shattering from the panicle during autumn presumably were what drove the differences in seed biomass differences between idled fields with standing vegetation and disked fields. Repeated disking likely inhibited growth and maturation of natural vegetation, and or buried seeds beneath the zone of sampling (10 cm).

Variety Effect on Gulf Coast Prairie Waste-Rice Biomass

In recent years, anecdotal reports have emerged suggesting ducks and geese may be avoiding ricelands planted with Clearfield® rice varieties. Hypotheses included reduced natural seed abundances because of the more effective weed control afforded by Clearfield® varieties or other traits (e.g., greater pubescence of rice hull associated with hybrid varieties) that may cause them to be less palatable food items. From 2010–2013, >60% of all planted rice in the United States was of a Clearfield® variety. Results from the SURVEYMEANS procedure indicated a statistically greater waste-rice (248.23

kg/ha) and natural seed (67.58 kg/ha) biomass in rice fields containing conventional vs. Clearfield® rice varieties. For both Clearfield® and conventional varieties, waste-rice biomass remained greater than the giving-up density of 50 kg/ha (Greer et al. 2009). For conventional varieties, November, 2010–2013 natural seed biomass was greater than the forage availability threshold of 170 kg/ha (Hagy and Kaminski 2015). However, November, 2010–2013 natural seed biomass in fields planted with Clearfield® rice was below the forage availability threshold of 170 kg/ha. Therefore it is plausible that waterfowl may be avoiding rice fields planted with Clearfield® rice because of reduced waste-rice and natural seed biomass. Clearfield® rice was developed to control and reduce red rice and natural seed production, therefore detecting a difference in natural seed biomass between varieties was not surprising. Perhaps differences in waste-rice biomass was attributed to producer or harvester efficiency, undocumented field treatments (e.g., herbicides, fertilizers, etc.), undocumented use of specific seed varieties within the overarching categories of conventional and Clearfield®, or a sampling anomaly. If the apparent deficit that I detected is real, then perhaps a significantly lesser amount of waste rice and natural seed occurs in Clearfield® fields, which could decrease waterfowl foraging efficiency and overall available energy. Hypothetically, waterfowl would be relegated to increase their time searching for fields planted with conventional rice varieties. If there is additional search time needed to find food resources, there may be possible negative implications related to birds' body mass or survival, which has been discussed in the food-limitation hypothesis (Williams et al. 2014). An expanding landscape of Clearfield® rice might hypothetically impose some of these negative consequences. Partly to this end, I investigated waterfowl use of rice fields planted with Clearfield® and conventional rice varieties (Chapter III).

Research and Management Implications

Aside from fields where no ratoon crop was grown, waste-rice and natural seed biomass remained >200 kg/ha (212–838 kg/ha) among field classifications and sampling periods, which contrasts trends for MAV rice fields (Manley et al. 2004, Stafford et al. 2006*b*). These trends will undoubtedly become increasingly important as restrictions on water usage in the GCP will likely only increase in the future, especially in the Texas growing regions where recent droughts and substantial urban expansions from Houston have occurred (LCRA 2013). For waterfowl, access to abundant rice and natural seeds in GCP ricelands will provide critical foraging resources needed during the non-breeding periods. I recommend that conservation, state, federal, and non-governmental organizations continue to implement and develop programs that help producers become more conservation oriented and efficient (e.g., install more energy efficient water pumps and water control structures). Without financial incentives from conservation programs, rice producers may be less inclined to flood ricelands for waterfowl conservation. I recommend that conservation partners promote programs and policies such as MBHI, which provided valuable wetland habitat for migrating and wintering wetland birds during the nonbreeding period, and subsequently one of the most severe droughts in GCP history. Within GCP rice producing regions, I recommend partners encourage the practice of ratoon cropping, and possibly offering incentives to leave ratoon crops (or portions of them) unharvested. Opportunities to produce ratoon crops are generally not afforded to producers in the MAV or the Central Valley of California because of a shorter growing season in these regions. I also recommend that conservation partners encourage producers to allow early successional vegetation and grasses to grow in idled rice fields as it provides the most natural seed for waterfowl in November. Although disking idled fields hinders the development and maturation of early succession vegetation to produce seed resources for waterfowl, when combined with shallow flooding, this practice may provide valuable invertebrate resources for many shorebirds and wading birds species during the non-breeding period. If disking is necessary, conservation programs should incentivize producers to wait until late October, when natural seeds have matured and dehisced. Importantly, conservation programs should emphasize the importance of, and incentivize producers to shallowly flood (e.g., 1–30 cm) both production and idled rice fields in autumn and winter to provide forage resources for migrating and wintering waterfowl and other waterbirds (Reinecke et al. 1989, Elphick and Oring 1998; Elphick et al. 2010). When shallowly flooded (e.g., ≤15 cm), the aforementioned practices may allow for economic opportunities in the form of hunting and crayfish aquaculture (Grado et al. 2001, 2011; McClain and Romaire 2004; Stafford et al. 2010).

To further increase profits and conserve natural resources, I advocate for conservation programs and policies that encourage implementation of water conservation practices such as closing water control structures, using tail water recovery systems (where feasible), and cost efficient irrigation pumps (Bouldin et al. 2004). Flooding post-harvest and idled ricelands may have economic, environmental, and agronomic benefits. For example, Manley et al. (2009) reported a decrease in export of suspended solids from Mississippi rice fields when farmers flooded standing stubble, versus fields tilled post-harvest. Moreover, Manley et al. (2005) reported that winter flooding could save farmers

\$22–63/ha (USD 2002) in subsequent field preparation costs by reducing stubble biomass by 43-68% and natural vegetation by 24–83%. Interspersion of stubble and open water may be a proximate cue attracting waterfowl to production and idled rice fields (Kaminski and Weller 1992, Havens et al. 2009). Results from Van Groenigen et al. (2003) indicated that foraging waterfowl increased residue decomposition and reduced weed pressure in the rice-growing region of northern California. Furthermore, Bird et al. (2000) reported that intensive foraging by waterfowl in flooded plots decreased straw biomass by 72-76%.

My results indicated that field classification and seed variety best predicted waste-rice biomass for production rice fields in the GCP. I recommend that conservation partners promote programs and policies that encourage rice producers to plant conventional rice varieties because they contained greater biomass of waste rice and natural seed than fields with Clearfield® varieties. I was however, unable to determine if rice varieties were hybrids. Hybrid rice varieties were developed to attain desirable production traits such as improved yield (Linscombe 2015). There is speculation among hunters that waterfowl avoid fields planted with hybrid rice varieties because of pubescent hulls that may be irritating when consumed. Therefore, I recommend that future research investigates potential differences among seed varieties, and how variables such as fertilizer and herbicide treatments affect natural seed biomass in production rice fields, as none of my a priori candidate models explained substantial amounts of the variation.

My spatially and temporally comprehensive study investigating waste-rice and natural seed biomass in GCP ricelands is an important step toward helping conservation planners make necessary amendments to bioenergetic carrying capacity models. Results from my study will allow conservation planners to more precisely estimate carrying capacity, which will enable refinement of habitat objectives and ensure more effective use of limited conservation resources. My results will be of great importance to policy makers, especially given that ricelands, natural wetlands, and marsh ecosystems are becoming increasingly threatened in GCP regions. My results may encourage policy makers to direct funds and promote policies that conserve and promote rice agriculture, and or the restoration of non-rice producing land to native wetlands and prairies. Furthermore, it is clear that the valuable riceland ecosystem in the GCP of Louisiana and Texas provide nutrient rich resources for millions of migrating and wintering waterfowl and other waterbirds annually during the non-breeding season.

Table 2.1 Seed taxa consumed by dabbling ducks in the Gulf Coast Prairies of Louisiana and Texas.

Common name	Taxon	Size classification	Reference[a]
Sedge (seeds)	*Cyperus* spp.	Small	1, 4, 5, 6, 7, 8, 10, 11, 14, 15
Sedge (tubers)	*Cyperus* spp.	Large	2, 14
Crabgrass	*Digitaria* spp.	Small	8, 9, 10
Virginia buttonweed	*Diodia virginiana*	Large	8, 9, 14
Barnyardgrass	*Echinochloa* spp.	Large	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 14, 15
Spikerush	*Eleocharis* spp.	Small	1, 5, 6, 8, 10, 11, 14, 15
Morningglory	*Ipomoea* spp.	Medium	16
Sprangletop	*Leptochloa* spp.	Small	16
Rice	*Oryza sativa*	Large	1, 3, 4, 5, 6, 7, 8, 11, 14, 15
Panicgrass	*Panicum* spp.	Small	1, 4, 5, 6, 7, 8, 9, 13, 14, 15
Dallisgrass	*Paspalum* spp.	Large	1, 5, 6, 7, 8, 9, 11, 15
Swamp smartweed	*Polygonum hydropiperoides*	Medium	3, 4, 5, 6, 7, 8, 9, 10, 13, 14, 15
Curlytop smartweed	*P. lapathifolium*	Medium	3, 9, 10, 13, 15
Pennsylvania smartweed	*P. pensylvanicum*	Medium	3, 7, 9, 10, 13, 15
Beaksedge	*Rhynchospora corniculata*	Large	5, 6, 9, 15
Curly Dock	*Rumex crispus*	Medium	16
Arrowhead	*Sagittaria* spp.	Medium	9
Foxtail grass	*Setaria* spp.	Medium	8, 9, 16
Signal grass	*Urochloa* spp.	Large	4, 6, 8, 9, 15

[a] 1 - Chamberlain (1959), 2 - Combs and Fredrickson (1996), 3 - Dabbert and Martin (2000), 4 - Delnicki and Reinecke (1986), 5 - Dillon (1957), 6 - Dillon (1959), 7 – Forsythe (1965), 8 - Glasgow and Junca (1962), 9 – Hagy (2012), 10 - Heitmeyer (2006), 11 – Martin and Uhler (1939), 12 - Schoffman (1947), 13 - Tabatabai et al. (1983), 14 - Wills (1971), 15 -Wright (1959,) 16- Survey of Gulf Coast biologists.

Table 2.2 Bias corrected estimates of mean waste-rice and natural seed biomass in production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

			Rice biomass			Natural seed biomass		
Sample period	Field classification[a,b]	*n* cores	$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
August	FH	1947	252.77	27.6	10.9	139.98	17.7	12.6
November	SR	368	837.69	140.3	16.7	248.96	70.1	28.2
	HR	1069	212.24	45.3	21.3	183.54	28.9	15.7
	NR	529	119.25	22.1	18.5	103.55	18.3	17.7
August	SI	1016	15.52	12.4	79.7	187.19	22.3	11.9
October		279	0.34	0.3	97.0	268.85	65.4	24.3
November		756	8.97	3.7	40.8	304.77	52.0	17.1
August	DI	850	3.36	1.7	49.2	161.96	34.6	21.4
October		331	0.55	0.5	88.8	477.31	118.2	24.8
November		1118	25.51	17.6	68.9	210.94	43.7	20.7

Sample periods, field classifications, *n* cores, and gross bias corrected estimates[c] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), standard errors (SE), and coefficients of variation (CV; %) for production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

[a] FH, first harvest; HR, harvested ratoon; SR, standing ratoon; NR, no ratoon; SI, standing idle; DI, disked idle.

[b] Blanks denote same field classification.

[c] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011).

Table 2.3 Bias corrected estimates of mean waste-rice and natural seed biomass in seed-rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

Sample period	Field classification[a]	*n* cores	Rice biomass			Natural seed biomass		
			$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
August	FH	300	127.60	18.3	14.3	45.91	15.2	33.2
November	NR	333	53.98	21.2	39.3	51.40	22.0	42.8

Sample periods, field classifications, *n* cores, and gross bias corrected estimates[b] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), standard errors (SE), and coefficients of variation (CV; %) for seed-rice fields in the Gulf Coast Prairies of Texas, August–November, 2010–2013.

[a] FH, first harvest; NR, no ratoon.

[b] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011).

Table 2.4 Results of linear mixed models predicting waste-rice biomass in production rice fields in the Gulf Coast Prairies during November, 2010–2013.

Models[a]	AICc	ΔAICc	w_i	K	*LL*	R^2_{marg}	R^2_{cond}
FC*VAR	542.9	0.0	0.52	8	-263.0	0.27	0.43
FC	545.5	2.6	0.14	5	-267.6	0.24	0.39
FC+VAR	546.2	3.3	0.10	6	-266.8	0.24	0.42
FC+SOIL	546.5	3.6	0.09	6	-267.0	0.24	0.40
FC+VAR	546.7	3.8	0.08	7	-266.0	0.25	0.43
FC+VAR+SOIL+FC*SOIL+FC*VAR	547.2	4.3	0.06	11	-261.8	0.28	0.43
FC*SOIL	550.6	7.7	0.01	8	-266.9	0.24	0.39
VAR	588.7	45.8	0.00	4	-290.3	0.02	0.28
VAR+SOIL	588.8	45.9	0.00	5	-289.2	0.03	0.28
NULL	589.3	46.4	0.00	3	-291.6	0.00	0.24
SOIL	590.0	47.1	0.00	4	-290.9	0.01	0.24
VAR*SOIL	590.9	48.0	0.00	6	-289.2	0.03	0.28

Results of linear mixed models predicting November waste-rice biomass[b] in production rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] Field classification (FC); Soil (SOIL); Variety (VAR); Precipitation (PRECIP); Null model (NULL).

[b] Waste-rice biomass (kg[dry]/ha).

Table 2.5 Estimates of mean waste-rice biomass in Gulf Coast Prairie production rice fields during November by field classification and seed variety, 2010–2013.

Seed variety[a]	Field classification[b]	Mean seed biomass		
		$\bar{x}$	95% LCL	95% UCL
Clearfield®	NR	131.18	89.1	193.1
	HR	116.25	88.7	152.3
	SR	581.73	351.2	963.7
Conventional	NR	65.55	34.7	124.0
	HR	189.84	135.0	267.0
	SR	708.36	385.5	1301.6

Seed variety, field classification, predicted gross November bias corrected estimates[c] of mean ($\bar{x}$) waste-rice (kg[dry]/ha) biomass, and 95% confidence limits from linear mixed models for production rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] Blanks denote same seed variety.

[b] NR, no ratoon; HR, harvested ratoon; SR, standing ratoon.

[c] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011).

Table 2.6 Results of linear mixed models predicting natural seed biomass in production rice fields in the Gulf Coast Prairies during November, 2010–2013.

Models[a]	AICc	ΔAICc	w_i	K	*LL*	R^2_{marg}	R^2_{cond}
SOIL	663.2	0.0	0.20	4	-327.5	0.02	0.32
NULL	663.2	0.0	0.20	3	-328.5	0.00	0.30
FC	664.0	0.8	0.13	5	-326.8	0.02	0.31
FC+SOIL	664.2	1.0	0.12	6	-325.9	0.03	0.33
VAR	665.2	2.0	0.07	4	-328.5	0.00	0.30
VAR+SOIL	665.3	2.1	0.07	5	-327.5	0.02	0.32
FC+VAR	665.8	2.6	0.05	6	-326.7	0.02	0.32
FC+VAR+SOIL	666.2	3.0	0.04	7	-325.8	0.03	0.34
FC*SOIL	666.4	3.2	0.04	8	-324.8	0.05	0.35
FC*VAR	666.5	3.3	0.04	8	-324.8	0.04	0.33
VAR*SOIL	666.7	3.5	0.03	6	-327.1	0.02	0.33
FC+VAR+SOIL+FC*SOIL+FC*VAR	669.4	6.2	0.01	11	-323.0	0.07	0.38

Results of linear mixed models predicting November natural seed biomass[b] in production rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] Field classification (FC); Soil (SOIL); Variety (VAR); Precipitation (PRECIP); Null model (NULL).

[b] Natural seed biomass (kg[dry]/ha).

Table 2.7 Results of linear mixed models predicting total seed biomass in idled rice fields in the Gulf Coast Prairies during November, 2010–2013.

Models[a]	AICc	ΔAICc	w_i	K	R^2_{marg}	R^2_{cond}
FC	728.4	0.0	0.41	4	0.06	0.44
FC+PRECIP	728.2	0.8	0.27	5	0.07	0.45
FC*PRECIP	731.0	2.5	0.12	6	0.07	0.45
FC+PRECIP+SOIL	731.3	2.8	0.10	6	0.07	0.45
FC*SOIL	732.1	3.6	0.07	6	0.06	0.44
FC+SOIL+PRECIP+FC*PRECIP+FC *SOIL+SOIL*PRECIP	733.4	4.9	0.04	9	0.09	0.46
NULL	738.1	9.6	0.00	3	0.00	0.47
PRECIP	739.4	10.9	0	4	0.00	0.47
SOIL	739.9	11.4	0	4	0.00	0.47
SOIL*PRECIP	741.1	12.6	0	6	0.02	0.47
SOIL+PRECIP	741.1	12.7	0	5	0.01	0.47

Results of linear mixed models predicting November total seed biomass[b] in idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] Field classification (FC); Soil (SOIL); Precipitation (PRECIP); Null model (NULL).

[b] Total seed biomass (i.e., waste rice and natural seed combined; kg[dry]/ha).

Table 2.8 Estimates of mean total seed biomass in Gulf Coast Prairie idled rice fields during November by field classification, 2010–2013.

Field classification[a]	Mean seed biomass $\bar{x}$	 95% LCL	 95% UCL
DI	78.90	55.1	112.9
SI	175.79	117.6	262.8

Field classification, November estimates[b] of mean ($\bar{x}$) total seed[c] (kg[dry]/ha), and 95% confidence limits from linear mixed models for idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] DI, disked idle; SI, standing idle.

[b] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011).

[c] Waste rice and natural seed combined.

Table 2.9 Bias corrected estimates of mean waste-rice and natural seed biomass in production rice fields by seed variety in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

			Rice biomass			Natural seed biomass		
Sample period[a]	Seed variety[b]	*n* cores	$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
November	Clearfield®	1277	226.03	39.5	17.5	154.27	21.1	13.7
	Conventional	699	474.26	97.0	20.5	221.85	43.8	19.8

Sample period, seed variety, *n* cores, and gross bias corrected estimates[b] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), and 95% confidence intervals for production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

[a] Blanks denote same sample period.

[b] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians (Hagy et al. 2011).

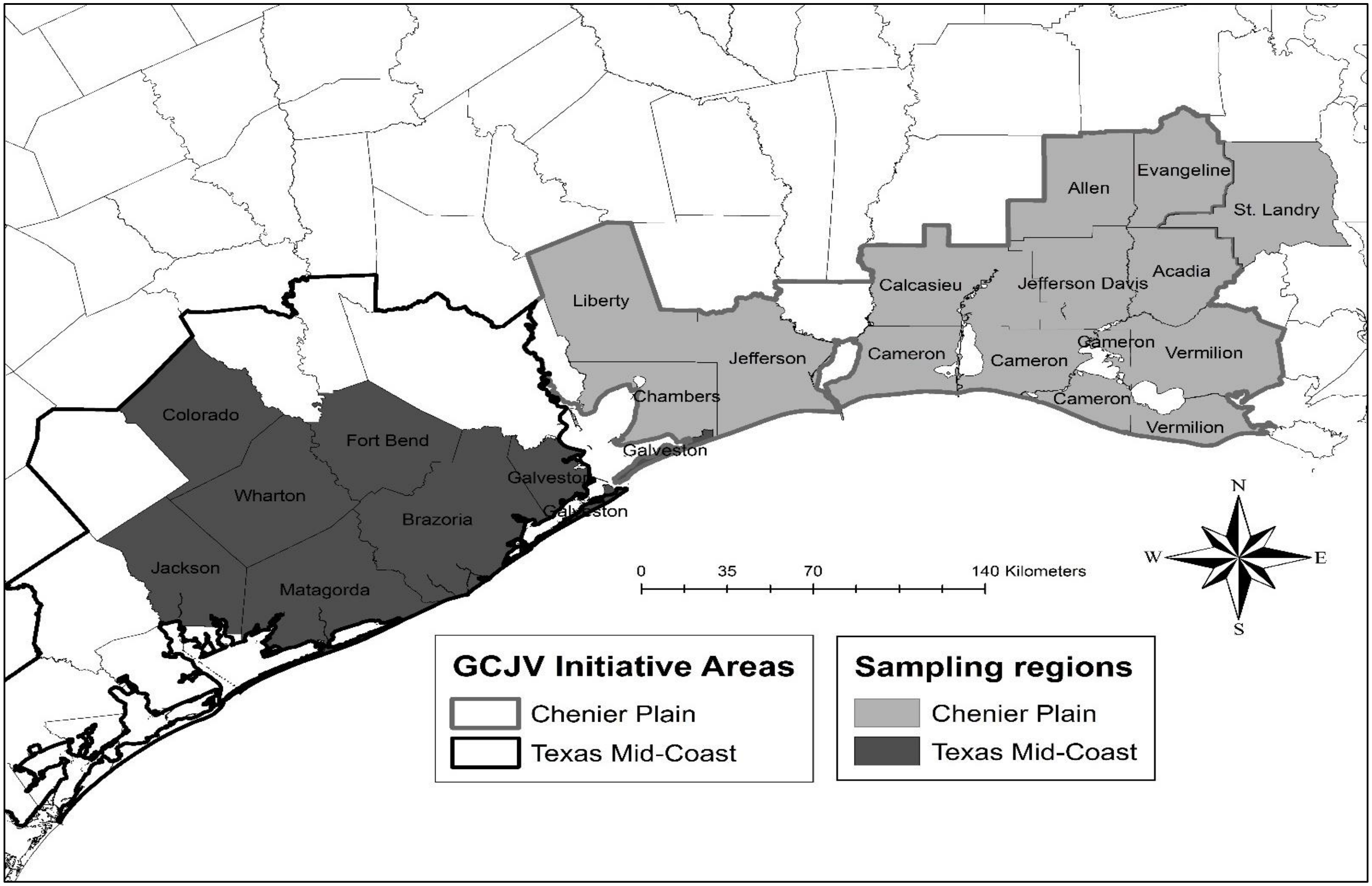

Figure 2.1 Gulf Coast Joint Venture Initiative Areas and sampling regions where soil cores were collected, August–November, 2010–2013.

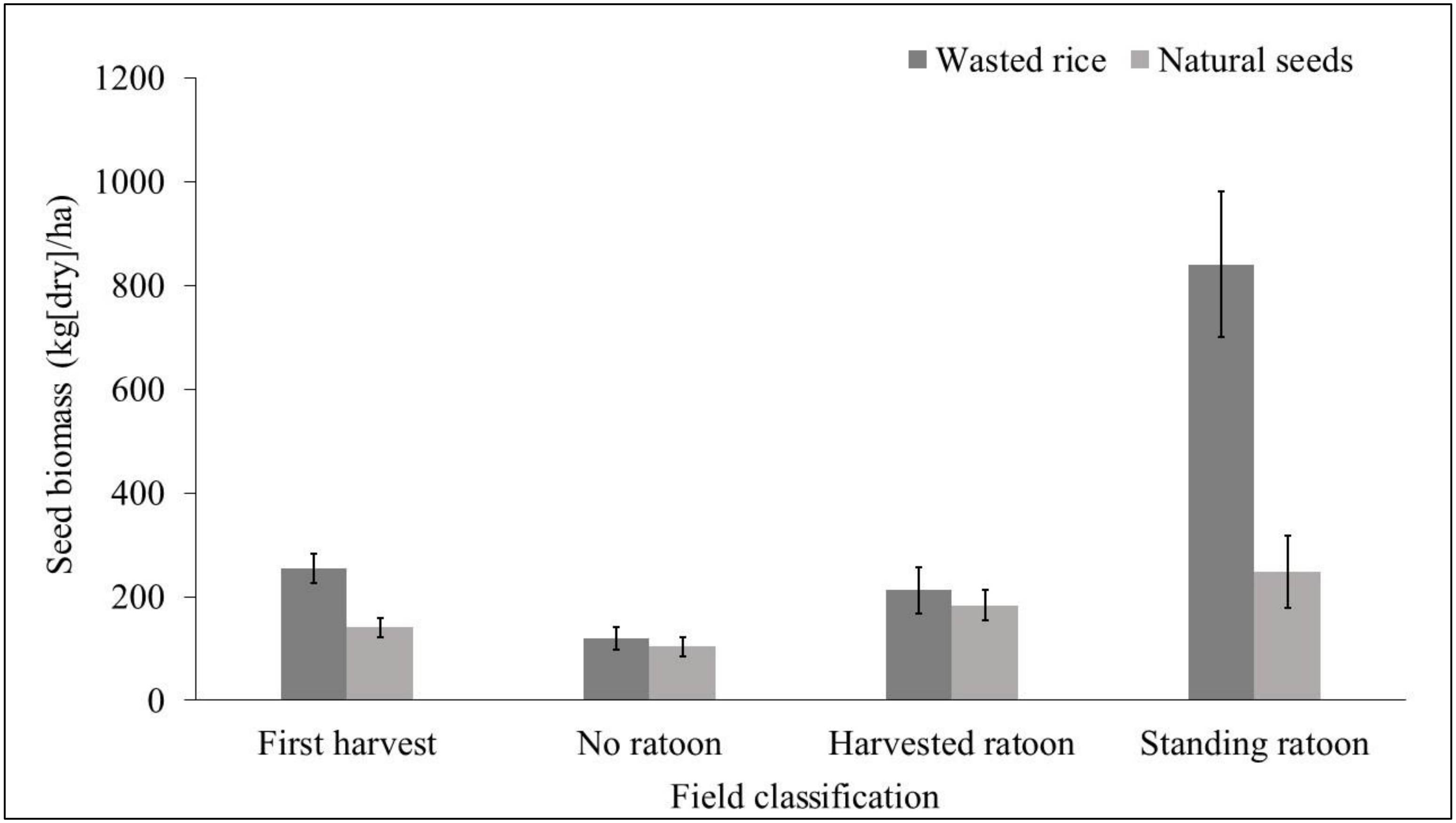

Figure 2.2 Estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) in production rice fields, August–November, 2010–2013.

Bias corrected estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) from PROC SURVEYMEANS, from soil cores (*n* = 3,909) collected in production rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

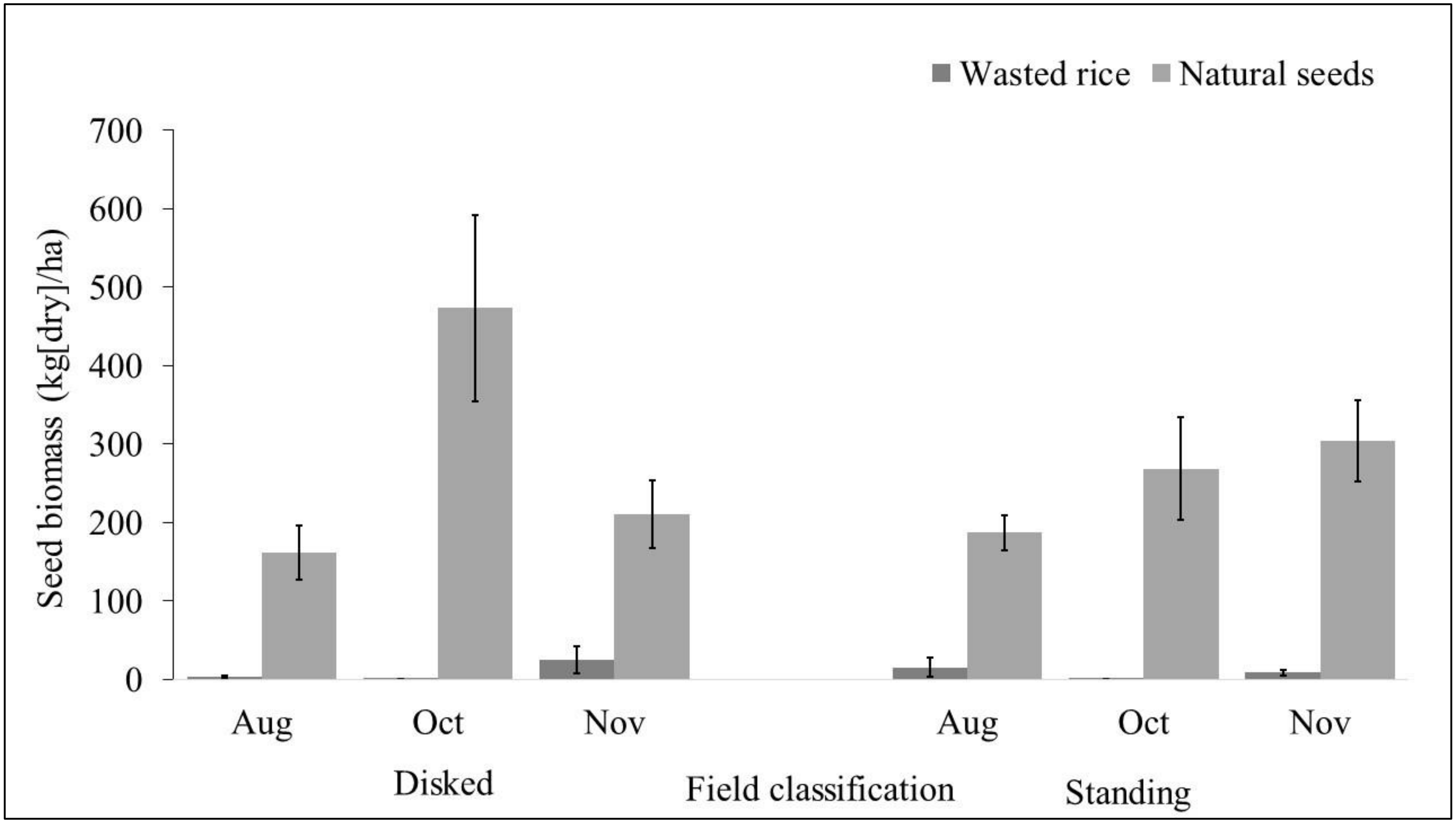

Figure 2.3 Estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) in idled rice fields, August–November, 2010–2013.

Bias corrected estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) from PROC SURVEYMEANS, from soil cores (*n* = 4,350) collected in idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

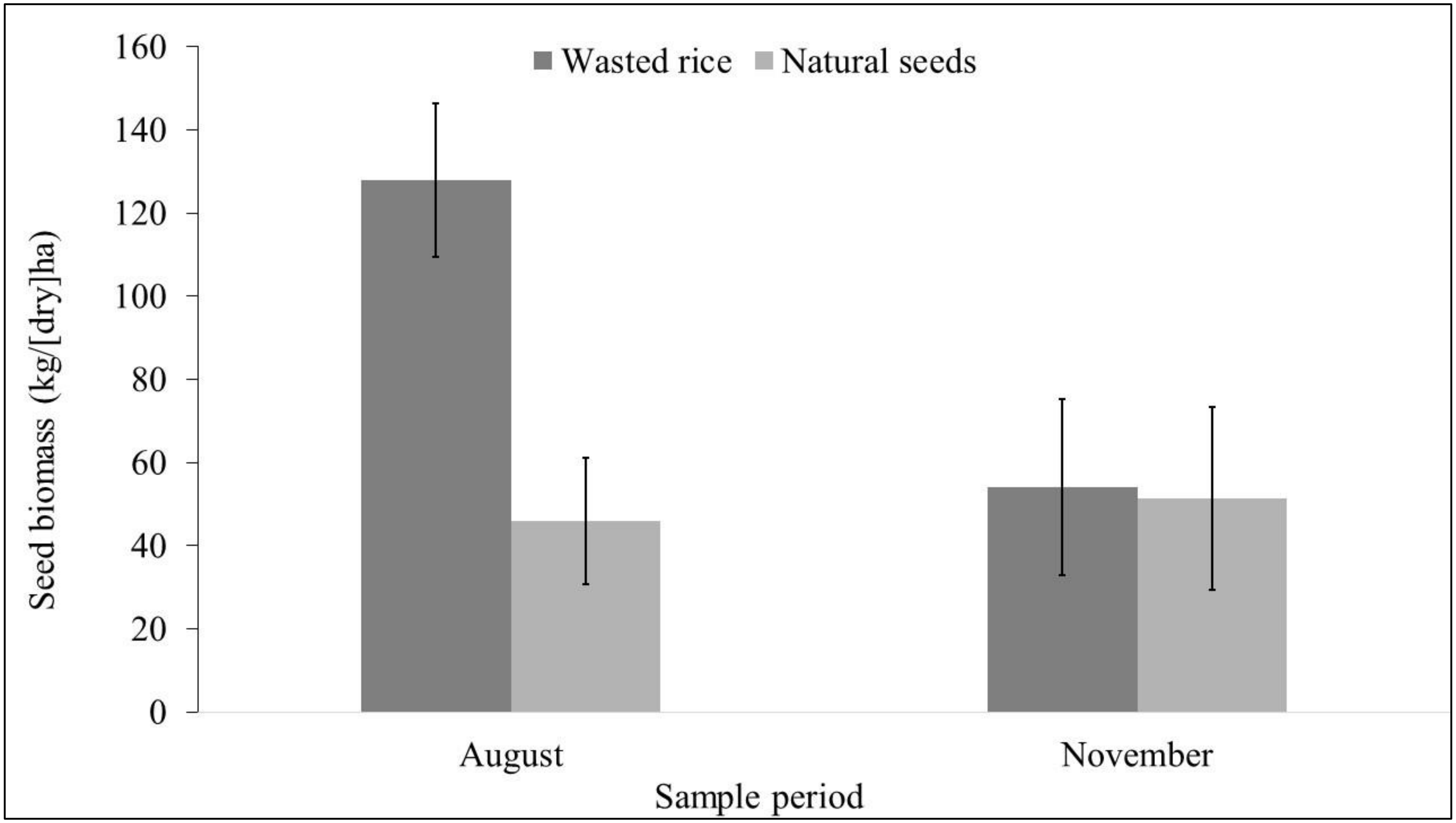

Figure 2.4 Estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) in seed-rice fields, August–November, 2010–2013.

Bias corrected estimates of mean waste-rice and natural seed biomass (kg[dry]/ha; ± SE) from PROC SURVEYMEANS, from soil cores (*n* = 633) collected in seed-rice fields in the Texas Chenier Plain and Texas-Mid Coast, August–November, 2010–2013.

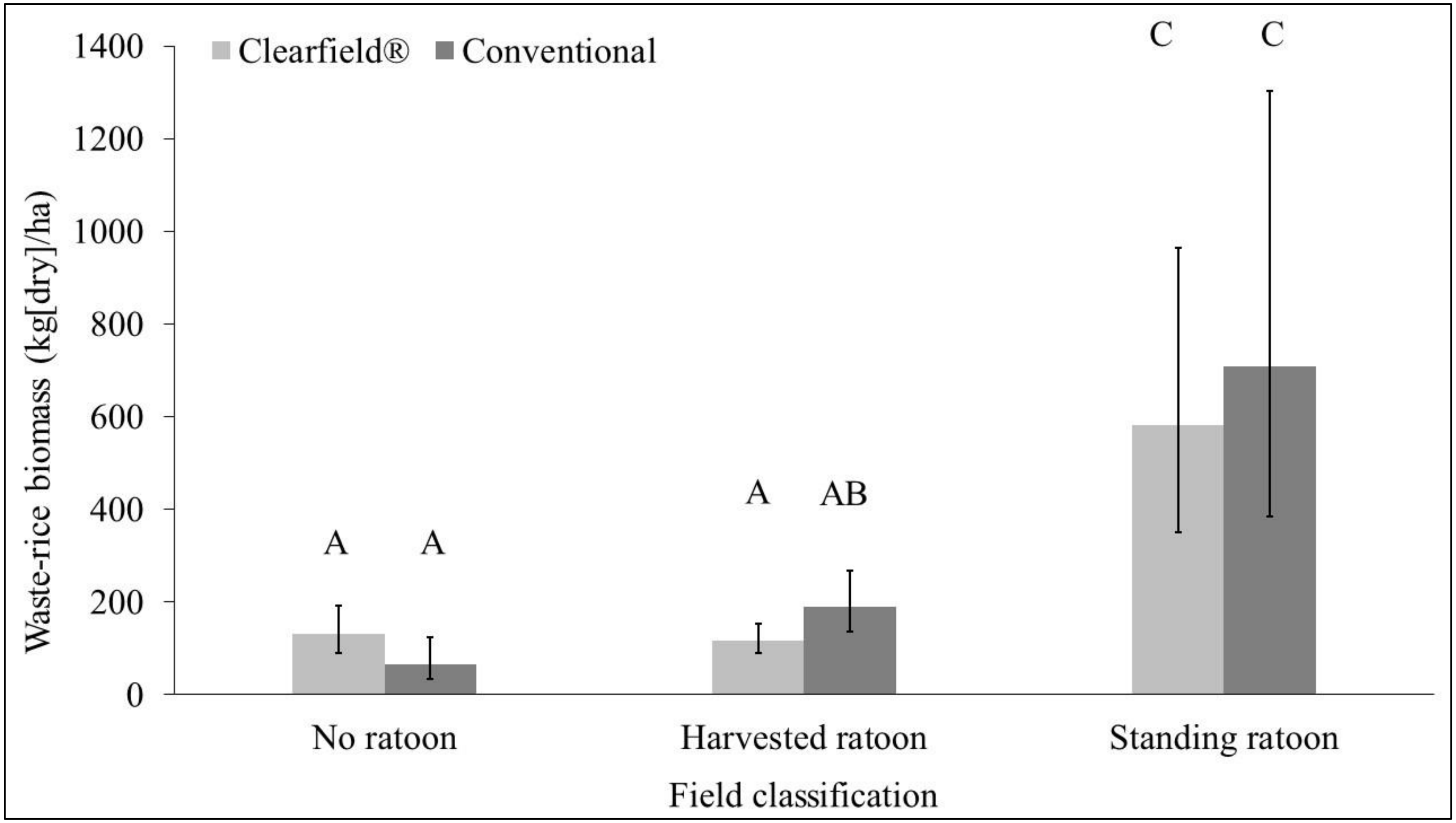

Figure 2.5 Mean waste-rice biomass (kg[dry]/ha) and 95% confidence limits by field classification and rice seed varieties in production rice fields, November, 2010–2013.

Natural log back-transformed mean waste-rice biomass (kg[dry]/ha) and 95% confidence limits by field classification and rice seed varieties (Clearfield® and conventional) from linear mixed models, from soil cores (n = 3,909) collected in production rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013. Different letters indicate significant differences ($\alpha \leq 0.05$)

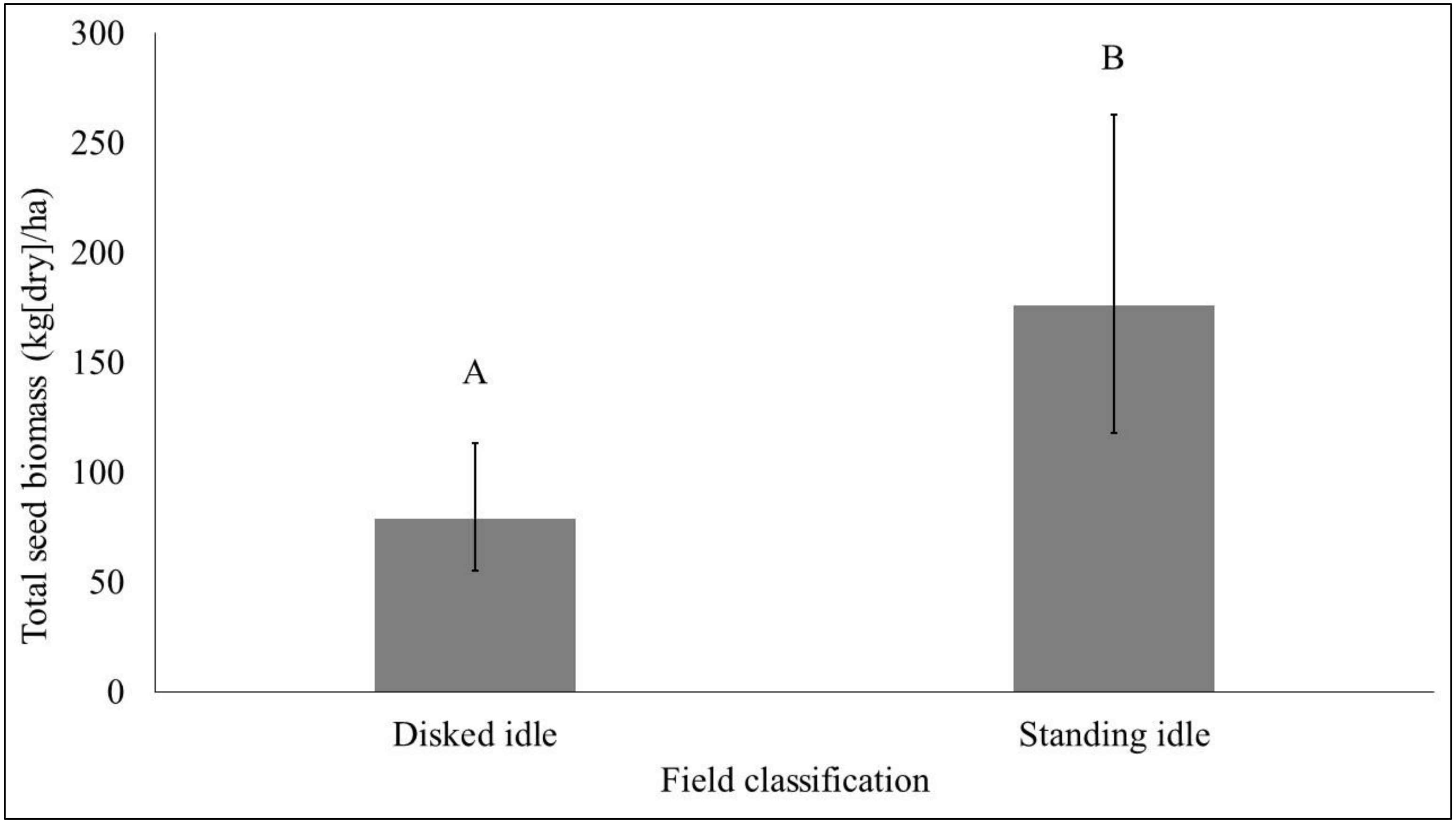

Figure 2.6 Mean total seed biomass (kg[dry]/ha) and 95% confidence limits by field classification in idled rice fields, November, 2010–2013.

Natural log back-transformed mean total seed biomass (i.e., rice and natural seed combined; kg[dry]/ha) and 95% confidence limits by field classification from linear mixed models, from soil cores (*n* = 4,350) collected in idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013. Different letters indicate significant differences ($\alpha \leq 0.05$)

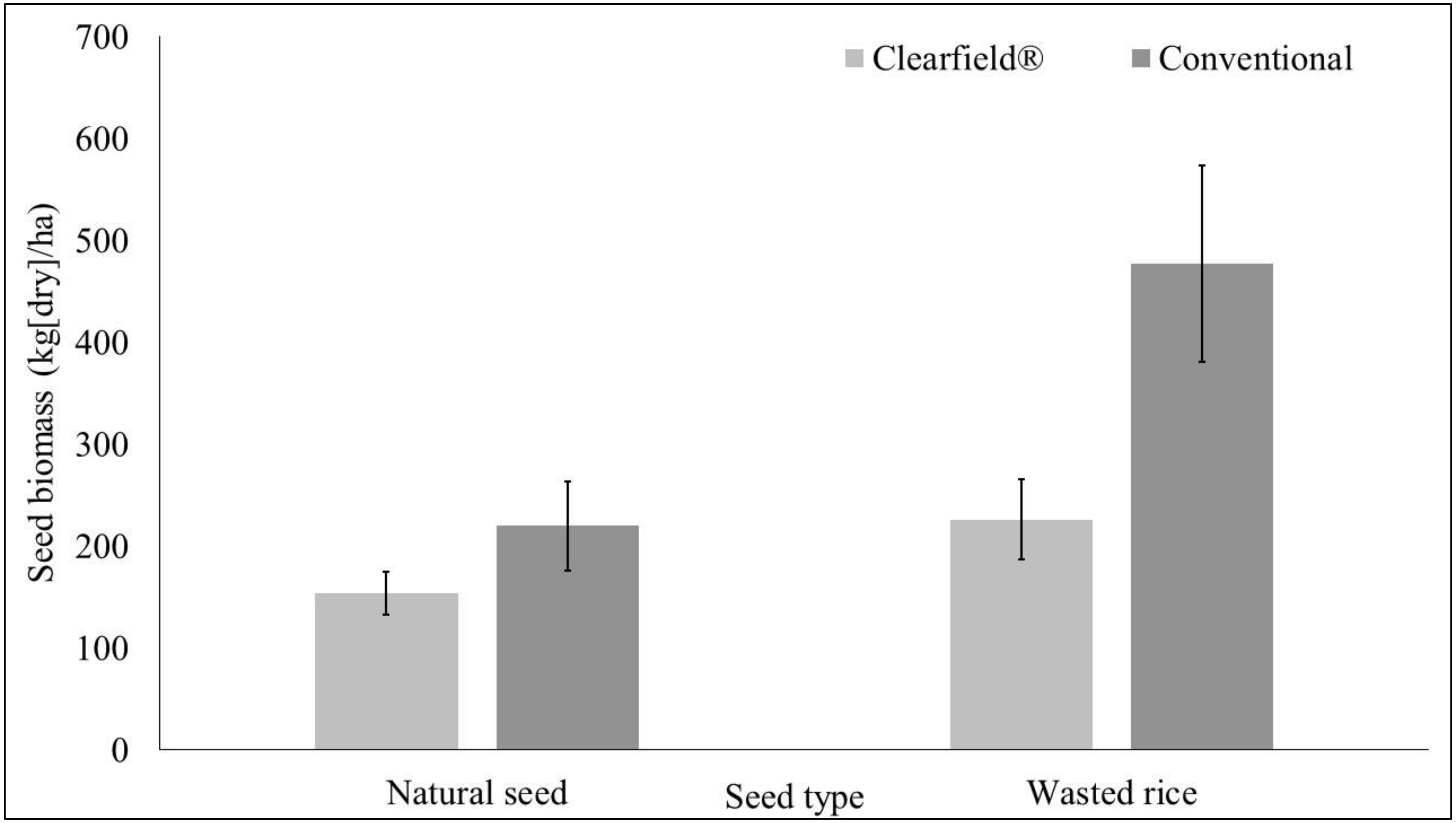

Figure 2.7 Mean waste-rice and natural seed (kg[dry]/ha, ±SE) biomass by rice seed variety, November, 2010–2013.

Bias corrected estimates of mean waste-rice and natural seed (kg[dry]/ha, ±SE) biomass by rice seed variety, from soil cores (*n* = 3,909) collected in production rice fields in the Gulf Coast Prairies of Louisiana and Texas, November, 2010–2013.

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CHAPTER III

WETLAND BIRD USE OF RICELANDS IN THE GULF COAST PRAIRIES OF LOUISIANA AND TEXAS

Since early-20th century, tall-grass prairie and wetlands in the Gulf Coast Prairie (GCP) regions of Louisiana and Texas were converted to agricultural lands, especially for rice production. Rice is grown on irrigated or flooded land, hence this agriculture creates wet croplands that provide breeding, migration, and wintering habitats for waterbirds, including anhingas (Anhingidae); coots, rails, and gallinules (Rallidae); cormorants (Phalacrocoracidae); grebes (Podicipedidae); gulls (Laridae); kingfishers (Cerylidae); pelicans (Pelecanidae); shorebirds (Charadriidae, Recurvirostridae, Scolopacidae); terns (Sternidae); waders (Ardeidae, Threskiornithidae); and waterfowl (Anatidae; Hohman et al. 1994, Elphick 2000, Huner et al. 2002, Eadie et al. 2008, Marty 2013). Thus, previous research has provided a basis for the habitat importance of ricelands to birds worldwide (Elphick et al. 2010*a*).

For example, an estimated 335 bird species (i.e., 169 aquatic and 166 land-bird species) use rice fields in ten world countries (Acosta et al. 2010). In North America, hundreds of bird species use rice fields, which include 28 species of conservation concern (Eadie et al. 2008, Dittmann et al. 2015). Within the GCP region of the United States, the Chenier Plain (CP) of Louisiana (LCP) and Texas (TCP) and the Texas Mid-Coast (TMC) are major rice producing regions that provide habitat for millions of wetland birds annually (Chabreck et al. 1989, Hobaugh et al. 1989, Stafford et al. 2010, Marty et al. 2015). Remsen et al. (1991) observed 260 species of waterbirds using GCP ricelands as wintering habitat in south-central Louisiana. The Gulf Coast Joint Venture (GCJV) endeavors to provide foraging habitat for approximately 14 million ducks, 1.6 million geese, and over 12 million shorebirds annually during autumn-winter, which emphasizes the importance of the GCP to sustain North American waterfowl and wetland bird populations (U.S. Department of the Interior and Environmental Canada 1986, Esslinger and Wilson 2001, U.S. Department of the Interior et al. 2012, Vermillion 2012).

Although ricelands contain rice and some other natural grasses, these croplands are structurally similar to emergent wetlands (Elphick et al. 2000). In the GCP, ricelands uniquely are used often for rice and crayfish (*Procambarus clarkii*) production. These seasonally sequential agricultural practices (i.e., rice followed by crayfish production) create habitats used by resident and migratory wetland birds (Nassar et al. 1988, Reinecke et al. 1989, Fasola and Ruiz 1996, Eadie et al. 2008, King et al. 2010, Stafford et al. 2010). For instance, values of ricefields span from providing nesting substrates for some species (e.g., purple gallinule, *Porphyrio martinicus*; king rail, *Rallus elegans*; fulvous whistling duck, *Dendrocygna bicolor*; Pierluissi et al. 2010), to provision of high energy grain for birds and other wildlife (Kaminski et al. 2003; Elphick et al. 2010*b*, Stafford et al. 2010). Importantly, ricelands provide valuable nesting and brood-rearing habitat for mottled ducks (*Anas fulvigula*), fulvous whistling ducks, and black-bellied whistling ducks (*D. autumnalis*; Durham and Afton 2003, Pickens and King 2012, Baldassarre 2014). Worldwide, approximately 86% of ricelands are shallowly flooded (i.e., <30 cm) at least part of the year (Elphick et al. 2010*b*). Flooded ricelands provide abundant foraging opportunities for wetland birds, because they yield waste rice, natural seeds, tubers, and aquatic invertebrates, as well as habitat for loafing and courtship (Rave and Cordes 1993, Manley et al. 2004, Eadie et al. 2008, Stafford et al. 2010). For example, diurnal activities of northern pintails (*Anas acuta*) in non-hunted rice fields in southwest Louisiana included 21% feeding, 52% resting, 16% comfort movements, and 4% courtship (Rave and Cordes 1993).

Avian community structure and optimal foraging by birds are influenced by food diversity and availability, both of which influence avian life histories (Lack 1954, Hutchinson and MacArthur 1959, Hairston et al. 1960, Emlen 1966, MacArthur and Pianka 1966, Martin 1987). Production and idled rice fields typically contain food resources, access to which may vary dynamically, based on water depth, vegetation height and density, disturbance (e.g., farming and hunting), weather events such as drought, floods, and temperature, seed decomposition, other landscape and local factors (Newton 1998; Schummer et al. 2010; Hagy and Kaminski 2012*a*,*b*; Hagy et al. 2014). Moreover, seed position for avian exploitation in relation to water depth or burial in substrates, naturally renders potential food items unavailable, which influences differences between actual food density and food availability (Boutin 1990, Gawlik 2002).

Across many parts of North America, agricultural lands may be dominant landscape features, but wetlands and uplands form habitat complexes that influence abundance and distribution of wetland birds (Pearse et al. 2012). Gulf coastal rice landscapes generally contain an interspersion of production and idled rice fields, other agricultural lands, natural wetlands, pastures, forest patches, and urban areas that cumulatively also may influence wetland bird abundance and distributions. Developing conservation initiatives and incentives for landowners to promote spatial and temporal flooding of wetlands and production or idled ricelands is an important strategy by conservation partners in the GCP. These directed efforts are needed to meet desired population goals for priority avian and other wildlife species. Sometimes, opportunities to enhance local and regional wetland and agricultural habitat conditions emerge unexpectedly. For example, following the April 2010 Deepwater Horizon Oil Spill in the Gulf of Mexico, the United States Department of Agriculture, Natural Resources Conservation Service (NRCS) established the Migratory Bird Habitat Initiative (MBHI). Part of MBHI's goal was to incentivize private landowners in eight states (Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, Missouri, and Texas) to flood production and idled rice fields and managed wetlands to increase availability of habitats for wetland birds away from potential oil affected areas (Davis et al. 2014). Specifically for ricelands, the primary management practice was to shallowly flood harvested and idled rice fields during autumn and winter in coastal areas of Louisiana and Texas (Davis et al. 2014). Flooding post-harvest production and idled rice fields enrolled in MBHI increased available habitat in the GCP of Louisiana and Texas from 2010-2013 (Kaminski and Davis 2014, Davis et al. 2014). Thus, MBHI created a unique opportunity to assess wetland bird use of riceland management practices promoted by MBHI.

Another unique aspect of my research involved assessment of waterfowl use of production fields planted to Clearfield® Rice. Over 60% of all rice hectares in the United States are planted in Clearfield® rice varieties (Wilson et al. 2010). Clearfield® is non-genetically modified rice that provides selective herbicide resistance to plants, thereby enabling increased control of broadleaf and grass plants in rice fields (Croughan 2003). Despite apparent advantages for producers, there is growing speculation among waterfowl hunters that traits related to Clearfield® (e.g., more effective weed control) are leading to decreased use or avoidance of fields by waterfowl. Although possible, results presented in Chapter I indicate natural seed biomass did not differ between rice seed varieties. When I analyzed for differences in waste-rice biomass between rice varieties, I detected that mean waste-rice biomass was significantly less in fields planted to Clearfield® rice than non-Clearfield varieties. If waterfowl and other granivorous wetland birds use less or avoid fields planted with Clearfield® rice, there could be landscape-scale, carrying capacity implications related to food availability.

Beyond these implications, lingering research needs in the GCP include investigating relationships between wetland bird use of ricelands during autumn and winter and factors such as field classifications, water depths, vegetation height and density, seed variety, and agricultural wetland size. These factors could influence the landscapes capacity to meet needs of millions of wetland birds of conservation interest to the Gulf Coast Joint Venture (GCJV). To address these uncertainties, I conducted diurnal surveys of waterbirds in production, seed-, and idled rice fields in the LCP, TCP, and TMC regions to estimate species richness and abundance of these birds from August–March, 2010–2013. This period was selected because it spanned the rice-harvest, fall-migration, wintering, and spring-migration periods for which MBHI data were desired. My objective was to estimate and model variation in duck and other waterbird (i.e., waders, shorebird, rails, and other birds) species richness and abundance in relation to habitat characteristics and rice-seed varieties of production rice fields, and habitat characteristics of idled rice fields. I hypothesized that diurnal wetland bird species richness would best be predicted by time periods (i.e, month), vegetation characteristics, and water depths. I predicted that variation in duck and waterbird abundances would best be explained by vegetation characteristics, water depth, and time periods; and would occur in shallowly flooded (≤15 cm) ricelands which contained sparse vegetation. Elphick and Oring (1998) indicated that median water depths used by wetland birds ranged from 3-13 cm for shorebirds, and 9-20 cm for herons and ibis. Besides water depth, vegetation characteristics in differently-treated post-harvest rice fields affected wetland bird density (Elphick and Oring 1998, 2003). For example, density was greatest in flooded fields where no vegetation manipulations occurred, and in fields where vegetation was incorporated into the soil by disking (Elphick and Oring 2003). Lastly and specifically pertaining to ducks, I hypothesized that duck abundance in production rice fields would not differ among rice seed varieties. Understanding how this community of wetland birds uses ricelands amid variable seed dynamics and other field treatments (i.e., Chapter I) will improve the overall vision for identifying bottlenecks in habitat needs for conservation planning in the GCP.

Study Area

I conducted my study in agricultural landscapes of the CP of Louisiana and Texas and the TMC. The CP encompasses areas of southwest Louisiana and southeast Texas, roughly spanning from Lafayette, Louisiana, westward to Houston, Texas, and extending inland 130–160 km from the coastline (Figure 2.1). The TMC extends from Galveston Bay to Corpus Christi, Texas, and inland from the coastline approximately 170 km (Figure 2.1). My specific study area included the Louisiana parishes of Acadia, Allen, Calcasieu, Cameron, Evangeline, Jefferson Davis, St. Landry, and Vermilion, and the Texas counties of Brazoria, Chambers, Colorado, Jackson, Jefferson, Liberty, Matagorda, and Wharton. These counties aligned closely with the GCJV's Chenier Plain and Texas Mid-Coast Initiative Areas.

Historically, these regions contained extensive coastal marshes and prairies, freshwater wetlands, and savannahs. Today, the CP and TMC contain coastal marshes along the Gulf of Mexico, but many of the historic coastal prairies and savannas have been converted for cultivation of rice and other crops (Esslinger and Wilson 2001). The climate is sub-tropical and humid with an average growing season of 270 days, 13 freeze-days per year, and temperatures ranging from 14° C in December–January to 30° C July–August (Chabreck et al. 1989). Average annual precipitation decreases east to west in the CP from 144 cm near Lafayette, Louisiana, to 113 cm near Houston, Texas, and 77 cm near Corpus Christi, Texas (Gosselink et al. 1979, Hobaugh et al. 1989). The CP and TMC regions are subject to frequent and sometimes intense weather disturbances; on average, tropical storms make landfall approximately once every 1.6 years and hurricanes every 3.3 years (Roth 1999).

Methods

Wetland Bird Surveys

I initially surveyed wetland birds from December–March 2010–2011 in response to the MBHI (Marty 2013). Subsequently, I conducted avian surveys from August–March, 2011–2013 to acquire data from bird migration and winter periods important to GCP conservation planning. My populations of surveyed fields included those enrolled in MBHI, the GCJV Texas Prairie Wetlands Project (TPWP), and agricultural fields managed similarly to practices promoted by MBHI. Specifically, I conducted wetland bird surveys in the same randomly selected production, idled, and seed-rice rice fields from which I collected soil cores (Chapter II). The combination of these fields and potential food resources were believed to be representative of common agricultural land management practices in the GCP (S. Linscombe, Louisiana State University Agricultural Center, personal communication). Field classifications of production and idled rice fields included: 1) July–August harvest only (first harvest, FH); 2) fields harvested in August and again in November for a ratoon crop (harvested ratoon, HR); 3) fields in which a second crop was grown but not harvested and left standing, generally for crawfish aquaculture or waterfowl habitat (standing ratoon, SR); 4) fields harvested in July–August but with no ratoon crop grown (no ratoon, NR); 5) idle fields with standing natural vegetation (standing idle, SI); and 6) disked idled fields (disked idle, DI). Application of these field classifications was not mutually exclusive. For example, all production rice fields were harvested July–August, but each was subject to one of several unique practices (e.g., classifications 2–4) that affected field dynamics (e.g., food dynamics, water depth, vegetation conditions) during autumn. Thus, some of my identified field classifications are best viewed as a combination of farming activity and sampling period.

I surveyed birds from one or multiple vantage points, following guidelines from the Integrated Waterbird Management and Monitoring Program ([IWMMP]; IWMMP 2010, 2015). I estimated abundance of wetland birds (total birds/species/survey), because ricelands typically contained vegetation and levees, which in some instances created visual obstructions preventing me from detecting all birds present. To minimize multiple counting of individual birds, I visually followed flushed birds and noted their location if they alighted in areas yet to be surveyed (Kaminski and Prince 1981, Fleming et al. 2015). I conducted surveys from sunrise to sunset and only in favorable weather (i.e., not on days with fog, rain, and winds >20 mph; O'Neal et al. 2008, Fleming et al. 2015). Immediately after conducting a survey, I measured water depth, vegetation height, and vertical vegetation density at two randomly selected sites within each field (Robel et al. 1970). I created classes for water depth and vegetation height and density (sensu IWMMP 2010, 2015). Water depth classes included saturated soil (<1 cm), shallow (1–15 cm), intermediate (15–30 cm), and deep flooded (>30 cm). Vegetation height classes included none, short (1–15 cm), intermediate (16–40 cm), and tall (>40 cm). Vertical vegetation density classes included none, sparse (1–20 cm), intermediate (21–40 cm), and dense (>40 cm). I visually estimated percent coverage of water in each field during each visit and used ArcMap10 to calculate wet area (ha) of each field.

Statistical Analysis

Modeling Variation in Wetland Bird Richness

I evaluated for differences in seasonal wetland bird (Ducks and Waterbirds) species richness across production and idled rice fields in the GCP in relation to various explanatory variables. I included variables that may explain variation in wetland bird species richness. My objective was to explain variation in wetland bird species richness in saturated–flooded wetland areas, thus I excluded portions of fields during surveys that were dry and the entire survey if a field was completely dry for this and subsequent analyses. In evaluating wetland bird species richness, I used linear mixed models in R (lme4; Bates and Maechler 2016; R Development Core Team 2016). I used mixed effects models because models included fixed and random effects. I identified the following covariates as fixed effects for wetland bird species richness: 1) month, 2) water depth, 3) vegetation height, 4) vegetation density, and 5) wetland size (i.e., area of field surveyed). I included year as a random effect because of yearly variability in bird migration and distribution within the Mississippi Flyway. I natural log transformed species richness data prior to analysis, because inspection of residual plots and histograms indicated data were not normally distributed. I included year as a random effect, because evidence (i.e., lowest AICc) suggested it increased explanatory power of my models (Zuur et al. 2009). I developed a set of a priori candidate models, each representing a possible biological scenario for wetland bird species richness. I did not include precipitation variable because all survey fields included in the analysis contained flooded agricultural wetlands, and I reasoned that if rainfall created ephemeral wetlands in fields, my surveys would capture birds using these and be categorized in saturated soil or shallowly flooded categories. I compared models using Akaike's Information Criterion adjusted for sample size (AICc; Burnham and Anderson 2002), and considered models with ΔAICc ≤2 units from the top model as competitive (Burnham and Anderson 2002). I calculated R^2 statistics as a means to assess the fit of each candidate model (Nakagawa and Schielzeth 2013). I calculated back-transformed estimates from only the best supported model. I did not model average because my goal was to investigate parameter estimates from each supported model, and models contained a random variable of year.

Modeling Variation in Duck and Waterbird Abundance

I separated wetland birds into two guilds: 1) Ducks and 2) Waterbirds (waders, shorebirds, rails, and other [e.g., *Grus americana*, *Larus* spp., *Podilymbus* sp.]). I did not separate shorebirds from wading birds because of sample size limitations. I excluded dry areas of fields and the entire survey if the field was completely dry, as described above. I excluded geese from analyses because they were observed infrequently (i.e., 2% of all surveys across years, n = 5,002). Additionally, I excluded seed-rice fields from analyses, because they were dry in 80% of all surveys across years (n = 338) and never flooded >1 cm.

Because birds were not detected in all fields during many surveys, I used zero-inflated Poisson, zero-inflated negative binomial models and Hurdle models. I compared AICc and Bayesian information criterion (BICc) values from the null model for both Ducks and Waterbirds. Results indicated that a zero-inflated negative binomial model was most appropriate for my Duck count data, and a negative binomial Hurdle model was best suited for the Waterbird data. Therefore, I used zero-inflated negative binomial regression model (*pscl*; Jackman 2015) to assess variation in Duck abundance and a negative binomial Hurdle regression model (*pscl*; Jackman 2015) for Waterbird abundance.

Zero-inflated and Hurdle regression models combine a standard discrete distribution (e.g., negative binomial; count data), with the binomial distribution (zeros present in greater number than predicted by the discrete distribution; Ridout et al. 1998). Multiple processes such as false negatives (zeros; e.g., when habitat is suitable and the observer fails to detect an organism that is actually present, or when the habitat is suitable but the organism is not present), and true zeros (e.g., when habitat is not suitable, thus the organism is not observed) are responsible for zeros in the response variable (Zuur et al. 2009). Zero-inflated regression models (i.e., mixture models), model false zeros separately from non-zero counts and true zeros (Zuur 2009). Whereas, a Hurdle model contains two processes; the first, models the occurrence of a zero (true and false) vs. non-zero counts; the second, the relationship between non-zero counts and covariates (Zuur et al. 2009).

I identified the following factors or covariates as potential influences on wetland bird abundance: 1) year, 2) month, 3) water depth, 4) vegetation height, 5) vegetation density, and 6) wetland size (i.e., area of field surveyed). I developed a set of a priori candidate models, each representing a possible biological scenario for Ducks and Waterbirds. I compared models using Akaike's Information Criterion adjusted for sample size (AICc; Burnham and Anderson 2002), and considered models with ΔAICc ≤ 2 units from the top model as competitive (Burnham and Anderson 2002). I calculated back-transformed estimates from the best supported model. To assess variation in duck abundance in relation to rice seed variety (Clearfield® vs conventional varieties), I back-transformed estimates from the "variety" model as described above.

Results

Wetland Bird Species Richness

I conducted 5,002 wetland bird surveys in 142 fields in the LCP, TCP, and TMC regions during August–March, 2010–2013 (i.e., production [2010, n = 10; 2011–2013, n = 50], idled [2010, n = 10; 2011–2013, n = 50], and seed-rice rice fields [2012, n = 10; 2013, n = 12]). Of the 5,002 surveys, 60% (2,996, [DI, n = 632; SI, n = 610; NR, n =

419; HR, *n* = 540; SR, *n* = 384]) contained wet ricelands and the remaining 40% (2,006) were dry. I observed the following number of species by taxon: 20 waterfowl, 9 shorebirds, 14 waders, 3 rails, and 7 species of other birds (Table 3.1). Greatest encountered wetland bird species richness (13) during all surveys occurred in idled (*n* = 4 surveys) and production (*n* = 1 survey) rice fields. Among all surveys, greatest waterfowl (ducks and geese) species richness (9) observed was in a rice field with no ratoon crop, and greatest Waterbird species richness (10) occurred in first harvest (*n* = 2 surveys) and harvested ratoon (*n* = 1 survey) rice fields.

Variation in wetland bird (Ducks and Waterbirds) species richness across my GCP survey region was best explained by an additive model containing vertical vegetation density, water depth, and wetland size (Table 3.2). The combination of vegetation density, water depth, and wetland size explained 10% of the variation in wetland bird species richness. When holding wetland size constant at the computed average of 17.9 wet ha (hereafter ha), species richness was greatest in ricelands with shallow water depth and sparse vertical vegetation density (3.5 wetland birds/survey, 95% CI = 3.1–3.8) and least in saturated ricelands with dense vertical vegetation (1.8 wetland birds/survey, 95% CI = 1. 6–1.9; Table 3.3). Species richness increased ~1% for each 1 ha increase in wetland size (Figures 3.1–3.4).

Wetland Bird Summary Statistics

I detected 456,565 wetland birds across all species during the aforementioned 2,996 surveys of wet ricelands. Despite great wetland bird abundance, I did not detect any birds in 31% of these surveys. Lesser snow geese (*Chen caerulescens*) was the most abundant bird species observed among dry and flooded ricelands (*n* = 65,546). This species was observed only in 66 (1%) of the 5,002 total surveys. Ducks and geese collectively accounted for 62% (281,070) of all wetland bird observations (456,565), while waders, shorebirds, rails, and others represented 17% (79,166), 17% (77,004), 3% (12,491), and 1% (6,834), respectively. Greatest duck density for an individual survey occurred in mid-February, in a LCP disked idled field with intermediate water depths (601 ducks/ha). This field contained 7,200 American green-winged teal (*Anas crecca*), 515 northern pintail (*A. acuta*), and 6 mallards (*A. platyrhynchos*). Greatest density of waders (223 birds/ha) for an individual survey occurred in mid-January, in a LCP production rice field with a standing ratoon crop of tall, dense rice flooded to an intermediate depth. This field contained 1,240 white ibis (*Eudocimus albus*). Moreover, greatest shorebird density for an individual survey was (312 birds/ha) in mid-December, in a saturated TMC disked idled field without vegetation. This field contained an estimated 100 sandpipers (*Calidris* spp.) and 1,500 dowitchers (*Limnodromus* spp.). Wetland bird abundance in seed-rice fields remained low among all surveys (0–1.4 birds/ha) as seed-rice fields were seldom flooded during winter. Thus, I did not include seed-rice fields in abundance analyses.

Variation in Duck Abundance

Variation in Duck abundance was best explained by an additive model that included vegetation height, water depth, and wetland size. This model had a weight (w_i) of 0.75 (Table 3.4). Holding wetland size constant at the computed average of 17.9 ha for all modeling analyses, duck abundance was greatest in ricelands with intermediate water depths and short vegetation (447.3 ducks, 95% CI = 264.0–757.7), and least in ricelands with saturated soils and intermediate vegetation height (14.6 ducks, 95% CI =

7.0–30.5; Table 3.5). In shallowly flooded ricelands, greatest duck abundance occurred with short vegetation (360.0 ducks, 95% CI = 216.8–597.9; Table 3.5). Duck abundance in deeply flooded and saturated ricelands remained low, but within deeply flooded ricelands was greatest in fields with short vegetation (i.e., vegetation height extending above the surface of the water) 73.3 ducks (95% CI = 39.4–136.4; Table 3.5). Duck abundance decreased 0.86% for each 1 ha increase in wetland size (Figure 3.5–3.8).

The probability of measuring a false negative (false zero), versus counts of detected birds and true zeros was greatest in ricelands with saturated soils and tall vegetation height (97%, 95% CI = 95–98%), but least in ricelands with shallow water and short vegetation height (20%, 95% CI = 9–40%; Table 3.6). The probability of measuring a false negative decreased 1.8% for each 1 ha increase in wetland size (Figures 3.9–3.12).

Variation in Waterbird Abundance

Variation in abundance Waterbirds was best explained by an additive model that contained field classification, water depth, and wetland size. The weight (w_i) of this model was 0.88 (Table 3.7). Waterbird abundance at the average wetland area (17.9 ha) was greatest in shallowly flooded fields with sparse vegetation (83.3 Waterbirds, 95% CI = 56.4–122.9) and least in saturated fields with dense vegetation (17.3 Waterbirds, 95% CI = 12.4–24.0; Table 3.8). Waterbird abundance increased ~1% for each 1 ha increase in wetland size (Figures 3.13 – 3.16).

The probability of a riceland (e.g., production or idled rice field) being used by Waterbirds was greatest in shallowly flooded fields with no vegetation (76%, 95% CI = 72–01%) and least in saturated fields with dense vegetation (56%, 95% CI = 52–61%;

Table 3.9). The probability of a riceland being used by waterbirds increased ~2% for each 1 ha increase in wetland size (Figures 3.17–3.20).

Variation in Duck Abundance Relative to Rice Seed Variety

Duck abundance did not differ between Clearfield® (65.4 ducks, 95% CI = 42.8-99.8) and conventional rice varieties (73.1 ducks, 95% CI = 43.8-122.1; Table 3.10; Figure 3.21). Moreover, the probability of measuring a false negative did not differ and was 51% (95% CI = 35-68) and 49% (95% CI = 31-67) for Clearfield® and conventional rice varieties, respectively (Table 3.11; Figure 3.22).

Discussion

Wetland Bird Species Richness

Shallow water (≤20 cm) provides foraging opportunities for the greatest number of wetland bird species (Elphick and Oring 1998, 2003), and fields devoid of or containing intermediate levels of vertical vegetation may have been important for foraging efficiency and predator detection or avoidance. Elphick and Oring (1998, 2003) suggested that water depths ranging from 10–20 cm are preferred for wetland bird management, with the lower end of the range excluding fewer wetland bird species than the upper end. Additionally, Hagy and Kaminski (2012*b*) reported ~90% of dabbling ducks foraged in managed moist-soil wetlands flooded <16 cm deep in western Mississippi. In Louisiana, Rettig (1994) found that 70% of shorebirds and 50% of wading birds used wet fields with less than 50% vegetation cover. Vegetation manipulations, such as disking, rolling, chopping, or mowing, are potential sources of variation in wetland bird use of ricelands. My results indicated that species richness was lowest when fields contained dense vertical vegetation. Some avifauna utilize flocking to increase their feeding efficiency or decrease their vulnerability to predators (Powell 1974, Morse 1977, Cresswell 1994). Perhaps dense vegetation precluded use for many flocking shorebird, wader, and waterfowl species because of visual and mobility obstructions. However, because of the secretive nature of many wetland birds which inhabit dense vegetation (e.g., bitterns, rails, etc.), detection is often difficult even when suitable habitat is surveyed and birds are present (Allen 2004, Conway 2005, Valente 2009). Therefore, some species may have been present but I could not detect them.

Wetland bird species richness increased ~1% for every hectare increase in wetland size. Larger agricultural wetlands likely contained a greater diversity of foraging habitats and food resources, facilitating use by a greater number of wetland bird species. Numerous hypotheses and theories have been posited to explain the species-area relationship. The species-area relationship, originally proposed by Arrhenius (1921), suggests that more species occur in larger than smaller areas. MacArthur and Wilson (1967) advanced this concept by developing the equilibrium model of species-area relationships on islands (i.e., the theory of island biogeography), postulating that smaller islands support fewer species than larger islands. Additionally, the habitat diversity hypothesis states that large areas have greater habitat diversity than small areas, and thus should contain more species (Williams 1943). The passive sampling hypothesis argues that larger areas should be greater 'targets' for immigration and subsequently contain more species (Coleman et al. 1982). Relating the habitat diversity and passive sampling hypotheses to my study, the diversity of agricultural practices and resulting habitat mosaics created by rice and crayfish production, and waterfowl and other wetland bird conservation create a diversity of important habitats for diversity of wetland avifauna. Below, I discuss avian communities more specifically as they relate to different types of field classifications, vegetation structure, and water depths in my study.

Duck Abundance

Duck abundance was best explained by an additive model containing vegetation height, water depth, and wetland size. Greatest duck abundances occurred in ricelands with shallow or intermediate water depths and short vegetation. My results approximate those for other rice agricultural systems in California and the Mississippi Alluvial Valley, where median water depths used by dabbling ducks in California rice fields ranged from 14–22 cm (Elphick and Oring 1998), and most dabbling ducks in the Mississippi Alluvial Valley foraged in <16 cm (Hagy and Kaminski 2012b). Shallow water depths allow ducks to access important food resources, such as waste rice, natural seeds, tubers, and aquatic invertebrates present in production and idled rice fields and managed moist-soil wetlands (Manley et al. 2004; Stafford et al. 2006; Kross et al. 2008a,b; Stafford et al. 2010, Hagy and Kaminski 2012a,b; Marty et al. 2015).

Idled ricelands which are frequently disked, and ricelands flooded for extended periods, often contain little to no vegetation. Additionally, disking incorporates plant biomass into the soil. Furthermore, harvesting a rice field involves clipping the rice stalk, which often reduces vegetation height across the entire field. During harvesting of production rice fields, openings are created when rice stalks are flattened by farm machinery. Flooding of production and idled ricelands promotes decomposition of plant biomass and provides landowners economic and agronomic benefits (Manley et al. 2005, Anders et al. 2008). Moreover, foraging actions by ducks in flooded ricelands exacerbate straw decomposition in winter (Smith 1992, Brouder and Hill 1995, Bird et al. 2000). My results revealed that ricelands with intermediate and tall vegetation typically attracted fewer ducks than those with no or short vegetation. Ducks tend to avoid fields with tall, dense vegetation until it decomposes, topples, or openings are otherwise created because of reduced predatory detection, mobility, and access to food resources (Kaminski and Prince 1981, 1984; Anderson and Smith 1999; Gray et al. 1999; Havens et al. 2009; Stafford et al. 2010; Hagy and Kaminski 2012b). In my study area, ricelands with tall dense vegetation generally were either SI or SR fields.

Duck abundance was greatest within deeply flooded ricelands when vegetation height was short or absent. Although SR fields typically used for crayfish production contained unharvested rice crops, once flooded, above water vegetation height typically ranged from 0–15 cm. In southwest Louisiana and parts of southeastern Texas, crayfish production is an important commercial enterprise (McClain and Romaire 2004). Flooding for crayfish production, associated aquaculture practices, and straw decomposition reduced above-water height of vegetation. Absent or short vegetation above the water surface may facilitate greater use by ducks in fields where crawfish are being harvested, especially during times of minimal disturbance when harvesting machinery is not in use. Additionally, dense vegetation persisting below the surface of the water is critical for the production of crayfish and aquatic invertebrates which are important food resources for waterfowl.

Flooding rice stubble establishes the detritus-based food web for crayfish and other aquatic invertebrates (McClain and Romaire 2004, Alford 2014). Aquatic invertebrates provide essential nutrients, such as proteins and their constituent amino acids that are important to pre-breeding waterfowl, especially female ducks in pre-basic molt during winter and early spring (Heitmeyer 1988, Richardson and Kaminski 1992, Barras et al. 2001). Foley (2015) reported that rice fields flooded for crayfish production in the LCP and TCP supported diverse aquatic invertebrate assemblages and contained 40 invertebrates/m2 in canal irrigated rice fields and 63 invertebrates/m2 in well irrigated rice fields. Albeit lower than estimates in the Central Valley of California, where Loughman and Batzer (1992) reported chironomid larvae densities of 50–>400 invertebrates/m2, waterfowl likely used deeply flooded rice fields in part to forage on aquatic invertebrates.

The probability of measuring false zeros versus true counts and zeros was greatest (84-97%) in saturated ricelands (i.e., water depths <1 cm) regardless of vegetation height. However, saturated soils were not frequently used by ducks in GCP ricelands; thus, I cannot infer why probabilities were so large. Furthermore, probabilities of observing a false negative were also large for shallowly (57%), intermediately (68%), and deeply (66%) flooded ricelands with tall vegetation. These results may indicate that waterfowl may have been present, but went undetected because of visual obstruction from tall vegetation. To reduce the probability of observing a false negative, an observer could walk or ride an all-terrain vehicle though fields to flush birds. Alternatively, ducks actually were not present because the majority of them foraged in ricelands nocturnally (Miller 1987, McNeil and Rodriguez 1996, Cox and Afton 1997).

Waterbird Abundance

Variation in waterbird abundance in GCP ricelands was best explained by vegetation density, water depth, and wetland size. Abundance of waterbirds was generally lowest in fields with dense vegetation regardless of water depth. Fields containing dense vegetation likely precluded use by avifauna who typically utilize flocking strategies to increase predator avoidance and foraging efficiency (Powell 1974, Morse 1977, Cresswell 1994). Moreover, greatest waterbird abundances generally occurred in fields with sparse vegetation regardless of water depth. Crayfish fields, and fields flooded for recreational purposes typically contained sparse above-water vegetation density. Sparse above-water vegetation density likely increased predator detection. Although above-water vegetation density may be sparse, below-water density is often dense and promotes the production of crayfish and other aquatic invertebrates (McClain and Romaire 2004).

Wetland birds use a diversity of available foods in production and idled rice fields including aquatic invertebrates, fish, and amphibians for essential nutrients during the non-breeding period (Krapu and Reinecke 1992, Gonzalez-Solis et al. 1996, Richardson 2001, Kosteke et al. 2005, Baldassarre and Bolen 2006, Ma et al. 2009). Wading and shorebirds vary greatly in body size and partition their foraging patches across water depths in wetlands and agricultural fields; these strategies theoretically may reduce intra- and interspecific competition for food (Nudds and Kaminski 1984, Davis and Smith 2001). Gawlik (2002) suggested that wading bird feeding constraints can be viewed as a continuum with searchers (e.g., white ibis, wood storks [*Mycteria americana*], snowy egrets [*Egretta thula*]) and exploiters (e.g., great blue heron [*Ardea herodias*], great egret [*Ardea alba*]) occupying opposite ends of behavioral foraging regimes. Searchers forage primarily in shallow and intermediate water depths and abandon foraging plots quickly when prey density begins to decrease, whereas exploiters persist in wetlands and forage in all water depths because of adaptations that mitigate the effects of decreasing prey density (e.g., morphology, behavioral plasticity; Maurer 1996, Gawlik 2002). During surveys, I observed "searchers," such as white ibis and snowy egrets, exploiting newly flooded ricelands, possibly exploiting emerging foods including crayfish from their boroughs. Furthermore, I witnessed exploiters such as great blue herons and great egrets using freshly flooded fields; however, they continued to use fields over successive surveys.

Although I did not directly investigate water depth gradients used by individual species of waterbirds, I observed birds with shorter legs (i.e., shorebirds, rails, ibis, snowy egrets, little blue heron, etc.) generally occupying shallower depths (1–15 cm), while birds with longer legs, such as great egrets and great blue herons, foraged in shallow and deep water (1–>30 cm). Furthermore, the probability that waterbirds used ricelands was greatest for shallow (≤15 cm) and deep water depths (≥30 cm), regardless of field classification. Wading bird foraging depth is primarily partitioned by body morphology such as bill and leg length, and ranges from adjacent dry uplands to water depths ~40 cm (Kushlan 1986, Bancroft et al. 2002, Gawlik 2002). Elphick and Oring (1998) reported that median water depths used by wading birds ranged from 9–20 cm. Longer leg lengths provide opportunities to forage amid deeper water depths, whereas those with shorter legs (e.g., sandpipers) are more restricted in foraging opportunities. Bill morphology is also related to birds' diet and prey foraging success (Kushlan 1978, Gawlik 2002). Smith (1977) reported that little blue herons and great egrets, which have thicker bills than snowy egrets, switched prey types as hydrological conditions changed in foraging areas, whereas snowy egrets did not switch. Additionally, behavioral plasticity permits birds to exploit a wider range of water depths, such as tricolored herons that forage atop floating vegetation and also in amid deep water (Smith 1995, Gawlik 2002). While conducting surveys, I witnessed white ibis, white-faced ibis, snowy egret, and little blue heron perching on crayfish traps, perhaps using these structures as an extension "ladder" to access prey near traps that otherwise would not be inaccessible due to water depths. Similar to wading birds, shorebird foraging depth generally ranges from moist adjacent uplands to water depths of 15 cm, and is primarily constrained by culmen and tarsus lengths (Baker 1979, Elner and Seaman 2003, Colwell 2010). Elphick and Oring (1998) found that median water depths used by shorebirds in California rice fields ranged from 3–13 cm. The probability that waterbirds used ricelands was generally greatest in shallowly flooded fields regardless of vegetation density. Elphick and Oring (2003) reported that shorebirds avoided fields with tall or dense vegetation, and used fields which had been disked in greater numbers. During spring migration in Louisiana, Rettig (1994) observed 70% of shorebirds in rice fields with <50% vegetation, although only 19% of fields contained <50% vegetative cover.

Duck Abundance in Relation to Rice Seed Variety

Anecdotal observations have led to speculation among local waterfowl hunters that ducks may be avoiding rice fields planted with Clearfield® rice varieties because of forage limitations. I detected a statistically greater waste-rice and natural seed biomass in fields planted with conventional rice varieties (Chapter I). For both Clearfield® and conventional varieties, waste-rice biomass remained greater than the giving-up density of 50 kg/ha (Greer et al. 2009). For conventional varieties, November, 2010–2013 natural seed biomass was greater than the forage availability threshold of 170 kg/ha (Hagy and

Kaminski 2015). However, November, 2010–2013 natural seed biomass in fields planted with Clearfield® rice was below the forage availability threshold of 170 kg/ha. Nonetheless, a zero-inflated negative binomial regression model indicated that duck abundance did not differ between Clearfield® and conventional rice seed varieties. Although there was no statistical difference in duck abundance between rice seed varieties, results indicated slightly fewer ducks in fields planted with Clearfield® rice than conventional rice, consistent with less seed biomass in the former than the latter. I also investigated the possibility that measuring a false negative existed, but did not find any differences between rice varieties. The probability of measuring a false negative was high (~50%) regardless of seed variety indicating that habitat was suitable and birds were not present, birds were present and I failed to detect them, or birds potentially foraged in fields nocturnally. Thus, I conclude that although a difference in waste-rice and natural seed biomass may exist between seed varieties, my surveys did not reveal any significant differences in duck use between rice varieties.

Importance of MBHI for Wetland Birds in the Gulf Coast Prairies

In 2010 and 2011, in response to the Deep Water Horizon Oil Spill and subsequently the beginning of one of the largest droughts in GCP history, the MBHI provided incentives for landowners to flood production and idled rice fields and other wetland habitats during autumn and winter (Davis et al. 2014, Kaminski and Davis 2014). Financial incentives from MBHI enabled farmers to pump and flood ricelands in Louisiana and Texas; the NRCS signed contracts and obligated approximately 93,388 ha of land in this effort (U.S. Department of Agriculture 2010a,b). Conservation programs such as MBHI provided critical wetland habitat for millions of wetland birds across southern Gulf of Mexico states (Borrow et al. 2013, Davis et al. 2014, Kaminski and Davis 2014). During my research, I observed 53 wetland bird species using fields enrolled in MBHI and those with similar management practices promoted by MBHI. The MBHI flooding regimes provided habitats attractive to diverse wetland bird guilds which migrate through and winter in the GCP regions during the nonbreeding period. I rarely observed wetland birds using dry rice fields, but, when observed, birds used dry fields adjacent to flooded fields. Similarly, Elphick and Oring (2003) found that wetland bird richness and density were greater in flooded than unflooded rice field in California. The most common species observed in MBHI fields were lesser snow geese, greater white-fronted geese (*Anser albifrons*), blue-winged teal (*Anas discors*), American green-winged teal, northern shoveler (*A. clypeata*), white-faced ibis (*Plegadis chihi*), dowitchers, and sandpipers. Additionally, I observed one whooping crane (*Grus americana*) in a LCP production rice field, and bald eagles (*Haliaeetus leucocephalus*) in TMC production rice fields, the latter of which prey on waterfowl on migration and wintering grounds (McWilliams et al. 1994).

Management Implications

Models explaining variation in wetland bird use varied among guilds. Duck use of ricelands was best predicted by vegetation height, water depth, and wetland size, while abundance of other waterbirds was best predicted by field classification, water depth, and wetland size. Water depth and wetland size influenced habitat use for all wetland bird guilds. In my study, wetland birds required variable water depths within ricelands ranging from saturated to >30 cm of water. Habitat complexes containing wetlands and agricultural resources are attractive and promote diverse guilds of wetland birds (Elphick and Oring 2003, Hagy and Kaminski 2012b, Pearse et al. 2012). I suggest conservation planners and policy makers create conservation programs that encourage landowners, rice producers, and complex managers to flood both production and idled rice fields during autumn–early spring for migrating, wintering, and locally breeding wetland birds. Management practices within programs should emphasize closing water control structures to capture rainfall following the first and ratoon harvests in production rice fields and in idled rice fields (Manley et al. 2004, Eadie et al. 2008). In addition to creating valuable shallow water and mudflat habitats, captured rainfall might save producers money through reduced pumping or canal water costs, as well as benefitting aquifer rejuvenation. I recommend conservation programs accommodate a suite of flooding regimes to promote habitat complexes with variable water depths to meet the needs of multiple wetland bird guilds, including 1–15 cm for shorebirds, 9–20 cm for herons and ibis, 14–22 cm for dabbling ducks, 18–26 cm for geese, and 24–34 cm for diving waterfowl species (Elphick and Oring 1998, 2003; Hagy and Kaminski 2012b). Furthermore, height and density of vegetation in production and idled rice fields will subsequently be reduced through the use of program flooding. Flooding fields eventually creates natural openings through decomposition, and immigrating wetland birds will further accelerate vegetation toppling (Anders et al. 2008). The hemi-marsh concept is a classic wetland paradigm, originally conceived in northern prairie wetlands (Kaminski and Prince 1981, Murkin et al. 1982), but can be extended to non-breeding habitats to benefit wetland birds (Smith et al. 2004, Havens et al. 2010, Hagy and Kaminski 2012b).

In addition to providing valuable wetland habitat, winter flooding of ricelands can save producers significant amounts of money annually. The costs of post-harvest field manipulations can range from \$6.65/ha for burning to \$197/ha for chopping, rolling, tilling, or disking (Brouder and Hill 1995, Horwath and van Kessel 2001). Compounding these costs across whole farms could be significant for some producers. Flooding rice fields to attract foraging waterfowl can reduce red rice and other weeds. Red rice, also *Oryza sativa*, is similar to commercial rice, but reduces yield and quality of commercial crops in the southern United States (Khodayari et al. 1987). Previous research has estimated that winter water management reduced red rice by as much as 97% and potentially saved the rice industry more than \$290 million in 1997 (Smith et al. 1977, Smith and Sullivan 1980, Hobaugh et al. 1989). Furthermore, retaining some straw and flooding fields during winter can improve nitrogen uptake in subsequent crops, reduce water volume runoff, and reduce suspended and dissolved solids (Anders et al. 2008, Manley et al. 2009).

Market prices for rice and other alternative crops, such as soybean, are a significant determinant for the extent of rice planted each year in the GCP. With potential for rising input costs associated with seed, fuel, fertilizers, and herbicides, producers may elect to grow alternative crops or stop farming altogether. If many producers stop farming or find it more profitable to grow alternative crops, abundance of wetland bird habitat in the form of ricelands could substantially decrease in the GCP. One of the greatest concerns for rice farmers and conservation organizations has been recent droughts and subsequent water restrictions implemented by the Lower Colorado River Authority (LCRA) in Texas. The LCRA controls the water supply for most of the TMC and supplies about 60% of total irrigation demands for agriculture (LCRA 2010, 2013). From 2011–2015, the LCRA either restricted or eliminated irrigation water for rice producers in the region, seemingly hindering the TMC rice producing industry. Recent rainfall has since replenished LCRA reservoirs and the supply of irrigation water resumed in 2016. For future considerations, conservation programs such as MBHI may be necessary to promote flooding of agricultural lands, especially if restrictions on irrigation water resume or rice productions costs continue to rise.

Loss of species and changes in community structure can sometimes be attributed to fragmentation and habitat loss (Diamond 1976). As fragmentation occurs, habitats become smaller and increasingly isolated (Farina 1998, Wiens 1995). Research supports that this process selects species better adapted to small, isolated wetlands, and affects the movement of individuals through a landscape, reducing alpha-diversity (i.e., local diversity; Brown and Dinsmore 1986, Fahrig and Merriam 1994, Farbairn and Dinsmore 2001, Whited et al. 2000). As a result, beta-diversity, or the difference in species diversity between habitats, is expected to increase in fragmented landscapes because of isolation effects (Harrison 1997, Kneitel and Chase 2004). Gamma diversity, or regional diversity, is then determined by the alpha and beta components affected by habitat loss and fragmentation (Cody 1993). The understanding of species composition and abundance patterns among sites is a central question in community ecology, but is poorly documented for wetland birds in fragmented wetlands (Cox et al. 2000, Gaudagnin et al. 2005). More than 99% of the prairie ecosystem in the Gulf Coast has been lost to urbanization, agriculture and range improvement, and the remaining 1% persists in highly fragmented patches (USGS 2000). I recommend future research investigating fragmentation and consequential avian community structure at regional and landscape scales in GCP agricultural and coastal marsh habitats. I hypothesize that as habitats become increasingly fragmented by an urban landscape, alpha avian diversity will decrease, beta diversity will increase, and gamma diversity will remain similar.

One of the greatest knowledge gaps in the GCP pertains to nocturnal wetland bird use of ricelands. Wetland bird species such as northern pintail, plovers (Pulvialis, Charadrius), sandpipers, stilts (Himantopus), and most other Scolopacidae regularly forage diurnally and nocturnally (Miller 1987, McNeil and Rodriguez 1996, Cox and Afton 1997). I recommend future wetland bird research that investigates nocturnal use of ricelands, and monitoring bird movements within and between ricelands and coastal marshes. Recent advancements in unmanned aerial drones, night vision, radar, and thermal imaging techniques could enhance our ability to quantify diurnal and nocturnal use of wetland birds in the GCP. Recent research has used drones and thermal imaging to locate nesting ducks in the Prairie Pothole Region of the United States (Delta Waterfowl 2016). Additionally, research investigating effects of disturbance should be high priority for conservation planners in the GCP. For example, crayfish is harvested daily from many of the flooded production and idled rice fields, especially in the LCP. Furthermore, many flooded fields are subjected to frequent waterfowl hunting activities from September–January. Quantifying effects of disturbance and ensuring that wetland birds have access to undisturbed habitats may be important to future conservation planning, especially if flooded areas are reduced, or habitat fragmentation occurs, all of which may cause greater densities of wetland birds on fewer habitats in this regional landscape. Although researchers have identified complexes of wetlands that attract greatest abundances of dabbling ducks in winter (e.g., Pearse et al. 2012), wetland complexes, including sanctuary components, have not been identified for waterfowl and other waterbirds to my knowledge. Moreover, complexes attractive to greatest abundances of wetland birds, although important for conservation of habitat landscapes, may invoke density dependent effects on individuals. Thus, studies that relate habitat use to demographic metrics (e.g., daily survival; Lancaster 2013) are needed to identify most suitable habitat complexes and incorporate this knowledge into local-landscape and reserve designs (sensu Fretwell 1972, Van Horne 1991).

Table 3.1 Wetland bird species encountered during surveys of Gulf Coast Prairies ricelands, August–March, 2010–2013.

Common name		Scientific name	*n*
Waterfowl			
	Snow goose	*Chen caerulescens*	65,546
	White-fronted goose	*Anser albifrons*	35,147
	Canada goose	*Branta canadensis*	1,256
	American green-winged teal	*A. crecca carolensis*	73,251
	Blue-winged teal	*Anas discors*	42,910
	Northern shoveler	*A. clypeata*	28,172
	Northern pintail	*A. acuta*	21,050
	Gadwall	*A. strepera*	7,842
	Mallard	*A. platyrhynchos*	2,182
	Mottled duck	*A. fulvigula*	2,082
	American wigeon	*A. americana*	371
	Wood duck	*Aix sponsa*	436
	Lesser scaup	*Aythya affinis*	283
	Redhead	*A. americana*	118
	Ring-necked duck	*A. collaris*	43
	Ruddy duck	*Oxyura jamaicensis*	133
	Black-bellied whistling duck	*Dendrocygna autumnalis*	183
	Fulvous whistling duck	*D. bicolor*	51
	Bufflehead	*Bucephala albeola*	10
	Hooded merganser	*Lophodytes cucullatus*	4
Waders			
	White-faced ibis	*Plegadis chihi*	47,431
	White ibis	*Eudocimus albus*	15,904
	Cattle egret	*Bubulcus ibis*	6,890
	Great egret	*Ardea alba*	3,921
	Great blue heron	*A. herodias*	544
	Little blue heron	*Egretta caerulea*	3,508
	Snowy egret	*E. thula*	597
	Tricolored heron	*E. tricolor*	122
	Green heron	*Butorides virescens*	97
	Roseate spoonbill	*Platalea ajaja*	86

Table 3.1 Continued

American bittern	*Botaurus lentiginosus*	28
Yellow-crowned night-heron	*Nyctanassa violacea*	23
Black-crowned night-heron	*Nycticorax nycticorax*	8
Wood stork	*Mycteria americana*	7
Shorebirds		
Dowitchers	*Limnodromus* spp.	31,928
Sandpipers	*Calidris* spp.	23,133
Killdeer	*Charadrius vociferus*	9,691
Yellowlegs	*Tringa* spp.	5,554
Black-necked stilt	*Himantopus mexicanus*	5,403
Wilson's snipe	*Gallinago delicata*	813
Long-billed curlew	*Numenius americanus*	207
Black-bellied plover	*Pluvialis squatarola*	206
American avocet	*Recurvirostra americana*	69
Rails		
American coot	*Fulica americana*	12,459
Purple gallinule	*Porphyrio martinicus*	18
Other rails	*Coturnicops* spp.; *Porzana* spp.; *Rallus* spp.	14
Others		
Sandhill crane	*Grus canadensis*	3,531
Whooping crane	*G. americana*	1
Gulls	*Larus* spp.	2,931
Pied-billed grebe	*Podilymbus podiceps*	185
Double-crested cormorant	*Phalacrocorax auritus*	172
Gull-billed tern	*Gelochelidon nilotica*	10
Belted kingfisher	*Megaceryle alcyon*	4

Common and scientific names and total detections (*n*) of wetland birds encountered during surveys of production and idled rice fields in the Gulf Coastal Prairies of Louisiana and Texas, August–March, 2010–2013.

Table 3.2 Results of linear mixed models explaining variation in wetland bird species richness in Gulf Coast Prairie ricelands, August–March, 2010–2013.

Models	AICc	ΔAICc	w_i	K	*LL*	R^2
Vegetation density + Water depth + Wetland size	5067.9	0.0	0.99	10	-2523.3	0.10
Vegetation height + Water depth + Wetland size	5077.3	9.4	0.01	10	-2528.0	0.10
Vegetation density + Water depth	5111.0	43.1	0	9	-2546.0	0.08
Water depth + Wetland size	5111.2	43.3	0	7	-2548.3	0.08
Vegetation height + Water depth	5121.4	53.5	0	9	-2551.2	0.08
Month + Water depth	5126.8	58.9	0	13	-2549.3	0.08
Water depth	5151.5	83.6	0	6	-2569.5	0.06
Vegetation density + Wetland size	6251.4	1183.6	0	7	-3118.4	0.04
Vegetation height + Wetland size	6283.5	1215.6	0	7	-3134.4	0.05
Wetland size	6314.3	1246.4	0	4	-3153.0	0.02
Vegetation density	6315.8	1247.9	0	6	-3151.7	0.02
Year + Month	6344.3	1276.4	0	13	-3158.1	0.02
Month	6345.5	1277.6	0	10	-3162.1	0.02
Vegetation height	6353.4	1285.5	0	6	-3170.5	0.01
Year	6377.7	1309.8	0	6	-3182.6	0.01
Null	6380.9	1313.0	0	3	-3187.4	0.00

Results of linear mixed models explaining variation wetland bird[a] species richness in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013. Models were ranked by Akaike's Information Criterion corrected for small sample size (AICc) and includes ΔAICc, model weight (w_i), number of estimable parameters (K), deviance (*LL*), and R^2 statistics.

[a] Waterfowl, wading birds, and shorebirds species combined.

Table 3.3 Estimated mean wetland bird species richness in Gulf Coast Prairie ricelands, August–March, 2010–2013.

			95% Confidence interval	
Water depth[a,b]	Vegetation density[c]	$\bar{x}$	LCL	UCL
Saturated	No vegetation	2.15	2.0	2.3
	Sparse	2.26	2.1	2.5
	Intermediate	1.97	1.8	2.2
	Dense	1.77	1.6	1.9
Shallow	No vegetation	3.28	3.0	3.6
	Sparse	3.46	3.1	3.8
	Intermediate	3.02	2.8	3.3
	Dense	2.70	2.5	2.9
Intermediate	No vegetation	2.69	2.4	3.0
	Sparse	2.83	2.5	3.2
	Intermediate	2.47	2.2	2.7
	Dense	2.21	2.0	2.4
Deep	No vegetation	2.43	2.2	2.7
	Sparse	2.56	2.3	2.9
	Intermediate	2.24	2.0	2.5
	Dense	2.00	1.8	2.2

Back-transformed estimated mean species richness (wetland bird species/survey) for wetland birds[d] and 95% confidence intervals (LCL, UCL) from linear mixed models by water depth and vertical vegetation density for the average wetland size of 17.9 ha, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

[a] Blanks denote same water depth.

[b] Saturated (<1 cm); Shallow (1–15 cm); Intermediate (16–30 cm); Deep (>30 cm).

[c] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[d] Waterfowl, wading birds, and shorebird species combined.

Table 3.4 Results of zero-inflated regression models explaining variation in duck abundance in Gulf Coast Prairie ricelands, August–March, 2010–2013.

Model[a]	AICc	ΔAICc	w_i	K	*LL*
Vegetation height + Water depth + Wetland size	8505.2	0.0	0.75	17	-4233.8
Vegetation height + Water depth	8509.6	4.3	0.09	15	-4238.4
Vegetation density + Water depth	8533.2	27.97	0	15	-4250.2
Vegetation density + Water depth + Wetland size	8533.3	28.1	0	17	-4247.8
Month + Water depth	8548.6	43.4	0	23	-4247.9
Water depth + Wetland size	8594.3	89.07	0	11	-4285.4
Water depth	8595.0	89.8	0	9	-4288.0
Vegetation density + Wetland size	10672.3	2167.04	0	11	-5324.4
Vegetation height + Wetland size	10709.9	2204.7	0	11	-5343.2
Wetland size	10757.8	2252.6	0	5	-5373.7
Vegetation density	10686.1	2180.8	0	9	-5333.5
Vegetation height	10746.2	2241.0	0	9	-5363.6
Year + Month	10701.3	2196.1	0	23	-5324.2
Month	10754.9	2249.7	0	17	-5358.6
Year	10743.8	2238.5	0	9	-5362.4
Null	10792.0	2286.8	0	3	-5392.9

Results of zero-inflated negative binomial regression models explaining variation in mean duck abundance in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013. Models were ranked by Akaike's Information Criterion corrected for sample size (AICc), including and model weight (w_i), number of estimable parameters (K), and deviance (*LL*). Parameterizations were identical in count and zero components of zero-inflation models.

Table 3.5 Estimated mean duck abundance in Gulf Coast Prairie ricelands, August–March, 2010–2013.

			95% Confidence interval	
Water depth[ab]	Vegetation height[c]	Count	LCL	UCL
Saturated	None	19.59	9.4	40.7
	Short	50.96	23.0	113.0
	Intermediate	14.56	7.0	30.5
	Tall	19.48	9.5	40.0
Shallow	None	138.38	84.8	225.9
	Short	360.04	216.8	597.9
	Intermediate	102.89	65.4	161.8
	Tall	137.61	92.0	205.9
Intermediate	None	171.90	93.1	317.5
	Short	447.26	264.0	757.7
	Intermediate	127.81	68.9	237.1
	Tall	170.94	99.4	293.9
Deep	None	28.17	15.1	52.6
	Short	73.30	39.4	136.4
	Intermediate	20.95	12.3	35.6
	Tall	28.01	17.1	46.0

Estimated mean duck abundance and 95% confidence intervals (LCL, UCL) from zero-inflated negative binomial regression models by water depth and vegetation height for the average wetland size of 17.9 ha, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

[a] Blanks denote same water depth.

[b] Saturated (<1 cm); Shallow (1–15 cm); Intermediate (16–30 cm); Deep (>30 cm).

[c] None (0 cm); Short (1–15 cm); Intermediate (16-40 cm); Tall (>40 cm).

Table 3.6 Estimated mean probability of measuring a false negative for ducks in Gulf Coast Prairie ricelands, August–March, 2010–2013.

			95% Confidence interval	
Water depth[a,b]	Vegetation height[c]	Probability	LCL	UCL
Saturated	None	0.92	0.87	0.95
	Short	0.84	0.75	0.91
	Intermediate	0.89	0.83	0.94
	Tall	0.97	0.95	0.98
Shallow	No None	0.34	0.21	0.50
	Short	0.20	0.09	0.39
	None	0.28	0.15	0.47
	Tall	0.57	0.45	0.68
Intermediate	None	0.45	0.30	0.61
	Short	0.29	0.14	0.49
	Intermediate	0.39	0.23	0.57
	Tall	0.68	0.58	0.76
Deep	None	0.43	0.27	0.60
	Short	0.27	0.13	0.47
	Intermediate	0.37	0.21	0.56
	Tall	0.66	0.54	0.76

Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals (LCL, UCL) from zero-inflated negative binomial regression models by water depth and vegetation height for the average wetland size of 17.9 ha, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

[a] Blanks denote same water depth.

[b] Saturated (<1 cm); Shallow (1–15 cm); Intermediate (16–30 cm); Deep (>30 cm).

[c] None (0 cm); Short (1–15 cm); Intermediate (16-40 cm); Tall (>40 cm).

Table 3.7 Results of negative binomial Hurdle models explaining variation in waterbird abundance in Gulf Coast Prairie ricelands, August–March, 2010–2013.

Model[a]	AICc	ΔAICc	w_i	K	*LL*
Vegetation density + Water depth + Wetland size	18942.0	0.0	0.88	17	-9453.9
Vegetation height + Water depth + Wetland size	18946.0	3.9	0.12	17	-9455.9
Vegetation density + Water depth	18966.2	24.2	0	15	-9468.0
Vegetation height + Water depth	18974.2	32.2	0	15	-9472.0
Month + Water depth	18977.6	35.5	0	23	-9465.6
Water depth + Wetland size	18982.5	40.5	0	11	-9480.2
Water depth	19008.1	66.1	0	9	-9495.0
Vegetation density + Wetland size	22841.6	3899.5	0	11	-11409.7
Vegetation height + Wetland size	22847.2	3905.2	0	11	-11412.6
Wetland size	22879.1	3937.1	0	5	-11434.6
Vegetation density	22877.8	3935.8	0	9	-11429.9
Vegetation height	22893.8	3951.8	0	9	-11437.9
Year + Month	22850.5	3908.5	0	23	-11402.1
Month	22872.7	3930.6	0	17	-11419.2
Year	22898.2	3956.1	0	9	-11440.1
Null	22917.9	3975.9	0	3	-11455.9

Results of negative binomial Hurdle regression models explaining variation in mean waterbird[a] abundance in the Gulf Coastal Prairies of Louisiana and Texas during August–March, 2010–2013. Models were ranked by Akaike's Information Criterion corrected for sample size (AICc), including and model weight (w_i), number of estimable parameters (K), and deviance (*LL*). Parameterizations were identical in count and zero components of Hurdle models.

[a] Shorebirds and wading birds combined.

Table 3.8 Estimated mean waterbird abundance in Gulf Coast Prairie ricelands, August–March, 2010–2013.

			95% Confidence interval	
Water depth[ab]	Vegetation density[c]	Count	LCL	UCL
Saturated	No vegetation	32.25	23.4	44.4
	Sparse	45.77	30.6	68.4
	Intermediate	30.17	20.3	44.8
	Dense	17.27	12.4	24.0
Shallow	No vegetation	58.66	41.8	82.3
	Sparse	83.25	56.4	122.9
	Intermediate	54.89	36.7	82.0
	Dense	31.42	23.1	42.8
Intermediate	No vegetation	44.93	29.3	68.9
	Sparse	63.77	41.4	98.3
	Intermediate	42.04	27.6	64.0
	Dense	24.06	16.6	34.8
Deep	No vegetation	42.66	27.4	66.3
	Sparse	60.55	37.9	96.7
	Intermediate	39.92	27.4	58.2
	Dense	22.85	15.2	34.2

Estimated mean waterbird[d] abundance and 95% confidence intervals (LCL, UCL) from negative binomial Hurdle regression models by vegetation density and water depth for the average wetland size of 17.9 ha, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

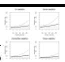
[a] Blanks denote same field classification.

[b] FH-first harvest, NR-no ratoon, HR-harvested ratoon, SR-standing ratoon, SI-standing idle, DI-disked idle.

[c] Saturated (<1 cm); Shallow (1–15 cm); Intermediate (16–30 cm); Deep (>30 cm).

[d] Shorebirds and wading birds combined.

Table 3.9 Estimated mean probability of waterbirds using Gulf Coast Prairie ricelands, August–March, 2010–2013.

			95% Confidence interval	
Water depth[ab]	Vegetation density[c]	Probability	LCL	UCL
Saturated	No vegetation	0.65 	0.61	0.69
	Sparse	0.63	0.57	0.69
	Intermediate	0.57	0.52	0.63
	Dense	0.56	0.52	0.61
Shallow	No vegetation	0.76	0.72	0.80
	Sparse	0.75	0.69	0.79
	Intermediate	0.70	0.65	0.75
	Dense	0.69	0.65	0.72
Intermediate	No vegetation	0.69	0.63	0.75
	Sparse	0.67	0.60	0.74
	Intermediate	0.62	0.56	0.68
	Dense	0.61	0.55	0.66
Deep	No vegetation	0.69	0.62	0.74
	Sparse	0.67	0.60	0.73
	Intermediate	0.61	0.55	0.67
	Dense	0.60	0.55	0.66

Estimated mean probability of waterbirds using Gulf Coast Prairie ricelands and 95% confidence intervals (LCL, UCL) from negative binomial Hurdle regression models by vegetation density and water depth for the average wetland size of 17.9 ha, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

[a] Blanks denote same field classification.

[b] Saturated (<1 cm); Shallow (1–15 cm); Intermediate (16–30 cm); Deep (>30 cm).

[c] Shorebirds and wading birds combined.

Table 3.10 Estimated mean duck abundance in Gulf Coast Prairie ricelands by rice seed variety, August–March, 2010–2013.

		95% Confidence interval	
Seed variety	**Count**	**LCL**	**UCL**
Clearfield®	65.39	42.8	99.8
Conventional	73.14	43.8	122.1

Estimated mean duck abundance and 95% confidence intervals (LCL, UCL) from zero-inflated negative binomial regression models by rice seed variety, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

Table 3.11 Estimated mean probability of measuring a false negative for ducks in Gulf Coast Prairie ricelands by rice seed variety, August–March, 2010–2013.

		95% Confidence interval	
Seed variety	**Probability**	**LCL**	**UCL**
Clearfield®	0.51	0.35	0.68
Conventional	0.49	0.31	0.67

Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals (LCL, UCL) from zero-inflated negative binomial regression models by seed variety, in ricelands in the Gulf Coast Prairies of Louisiana and Texas during August–March, 2010–2013.

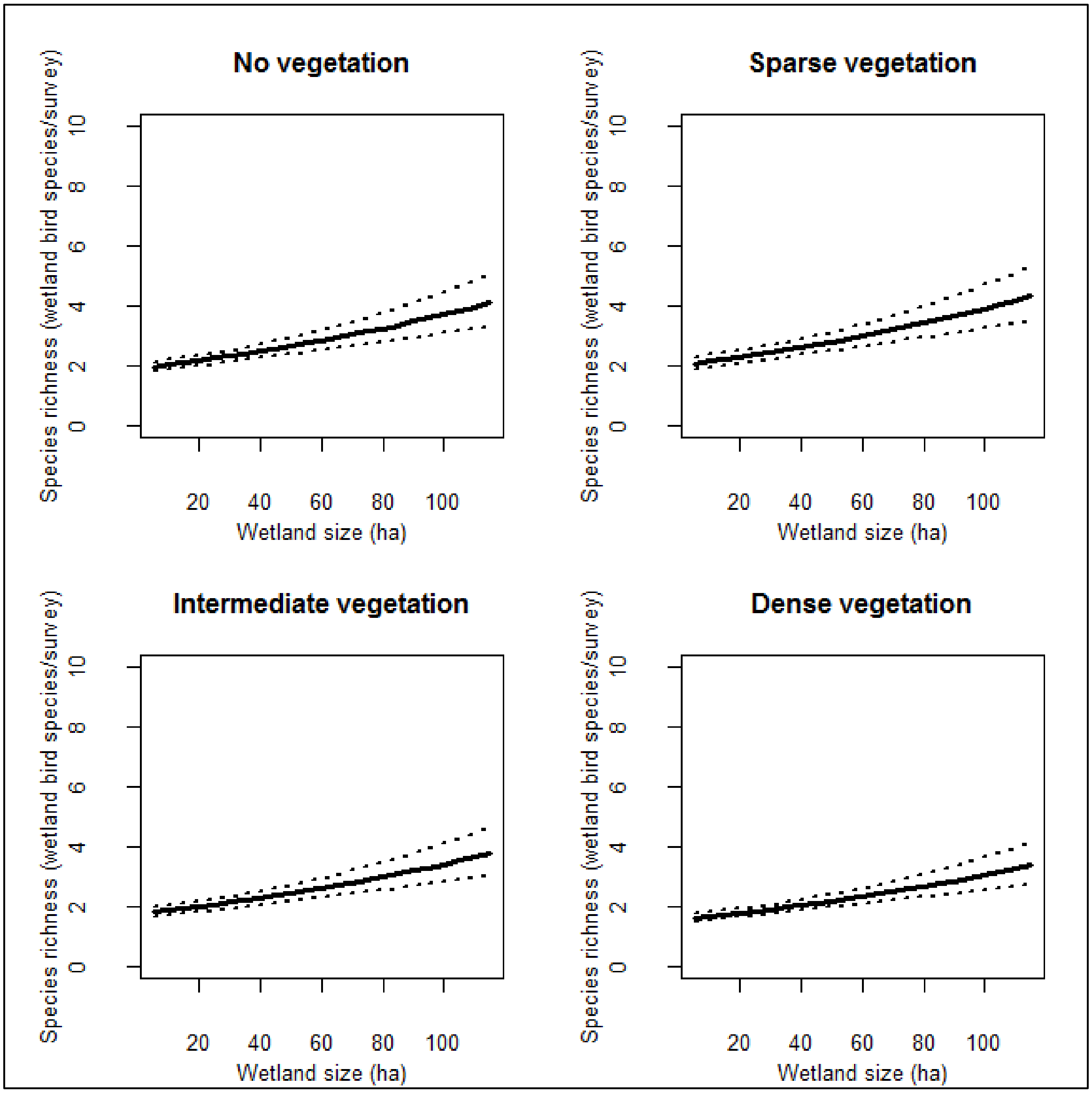


Figure 3.1 Mean wetland bird species richness (wetland bird species/survey) and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with saturated soils[b].

Back-transformed mean wetland bird species richness (wetland bird species/survey; indicated by solid lines) and 95% confidence intervals (dashed lines) from linear mixed models by vegetation density[a] and wetland size for ricelands with saturated soils[b], in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth <1 cm.

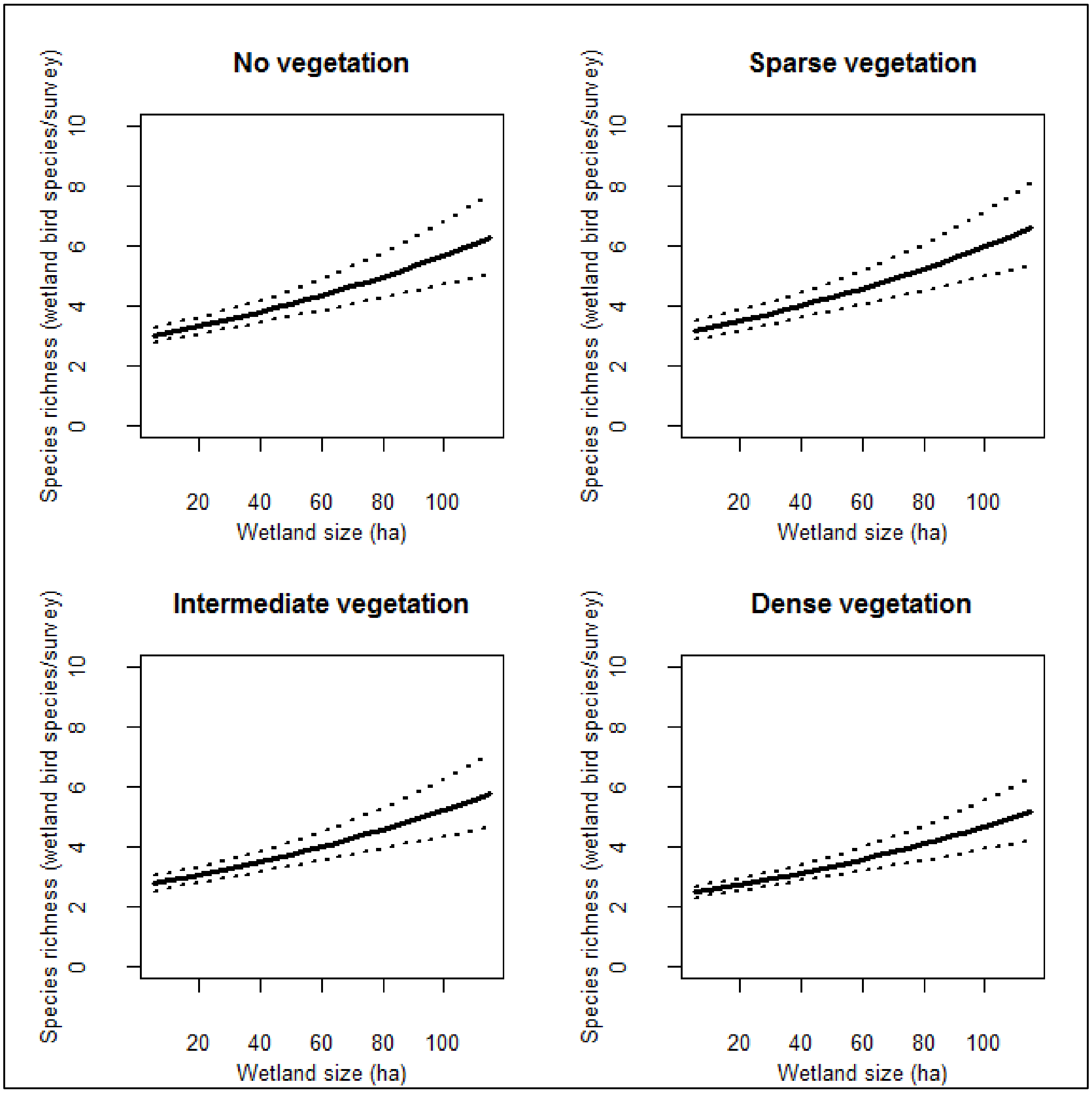


Figure 3.2 Estimated mean wetland bird species richness (wetland bird species/survey) and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands for shallowly[b] flooded ricelands.

Back-transformed estimates of mean wetland bird species richness (wetland bird species/survey; indicated by solid lines) and 95% confidence intervals (dashed lines) from linear mixed models by vegetation density[a] and wetland size for shallowly[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 1–15cm.

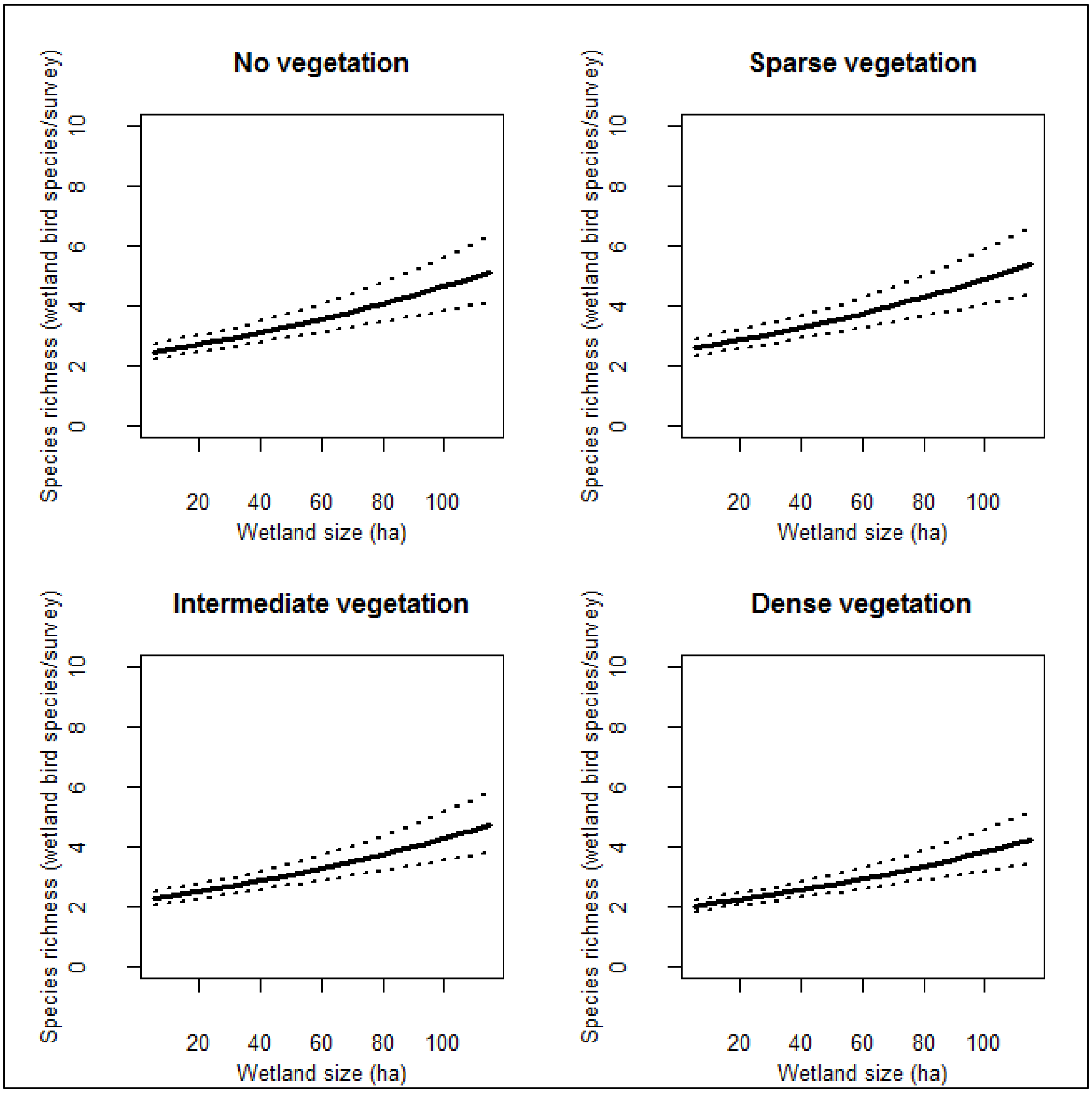


Figure 3.3 Estimated mean wetland bird species richness (wetland bird species/survey) and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands for intermediately[b] flooded ricelands.

Back-transformed estimates of mean wetland bird species richness (wetland bird species/survey; indicated by solid lines) and 95% confidence intervals (dashed lines) from linear mixed models by vegetation density[a] and wetland size for intermediately[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 15–30 cm.

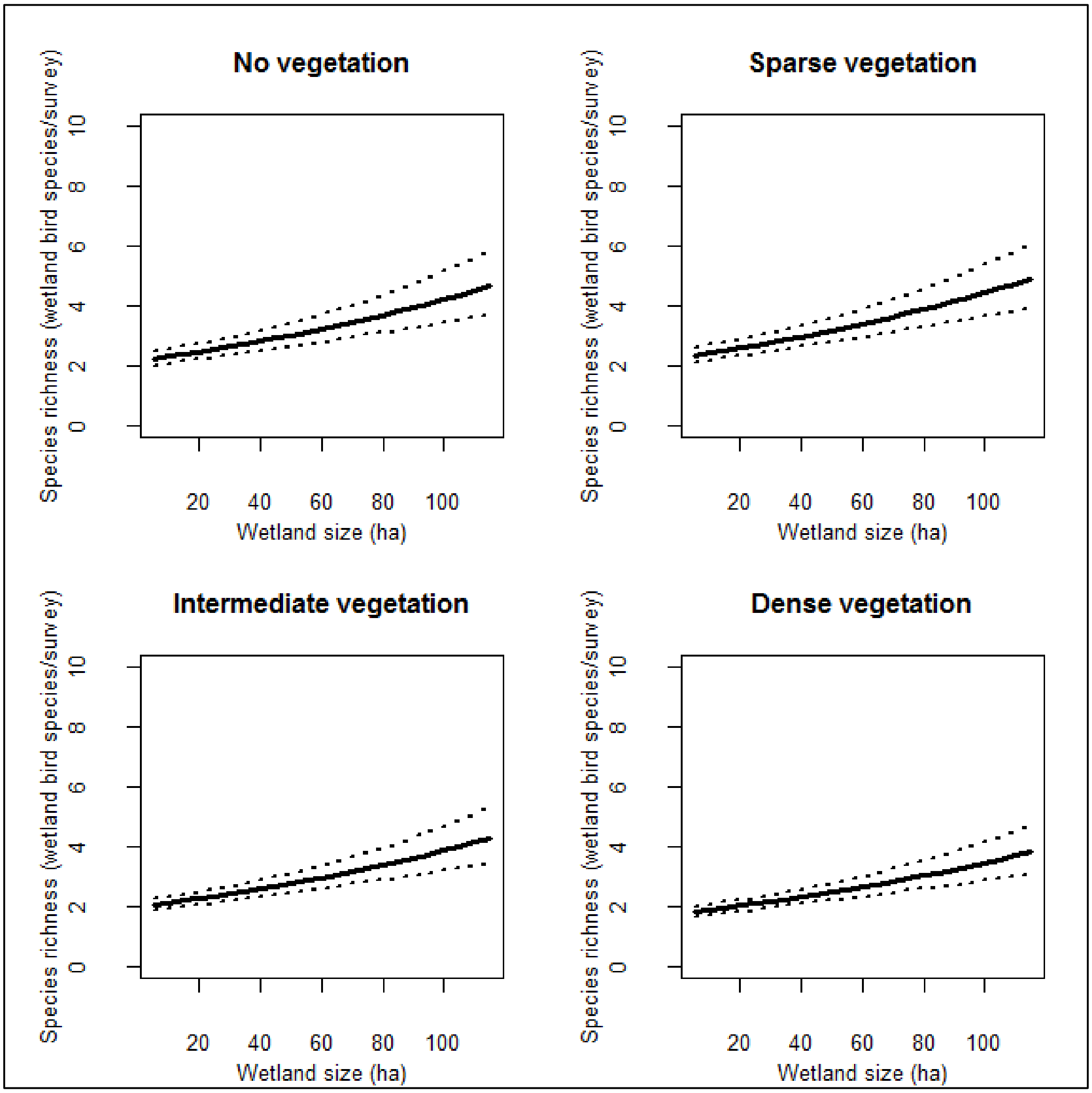


Figure 3.4 Estimated mean wetland bird species richness (wetland bird species/survey) and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands for deeply[b] flooded ricelands.

Back-transformed estimates of mean wetland bird species richness (wetland bird species/survey; indicated by solid lines) and 95% confidence intervals (dashed lines) from linear mixed models by vegetation density[a] and wetland size for deeply[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth >30 cm.

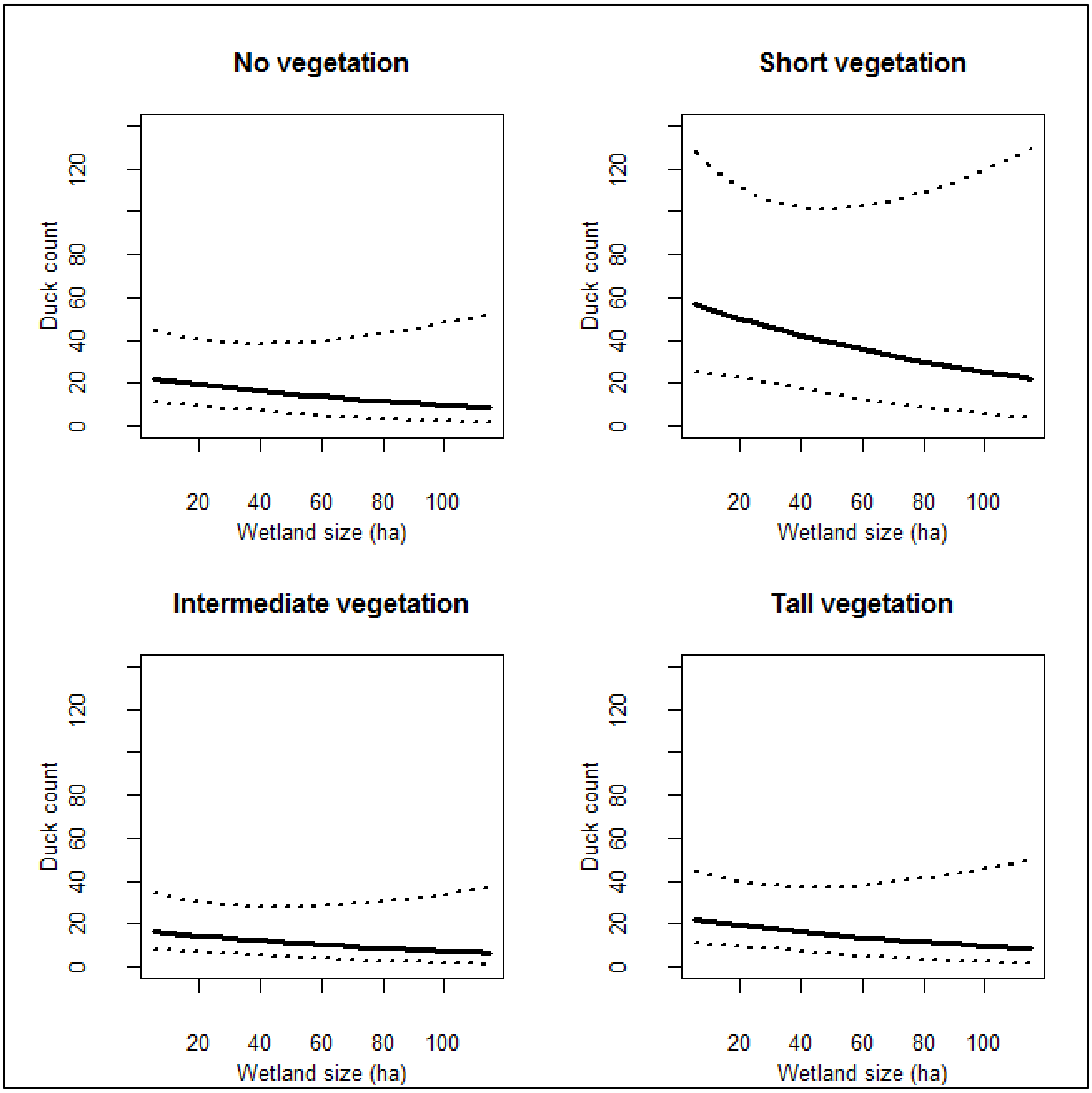


Figure 3.5 Estimated mean duck abundance and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with saturated soils[b].

Back-transformed estimates of mean duck abundance (indicated by solid lines) and 95% confidence (dashed lines) intervals from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for ricelands with saturated soils[b], in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth <1 cm.

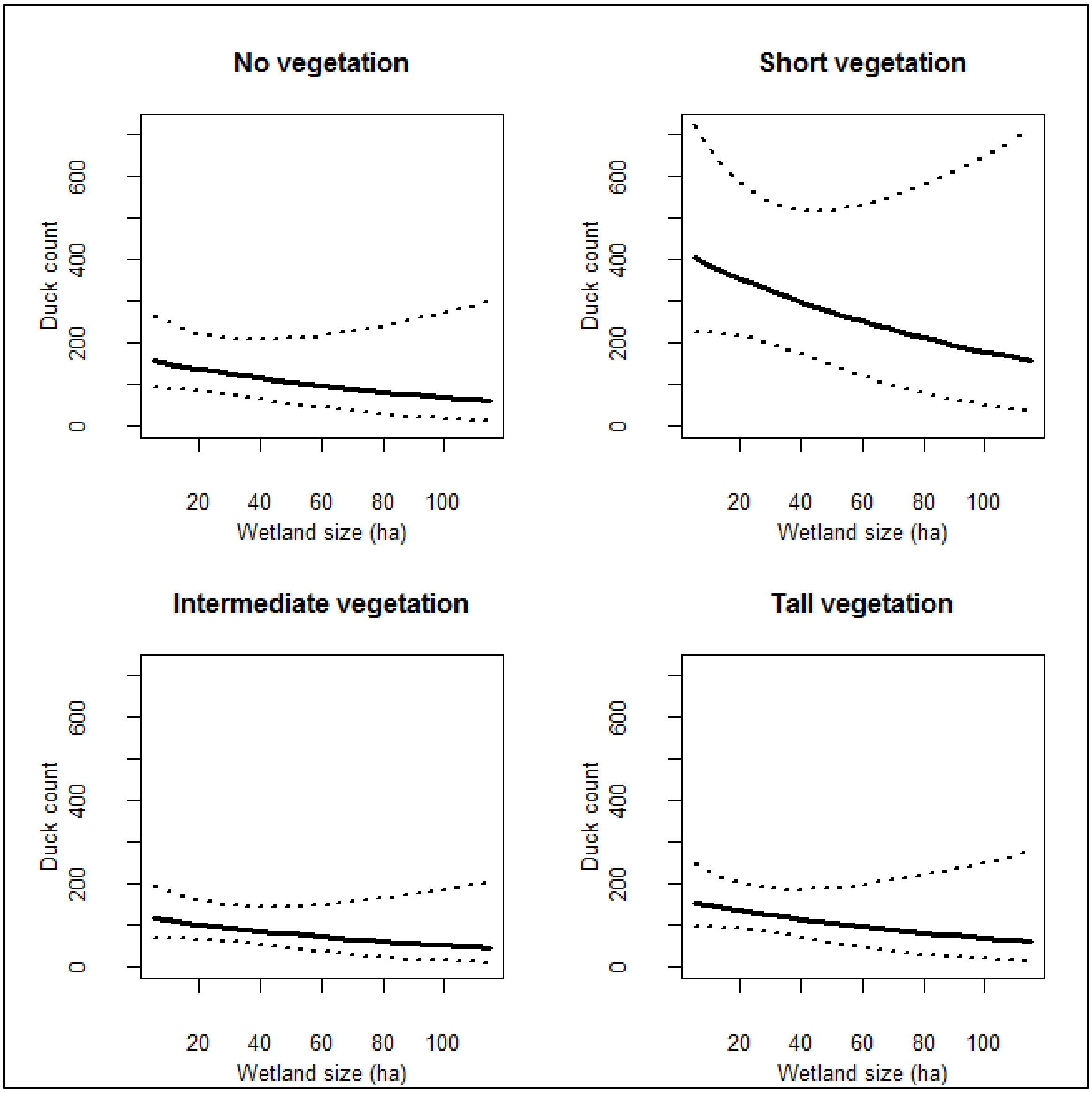


Figure 3.6 Estimated mean duck abundance and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with shallow water depths[b].

Back-transformed estimates of mean duck abundance (indicated by solid lines) and 95% confidence (dashed lines) intervals from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for ricelands for shallowly[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth 1–15 cm.

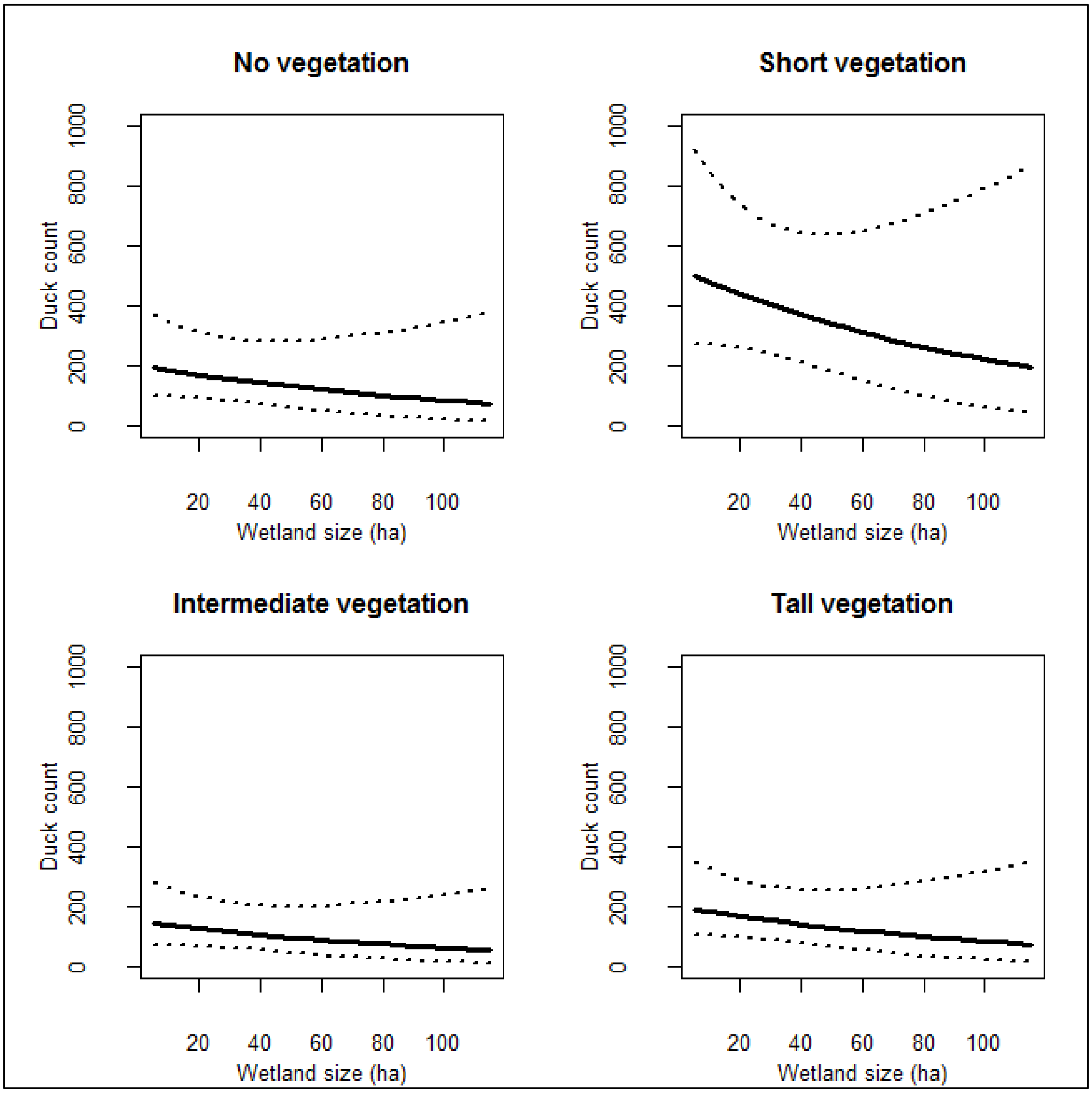


Figure 3.7 Estimated mean duck abundance and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with intermediate water depths[b].

Back-transformed estimates of mean duck abundance (indicated by solid lines) and 95% confidence (dashed lines) intervals from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for ricelands for intermediately[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth 16–30 cm.

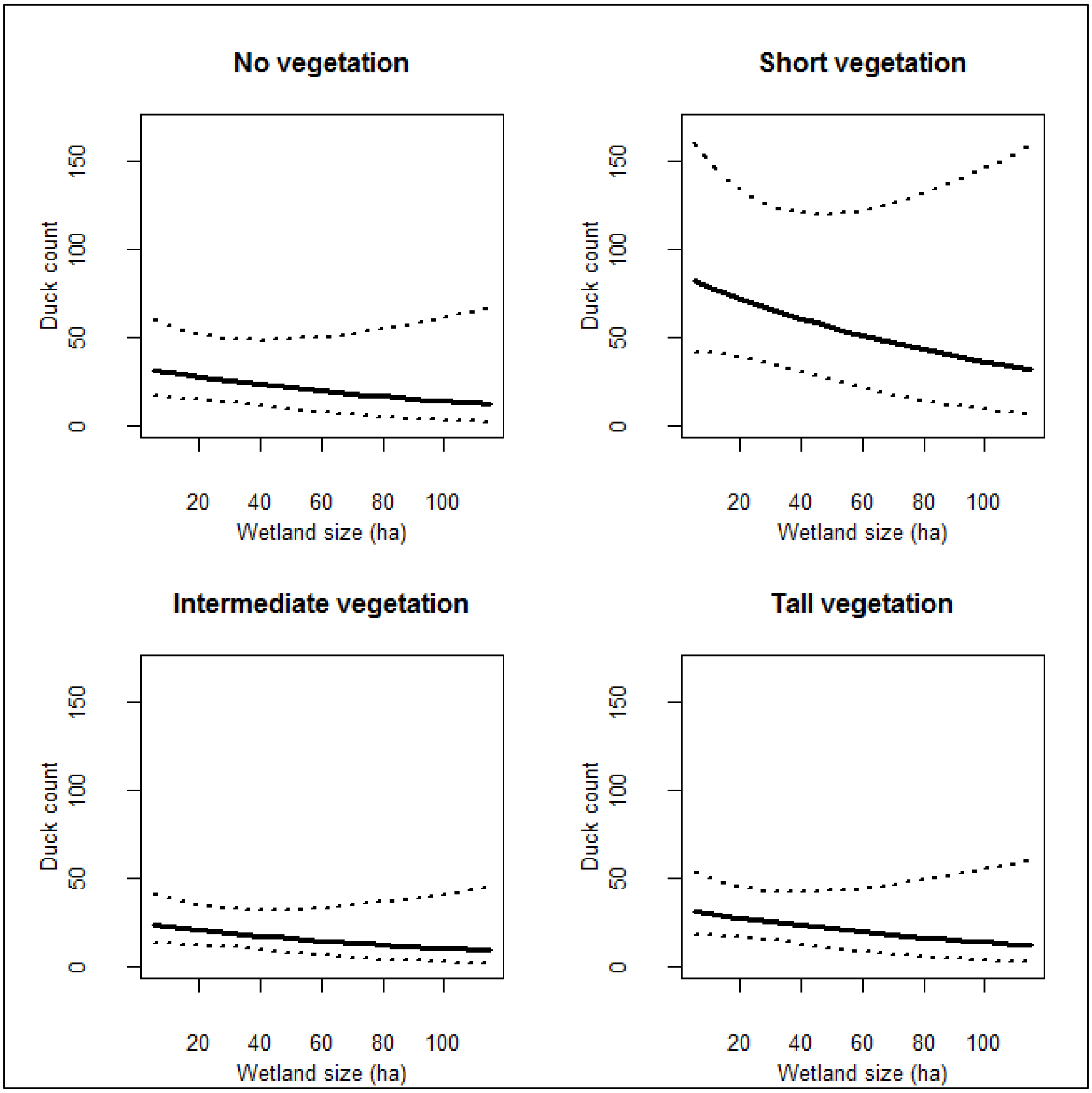


Figure 3.8 Estimated mean duck abundance and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with deep water depths[b].

Back-transformed estimates of mean duck abundance (indicated by solid lines) and 95% confidence (dashed lines) intervals from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for ricelands for deeply[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth >30 cm.

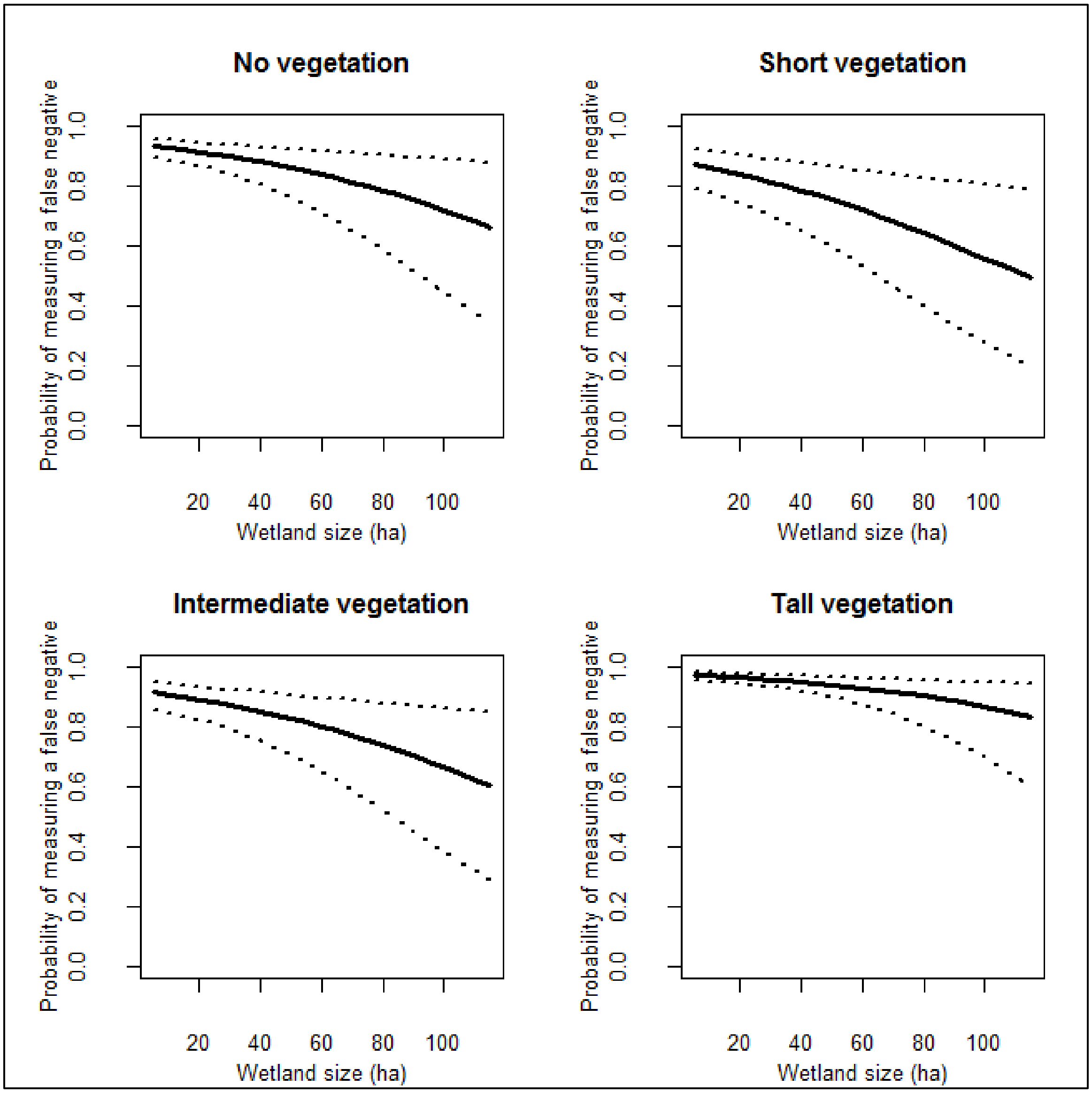


Figure 3.9 Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals by vegetation heighta and wetland size for ricelands with saturated soils[b].

Back-transformed estimated mean probability of measuring a false negative for ducks (indicated by solid lines) and 95% confidence intervals (dashed lines) from zero-inflated negative binomial regression models by vegetation height[a] and wetland size ricelands with saturated soils[b] in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth <1 cm.

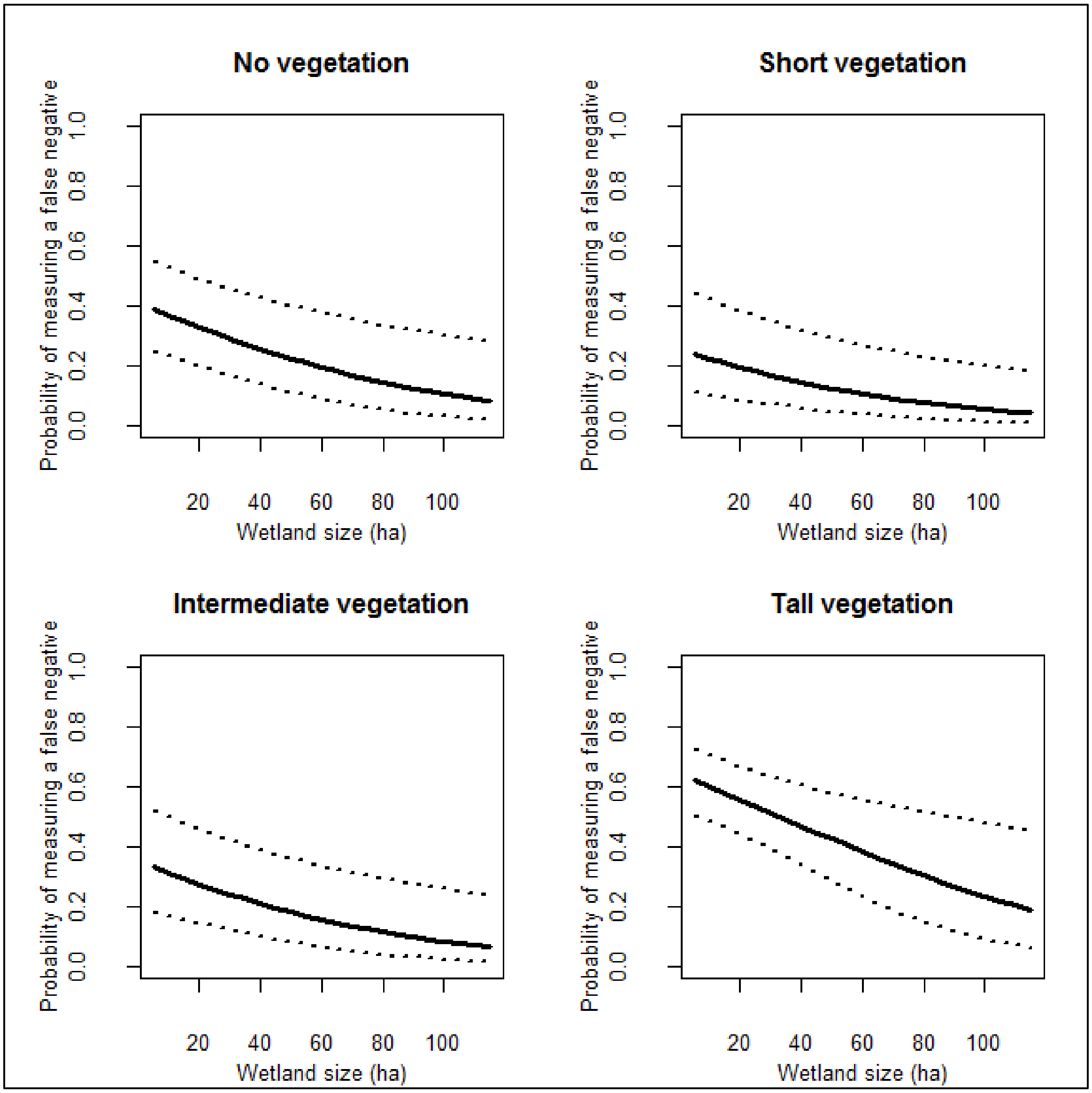


Figure 3.10 Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals by vegetation heighta and wetland size for ricelands with shallow water depthsb.

Back-transformed estimated mean probability of measuring a false negative for ducks (indicated by solid lines) and 95% confidence intervals (dashed lines) from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for shallowly[b] flooded ricelands in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth 1–15 cm.

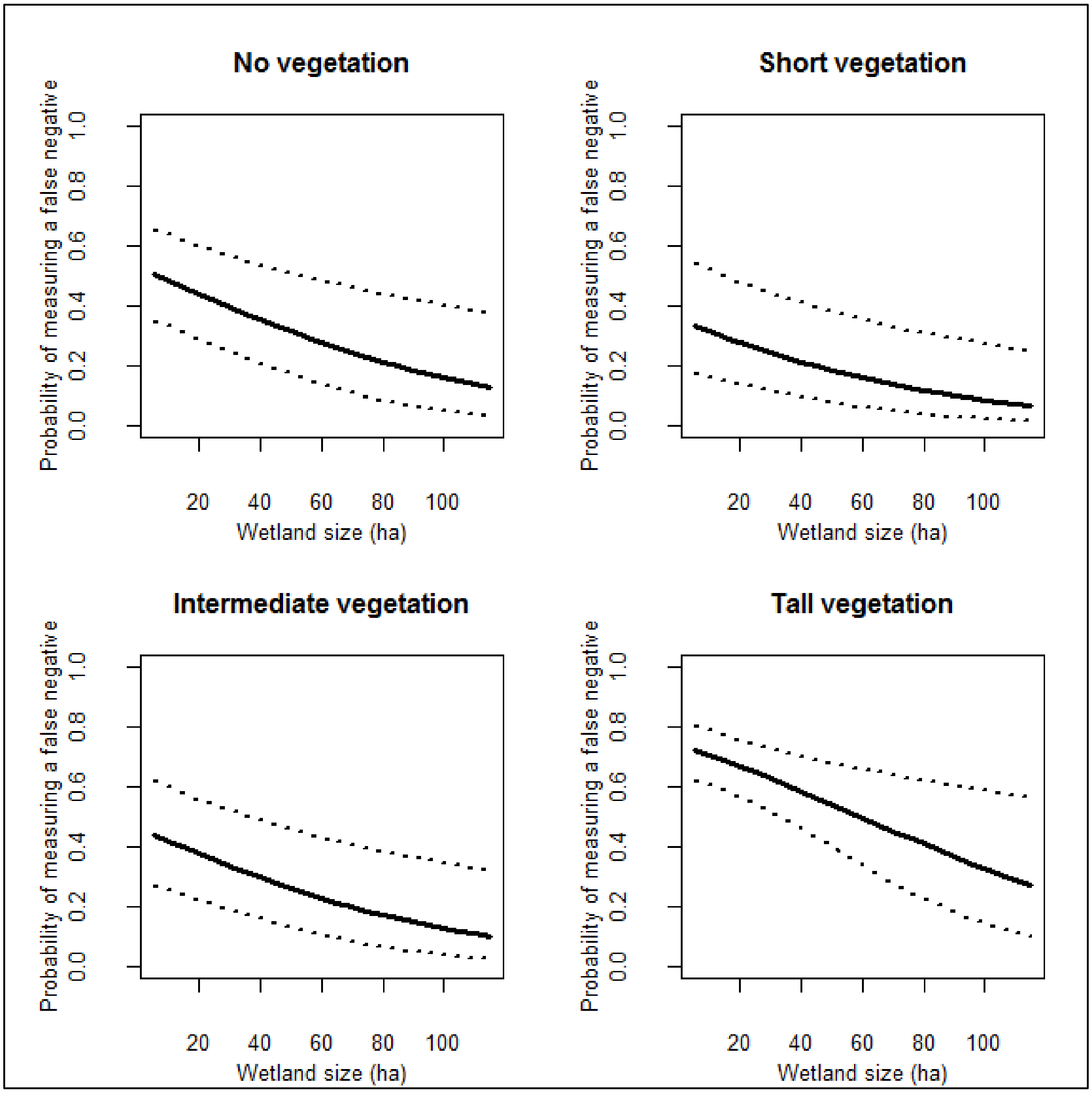


Figure 3.11 Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with intermediate water depths[b].

Back-transformed estimated mean probability of measuring a false negative for ducks (indicated by solid lines) and 95% confidence intervals (dashed lines) from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for intermediately[b] flooded ricelands in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth 15–30 cm.

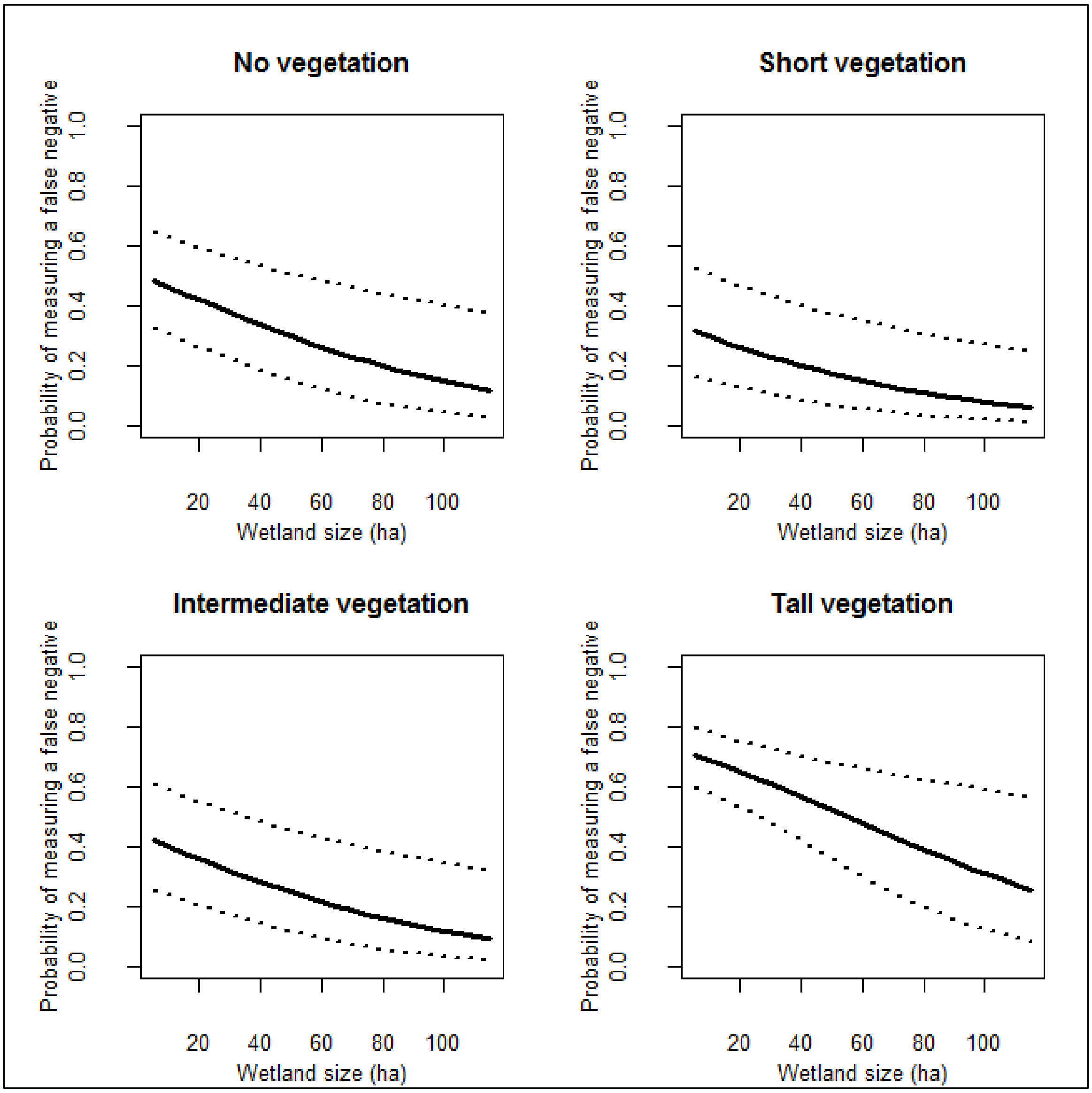


Figure 3.12 Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals by vegetation height[a] and wetland size for ricelands with deep water depths[b].

Back-transformed estimated mean probability of measuring a false negative for ducks (indicated by solid lines) and 95% confidence intervals (dashed lines) from zero-inflated negative binomial regression models by vegetation height[a] and wetland size for deeply[b] flooded ricelands in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] Short vegetation (1–15 cm); Intermediate vegetation (16-40 cm); Tall vegetation (>40 cm).

[b] Water depth >30 cm.

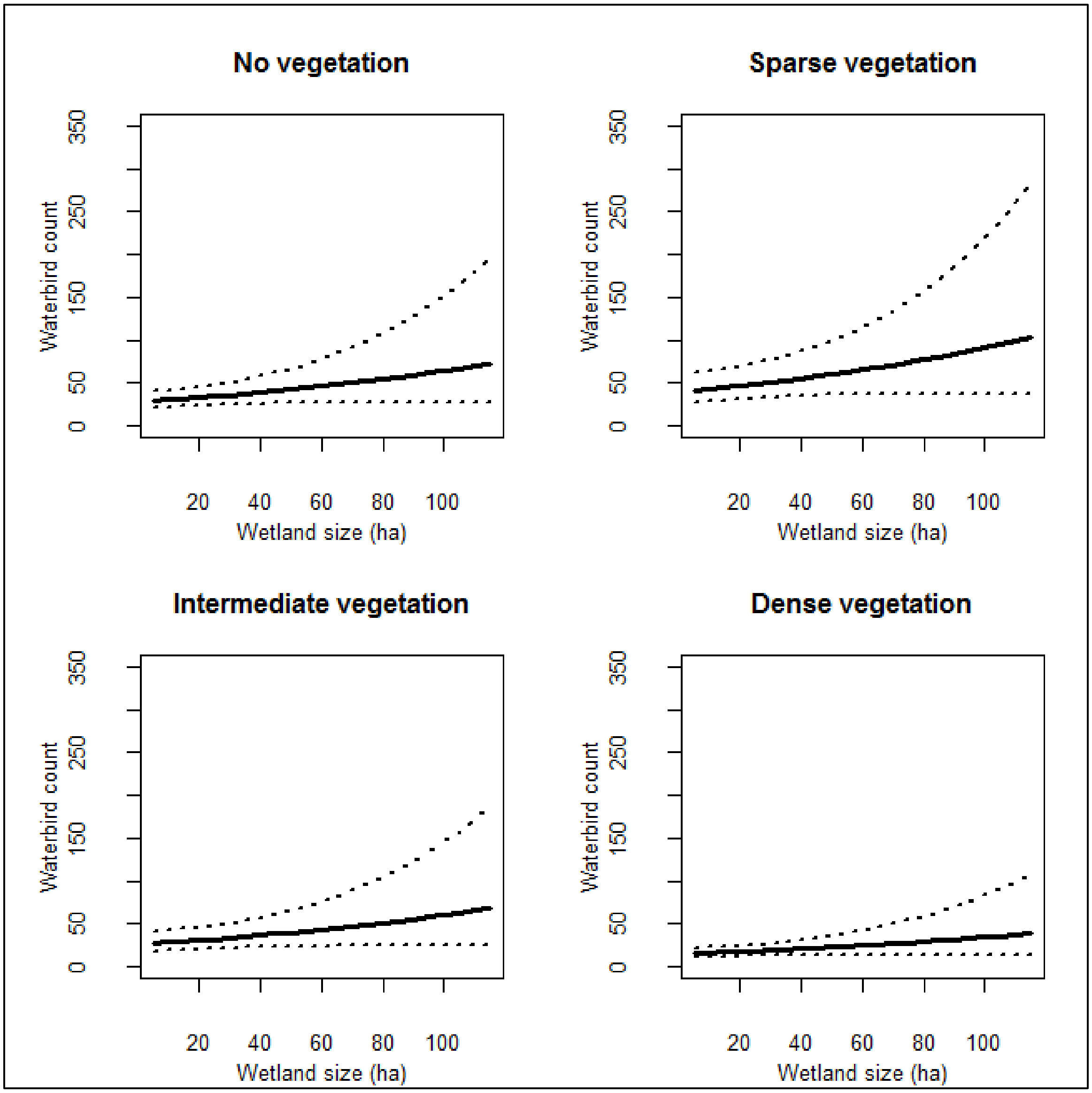


Figure 3.13 Estimated mean waterbird abundance and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with saturated soils[b].

Back-transformed estimated mean waterbird abundance (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for ricelands with saturated soils[b], in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth <1 cm.

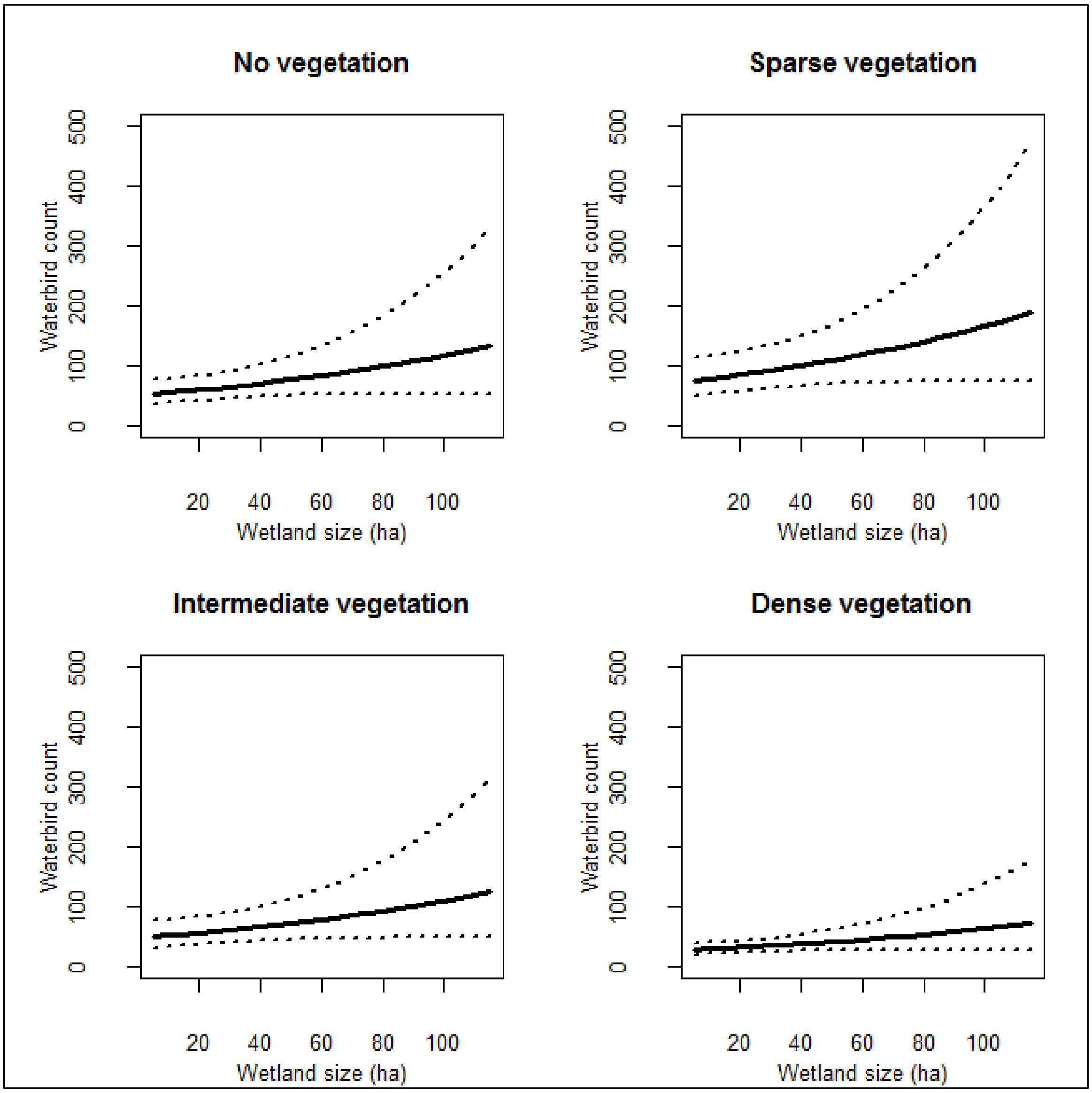


Figure 3.14 Estimated mean waterbird abundance and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with shallow water depths[b].

Back-transformed estimated mean waterbird abundance (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for shallowly[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 1–15cm.

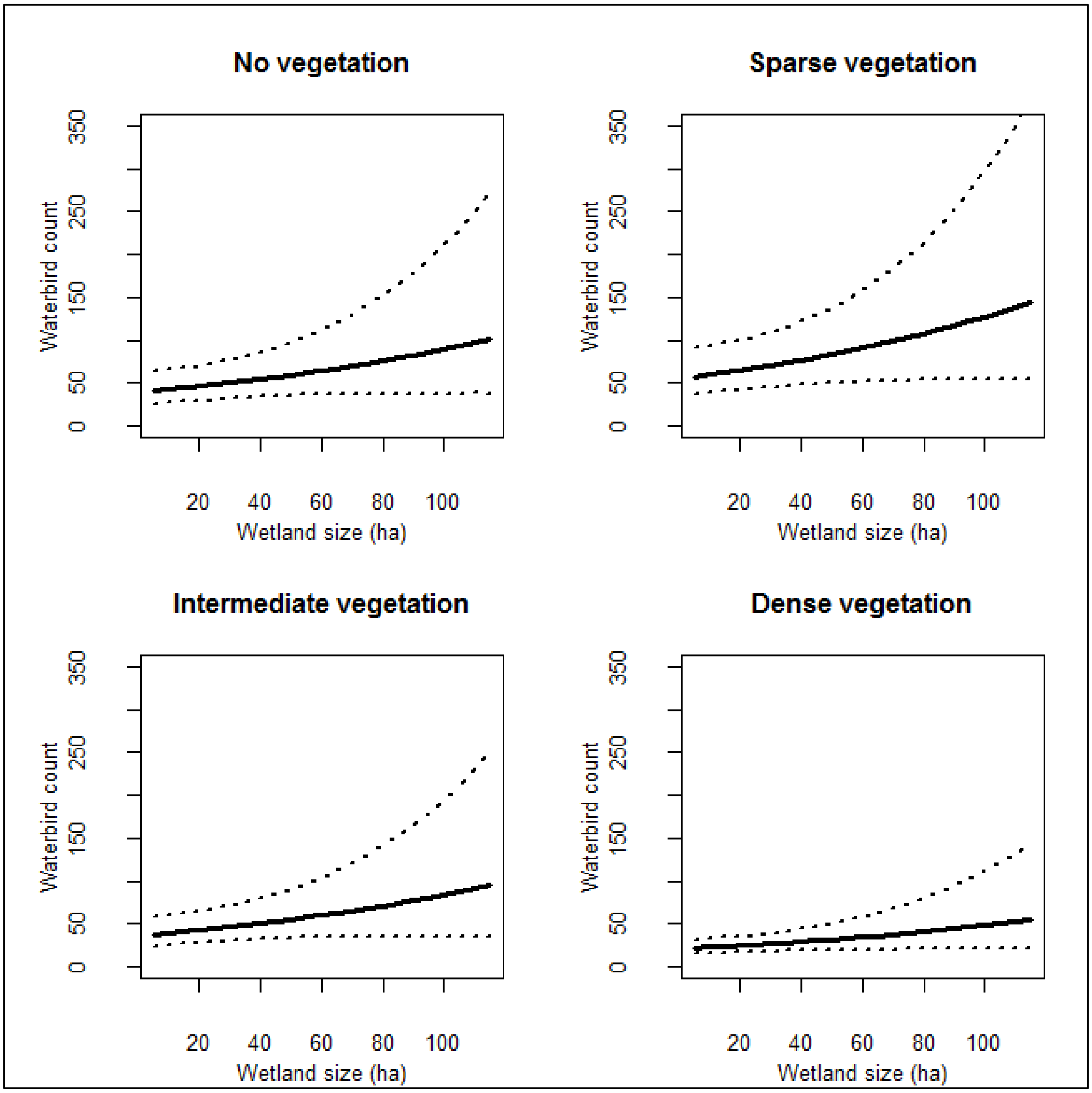


Figure 3.15 Estimated mean waterbird abundance and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with intermediate water depths[b].

Back-transformed estimated mean waterbird abundance (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for intermediately[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 15–30 cm.

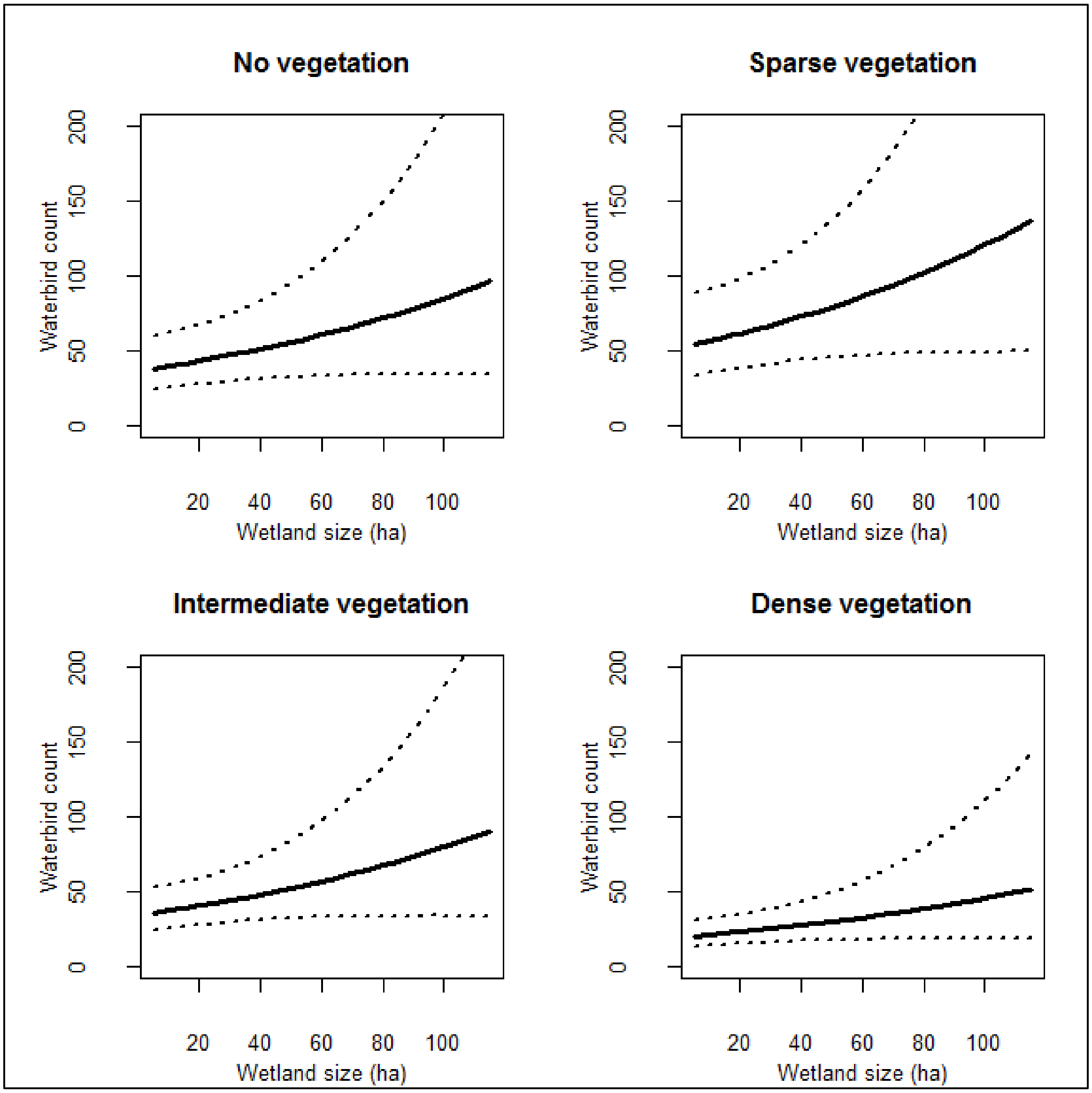


Figure 3.16 Estimated mean waterbird abundance and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with deep water depths[b].

Back-transformed estimated mean waterbird abundance (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for deeply[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth >30 cm.

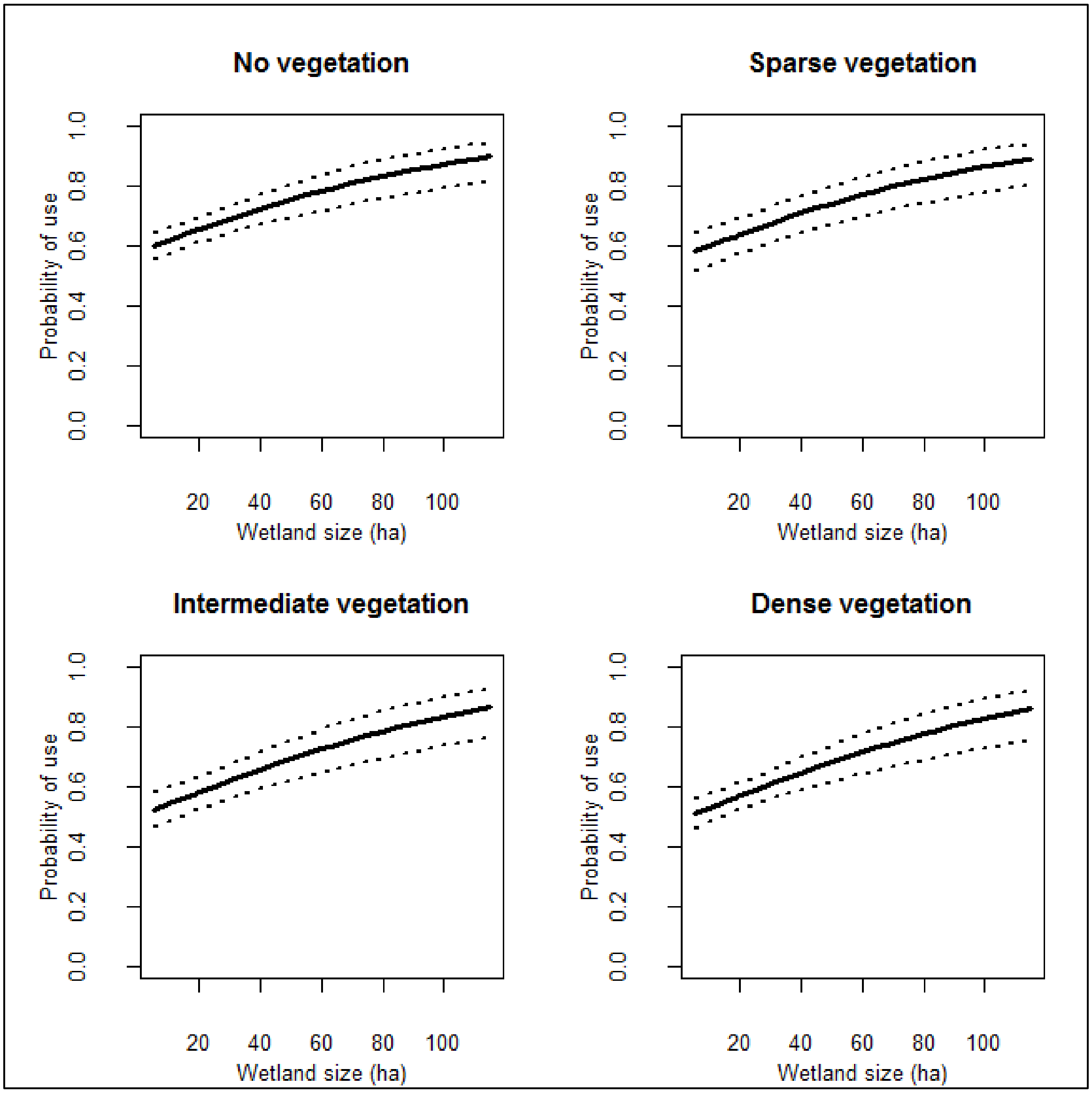


Figure 3.17 Estimated mean probability of waterbird use and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with saturated soils[b].

Back-transformed estimated mean probability of waterbird use (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for ricelands with saturated soils[b], in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth <1 cm.

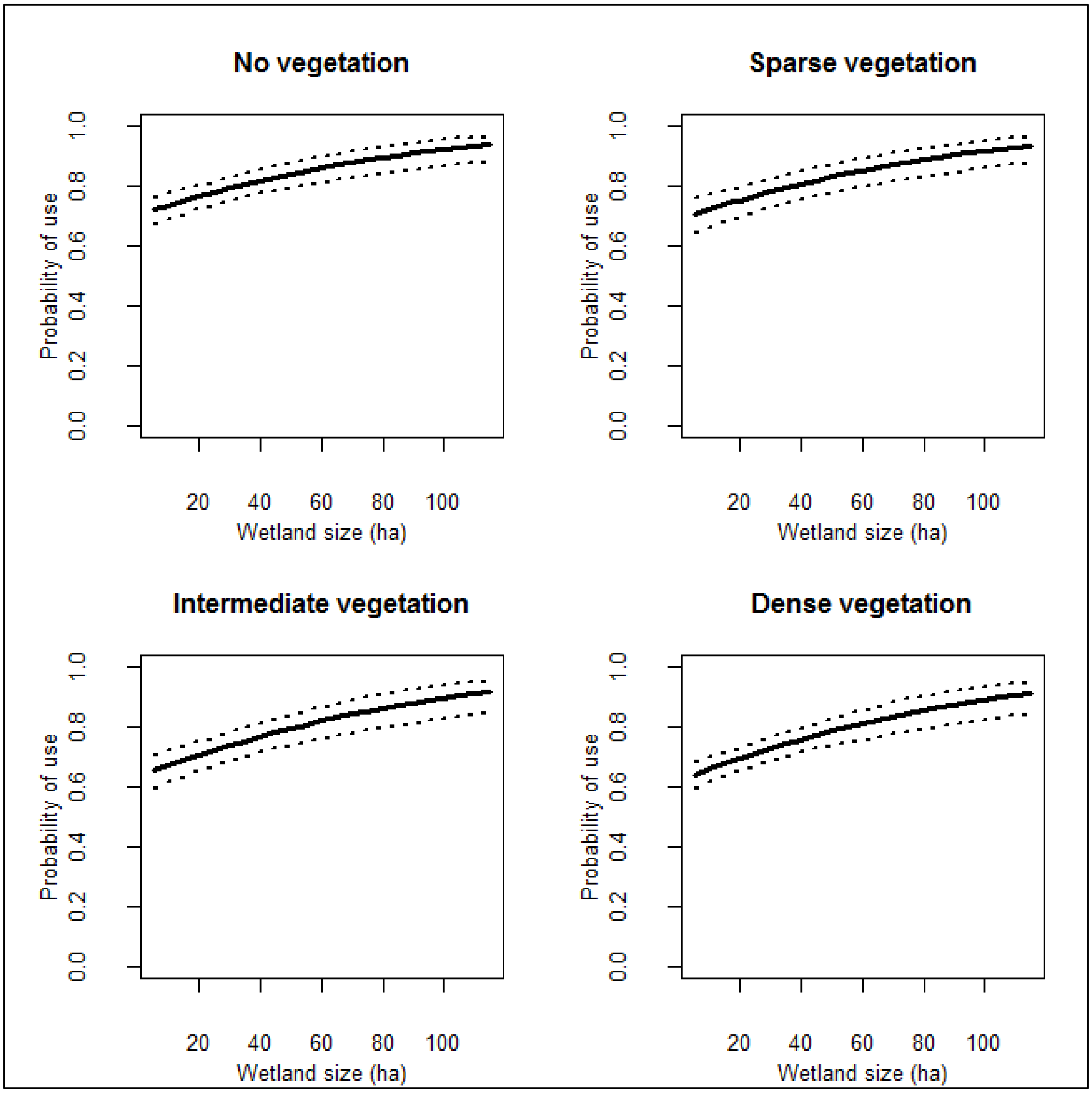


Figure 3.18 Estimated mean probability of waterbird use and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with shallow water depths[b].

Back-transformed estimated mean probability of waterbird use (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for shallowly[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 1–15cm.

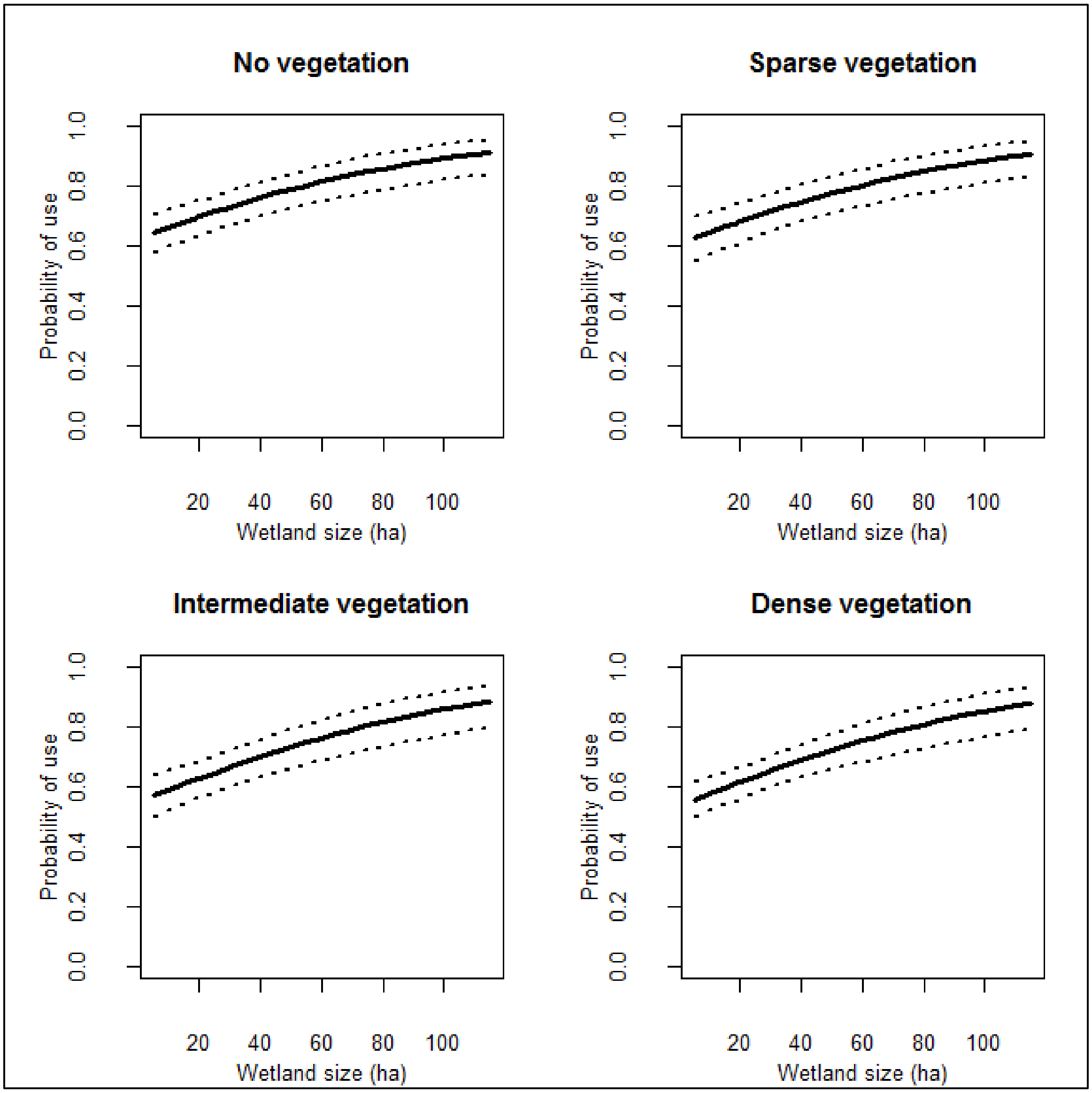


Figure 3.19 Estimated mean probability of waterbird use and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with intermediate water depths[b].

Back-transformed estimated mean probability of waterbird use (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for intermediately[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth 15–30 cm.

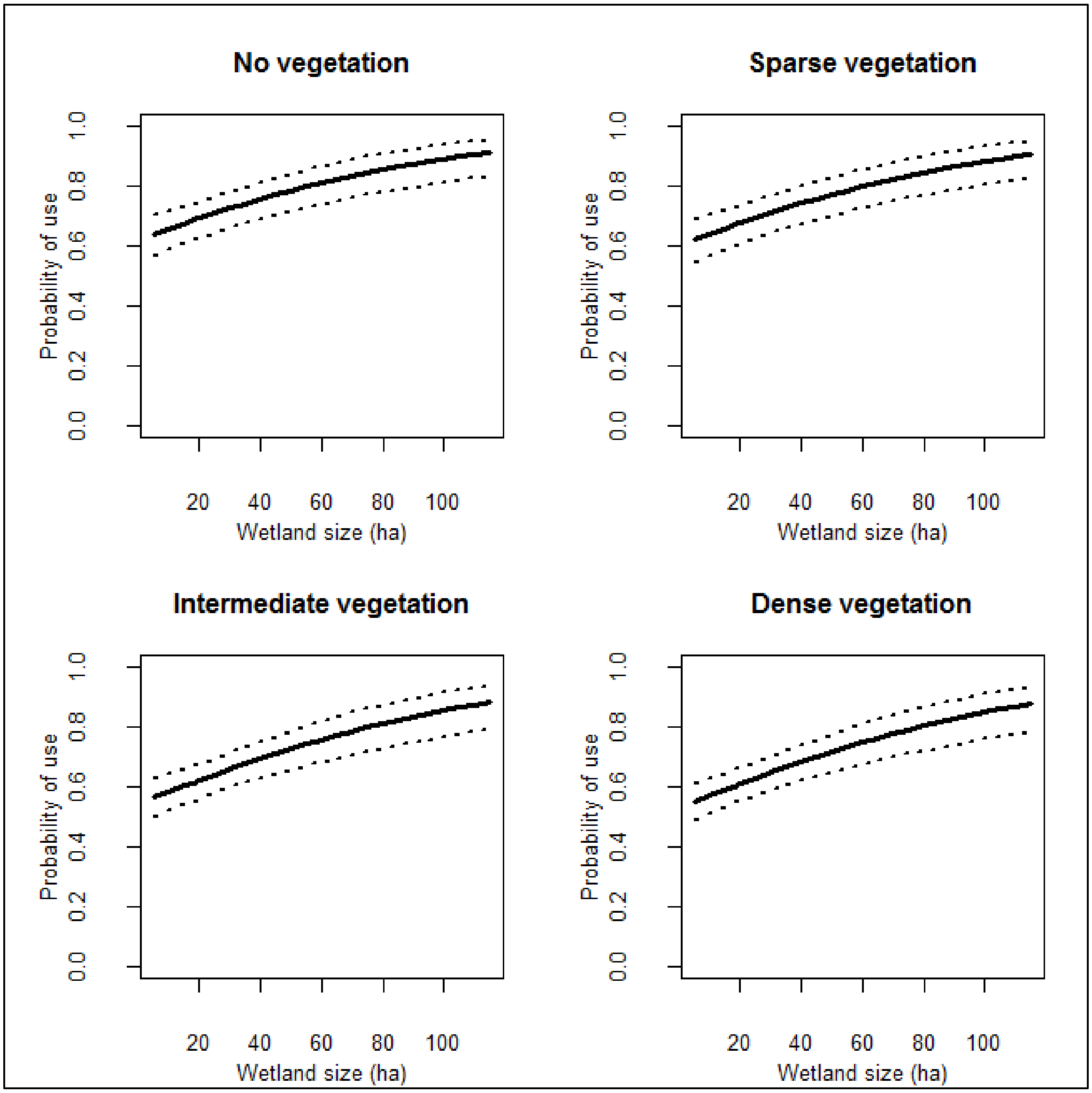


Figure 3.20 Estimated mean probability of waterbird use and 95% confidence intervals, by vertical vegetation density[a] and wetland size for ricelands with deep water depths[b].

Back-transformed estimated mean probability of waterbird use (indicated by solid lines) and 95% confidence intervals (dashed lines) from negative binomial Hurdle models by vegetation density[a] and wetland size for deeply[b] flooded ricelands, in the Gulf Coast Prairies of Louisiana and Texas, August–March, 2010–2013.

[a] No vegetation; Sparse (1–20 cm); Intermediate (21–40 cm); Dense (>40 cm).

[b] Water depth >30 cm.

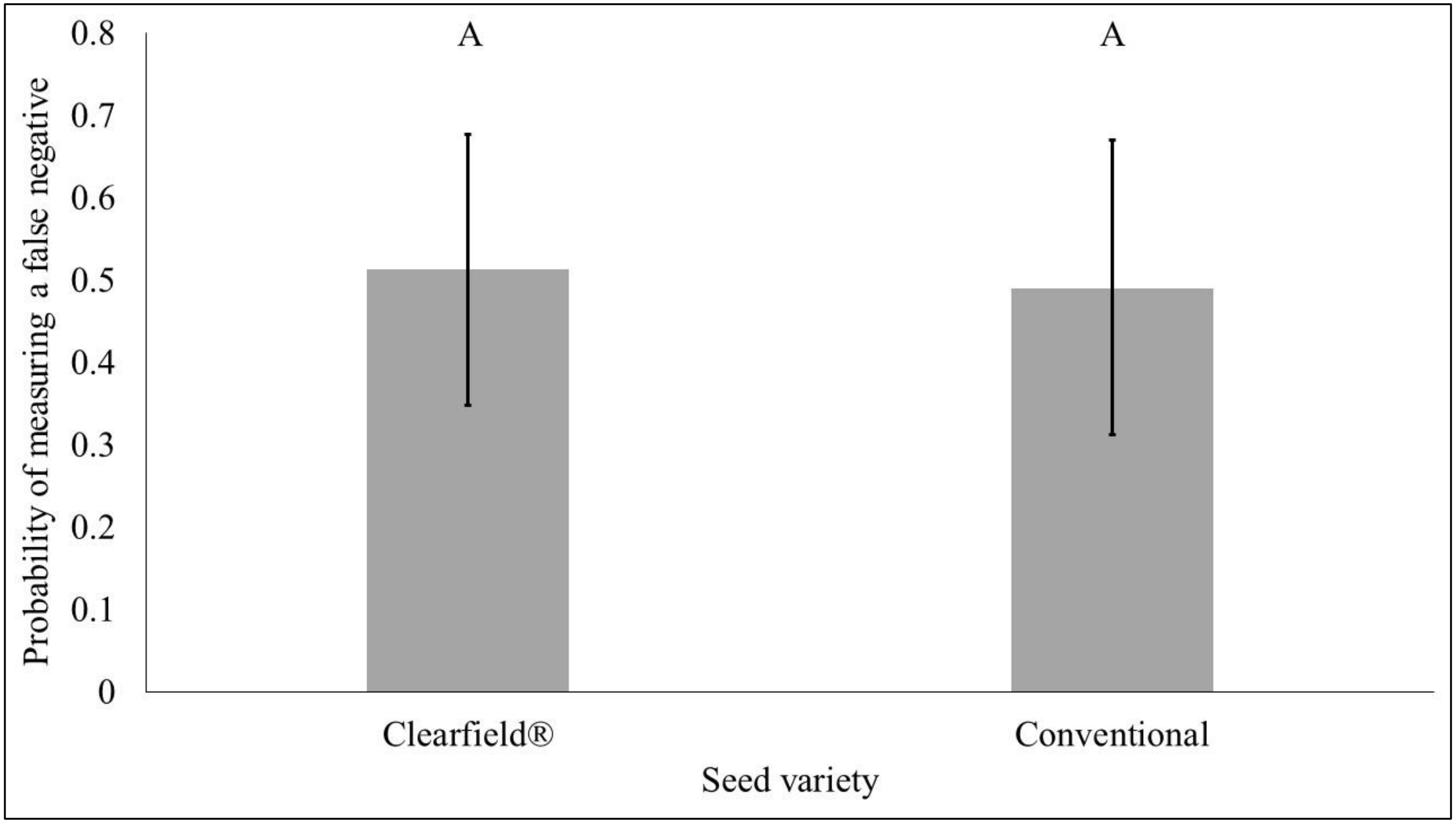


Figure 3.21 Estimated mean duck abundance and 95% confidence intervals by rice seed variety in Gulf Coast Prairie production fields.

Back-transformed estimated mean duck abundance and 95% confidence intervals from zero-inflated negative binomial regression models by rice seed variety in Gulf Coast Prairie production rice fields, August–March, 2010–2013. Same letter indicates no significant difference.

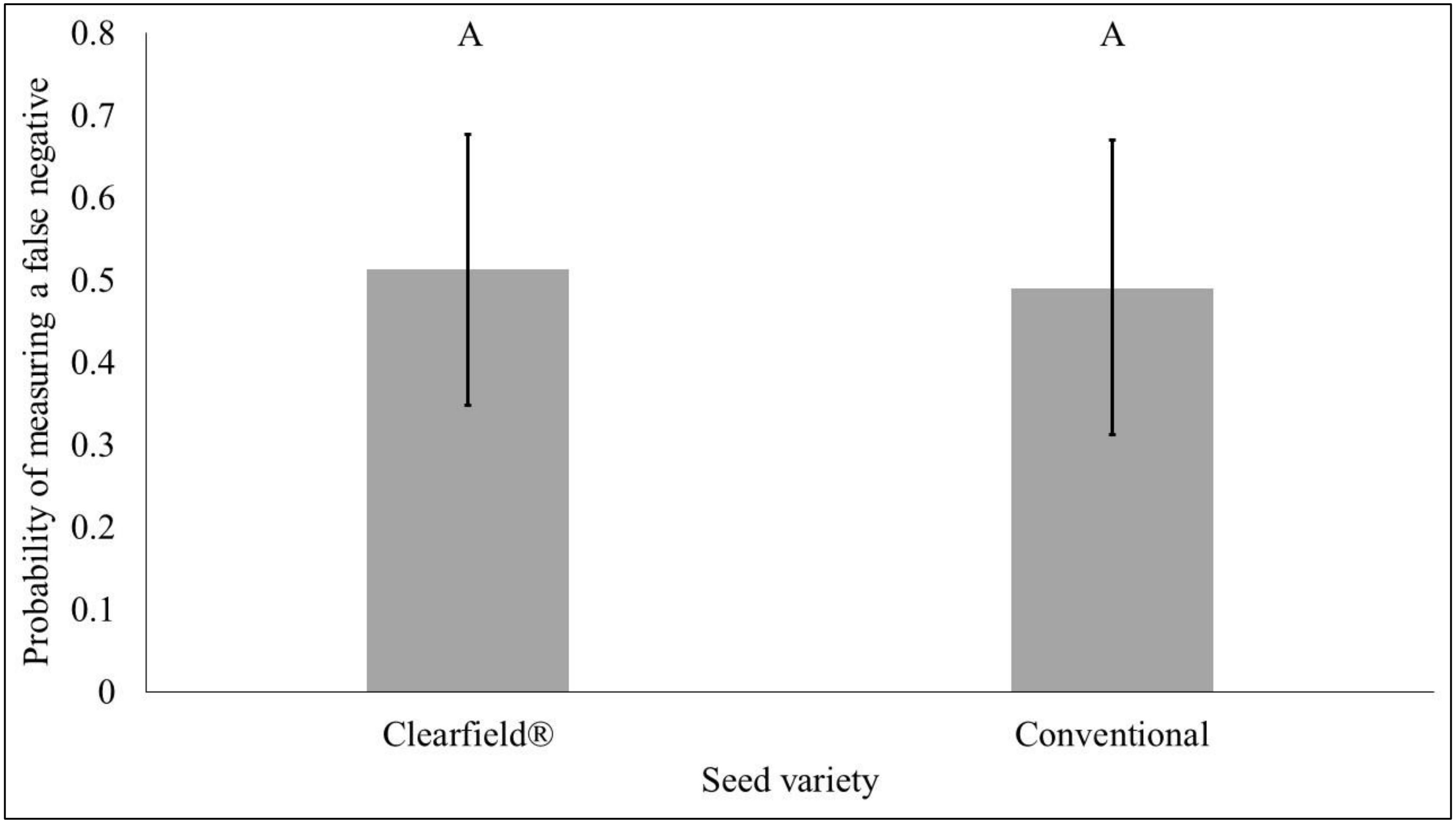


Figure 3.22 Estimated mean probability of measuring a false negative for ducks and 95% confidence intervals by rice seed variety in Gulf Coast Prairie production rice fields.

Back-transformed estimated mean probability of measuring a false negative for ducks and 95% confidence intervals from zero-inflated negative binomial regression models by rice seed variety in Gulf Coast Prairie production rice fields, August–March, 2010–2013. Same letter indicates no significant difference.

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CHAPTER IV

ESTIMATING SENSITIVITY TO ERROR IN WATERFOWL ENERGETIC CARRYING CAPACITY MODELS: AN INVESTIGATION OF FORAGING THRESHOLDS AND TRUE METABOLIZABLE ENERGY VALUES

The food-limitation hypothesis posits that nutrients may be limiting to organisms during a phase(s) of their life cycle. This hypothesis is rooted in classical studies of avian clutch size in relation to food availability (Lack 1946, Skutch 1949, Ashmole 1963). These and other studies broadened the knowledge of the nutrient-limitation hypothesis, centered on clutch size and other life history characteristics (Stearns 1976, Zammuto 1985), such as how nutrients influence longevity of life (Pianka 1970, Abrams 1983). By the late 1970s and 1980s, researchers began to realize that winter habitat conditions could influence recruitment of waterfowl on the breeding grounds (Heitmeyer and Fredrickson 1981, Kaminski and Gluesing 1987, Anderson and Batt 1983, Weller 1988, Smith et al. 1989). Since these initial studies, the potential effects of winter resource conditions (e.g., food and habitat) have been further explored (Raveling and Heitmeyer 1989, Shaeffer et al. 1998), and additional studies focused on how food influenced female body condition and nesting ecology, and subsequently duckling ecology (Arnold and Rohwer 1991, Batt et al. 1992). To further explore the effects of winter conditions on duck recruitment, a recent and more rigorous analysis confirmed previous work, again linking winter habitat conditions and waterfowl recruitment, specifically for midcontinent mallards (*Anas platyrhynchos*) and northern pintail (*A. acuta*; Osnas et al. 2016). Despite difficulty in establishing cause-and-effect of winter habitat conditions and recruitment in ducks, cross-seasonal effects seemingly have some degree of influence on populations of breeding waterfowl (Sedinger and Alisauskas 2014).

Studies linking winter resource conditions to waterfowl recruitment were an important impetus to developing a more holistic North American Waterfowl Management Plan (NAWMP) in 1986. The NAWMP established habitat and population goals for waterfowl species in North America, and charged Joint Ventures (JV) with implementing NAWMP recommendations at regional scales (U.S. Department of the Interior and Environmental Canada 1986, U.S. Department of the Interior et al. 2012). The Gulf Coast Joint Venture (GCJV) for example, seeks to provide foraging habitat capable of supporting approximately 14 million migrating and wintering waterfowl annually. Energetic carrying capacity of habitats for waterfowl is a fundamental concept used to prioritize habitat conservation efforts and improve planning, which could have implications for some populations of avian species (Pearse and Stafford 2014, Williams et al. 2014).

The food-limitation hypothesis is the primary guiding premise for conservation planning of JVs that occur in geographies of importance to waterfowl during migration and winter (i.e., the non-breeding periods; synthesized in Williams et al. 2014). Most JVs, including the GCJV, use bioenergetics models to estimate carrying capacity and project habitat needs for waterfowl during the non-breeding season. Daily ration models (DRM) are a type of resource depletion model used to estimate the theoretical carrying capacity of a given area (Miller and Newton 1999; Goss-Custard et al. 2002, 2003). Former (Reinecke et al. 1989) and contemporary (Williams et al. 2014) iterations of this model include waterfowl carrying capacity expressed in duck energy-days (DED):

$$\frac{\text{(Food available (kg [dry]/ha) x 1,000 g)x True metabolisable energy of each food (kcal/g [dry])}}{\text{Waterfowl daily energy expenditure kcal/day}} \qquad 4.1$$

While all DRMs require estimates of energy supply and demand, actual models used by JVs are far more complicated than this DED equation. For example, most JVs model energy supply and demand in time and space (e.g., Pacific Coast JV 2004, Central Valley JV 2006). Briefly, energy supplies may be influenced by natural or intentional flooding of habitats, and energy demand of birds may vary temporally based on population size, migration chronology, changes in species composition, physiological needs, weather, and other endogenous or exogenous factors (Williams et al. 2014). Nonetheless, all DRMs use some estimate of dietary energy available in waterfowl habitats (i.e., energy supply) and energy demands of target waterfowl populations. Thus, understanding abundance and dynamics of food on the landscape for non-breeding birds remains a viable contemporary research theme (Stafford et al. 2006; Hagy and Kaminski 2012*a*,*b*; Williams et al. 2014; Marty et al 2015).

Several studies have suggested that giving-up densities (GUD) and/or forage availability thresholds (FAT) of food may serve as a suitable foraging threshold for use in energetic carrying capacity models (Brown 1988, Reinecke et al. 1989, Greer et al. 2009, Hagy and Kaminski 2015). A GUD is a threshold of food abundance at which foragers cease eating in a patch to balance the metabolic costs of foraging, predation risk, and the missed opportunity costs of not foraging elsewhere (Brown 1988, Hagy and Kaminski 2015). In a simple environment where foragers are free to move among patches, a GUD of prey would be reached when intake rates decrease below those in other accessible habitats (Hagy and Kaminski 2015). A GUD of 50 kg/ha is used in most daily ration models when estimating waterfowl carrying capacity derived from mallard use of rice fields (Greer et al. 2009). Hagy and Kaminski (2015) found little evidence of a GUD for dabbling ducks wintering in the Mississippi Alluvial Valley; instead, they reported a food availability threshold (FAT) where predators (e.g., ducks) continued foraging but apparently did not acquire measurable food resources because food biomass remained relatively stable. A FAT occurs when food becomes functionally unavailable and predators fail to remove food despite active foraging (Hagy and Kaminski 2015). Hagy and Kaminski (2015) concluded that abundance of residual millet and other natural seeds and tubers was 3–4 times the GUD of waste rice with notable differences in residual seed biomass of natural seed taxa (i.e., 170.1 kg/ha; Range = 23.7–386.8 kg/ha). Estimates of residual foods remaining after foraging by dabbling ducks vary considerably (i.e., California, 30–163 kg/ha, Naylor 2002; Missouri, 43–56 kg/ha, Greer et al. 2009; Mississippi Alluvial Valley, 24–387 kg/ha, Hagy and Kaminski 2015). If GUD or FAT values for a given habitat type are not accurate estimates of food availability, subsequent habitat needs to meet desired bird objectives could be underestimated, or otherwise be unreliable (Hagy and Kaminski 2012b, Hagy and Kaminski 2015).

Pearse and Stafford (2014) investigated error propagation in waterfowl energetic carrying capacity models in the Mississippi Alluvial Valley and suggested that adjusting seed-biomass estimates was more complicated than previously described for currently accepted models (e.g., Reinecke and Loesch 1996, Esslinger and Wilson 2001, Wilson and Esslinger 2002, Central Valley Joint Venture 2006). Users of these models subtract a foraging threshold from an overall mean food biomass estimated by sampling multiple foraging patches (i.e., the mean-subtraction method; Pearse and Stafford 2014). Each patch with a food-biomass value below the foraging threshold is included in the data set as a negative number. Because fields cannot have negative amounts of food, Pearse and Stafford (2014) suggested recording negative and actual zero values as zero. For example, in the Mississippi Alluvial Valley, Stafford et al. (2006) estimated and reported a waste-rice biomass of 78 kg/ha. After subtracting a foraging threshold of 50 kg/ha (Greer et al. 2009) using the mean-subtraction method, they concluded that rice available to waterfowl averaged 28 kg/ha (Pearse and Stafford 2014). Inspection of field-specific estimates of waste-rice biomass revealed that 48% of fields contained less rice than the foraging threshold, and were included as negative numbers after subtracting the foraging threshold value (Pearse and Stafford 2014). Pearse and Stafford (2014) recommended applying a foraging threshold at the patch-level (i.e., field level). This approach increased the overall estimate of waste-rice biomass by 59%, to 45 kg/ha, because waste-rice biomass in fields equal to or below the foraging threshold were set to zero (Pearse and Stafford 2014). Moreover, these results suggest the importance of applying foraging thresholds at the correct ecological scale (Pearse and Stafford 2014).

True metabolizable energy value (TME; kcal/g) is the estimated amount of energy an individual bird derives from a specific food item, after accounting for metabolic fecal and urinary losses and endogenous metabolized energy (Miller and Reinecke 1984). A number of TME values for common waterfowl foods found in important waterfowl wintering areas have been estimated (Hoffman and Bookhout 1985, Reinecke et al. 1989, Sherfy 1999, Sherfy et al. 2001, Checkett et al. 2002, Kaminski et al. 2003, Ballard et al.

2004, Dugger et al. 2006). However, there is uncertainty associated with applying TME values to seed species other than the one from which it was derived (Williams et al. 2014). True metabolizable energy values are used to calculate available metabolizable energy by multiplying the mass of food items by its TME value and extrapolating the resulting energy value across an area of interest (Williams et al. 2014). Because limited TME values exist and species-specific seed-biomass estimates are likely not available, researchers are sometimes forced to apply a mean seed TME value derived from a related plant taxa (e.g., moist-soil plants; 2.47 kcals/g [Kaminski et al. 2003]). Little if any research investigating effects of incorporating species-specific TME values for natural seeds in carrying capacity models has been conducted. Current GCJV bioenergetics carrying capacity models use an average TME value for natural seeds (2.47 kcal/g; M. G. Brasher, GCJV, personal communication). Applying species-specific TME values to natural seed biomass may affect landscape scale estimates of available metabolic energy and habitat requirements. If TME values for abundant seeds are greater or less than the average TME value, the available metabolizable energy on a landscape may be accordingly over- or underestimated.

Application of foraging thresholds and TME values may be potential sources of bias in metabolizable energy and carrying capacity estimates (Williams et al. 2014). The ecological level at which a foraging threshold is applied, as well as the TME values of waterfowl foods used to estimate available metabolizable energy in ricelands, may result in discrepancies and gross over- or underestimation of energetic carrying capacity. Therefore, using contemporary waste-rice and natural seed-biomass estimates (Chapter I), my objectives in this dissertation were to: 1) investigate the effects that applying

GUDs and FATs at different ecological scales, and using average versus species-specific TME values have on available metabolizable energy (AME) estimates, and subsequent habitat requirements necessary to support LCP waterfowl populations from August–March; and 2) compare estimates of habitat requirements from my study to current GCJV estimates.

Study Area

I conducted my study in agricultural landscapes of the Chenier Plain (CP) of Louisiana and Texas and the Texas Mid-Coast (TMC; Chapters 2 and 3). The CP encompasses areas of southwest Louisiana and southeast Texas, roughly spanning from Lafayette, Louisiana westward to Houston, Texas and inland 130–160 km from the coastline of both states (Figure 2.1). The TMC extends from Galveston Bay to Corpus Christi, Texas and inland from the coastline approximately 170 km (Figure 2.1). My specific study area included the Louisiana parishes of Acadia, Allen, Calcasieu, Cameron, Evangeline, Jefferson Davis, St. Landry, and Vermilion, and the Texas counties of Brazoria, Chambers, Colorado, Jackson, Jefferson, Liberty, Matagorda, and Wharton. These counties aligned closely with the GCJV's Chenier Plain and Texas Mid-Coast Initiative Areas.

Historically, these regions contained extensive coastal marshes and tall grass prairies, freshwater wetlands, and savannahs. Today, the CP and TMC contain coastal marshes along the Gulf of Mexico, but coastal prairies and savannas have been converted largely to rice and other croplands (Esslinger and Wilson 2001). The climate is sub-tropical and humid with an average growing season of 270 days, 13 freeze-days per year, and temperatures ranging from 14° C in December–January to 30° C July–August

(Chabreck et al. 1989). Average annual precipitation decreases east to west in the CP from 144 cm near Lafayette, Louisiana, to 113 cm near Houston, Texas, and 77 cm near Corpus Christi, Texas (Gosselink et al. 1979, Hobaugh et al. 1989). The CP and TMC regions are subject to frequent and sometimes intense weather disturbances; on average, tropical storms make landfall approximately once every 1.6 years and hurricanes every 3.3 years (Roth 1999).

Methods

Sampling Design, Field Sampling, and Laboratory Procedures

I detailed sampling design, field sampling methods, and laboratory procedures in Chapter 2 (Pages 29–32).

Foraging Thresholds and True Metabolizable Energy Values

I subtracted a GUD of 50 kg/ha (Greer et al. 2009) and a FAT of 170 kg/ha (Hagy and Kaminski 2015) for production and idled rice fields, respectively. Additionally, I conducted a literature review to develop a database of species-specific TME values for natural seeds (Table 4.1)

Estimating Seed Biomass and Available Energy in Production and Idled Rice Fields

I used PROC MEANS in SAS v9.4 (SAS Institute 2015) to calculate mean biomass for waste rice and each taxon of natural seeds considered potential waterfowl food (Hagy and Kaminski 2012*a*, J. R. Marty, unpublished data, Chapter 1 [Table 1.1]) for each production and idled rice field sampled in the GCP, November, 2010–2013. I calculated functional seed biomass for each seed taxa and field by subtracting 50 kg/ha for production fields (Greer et al. 2009) and 170 kg/ha for idled fields (Hagy and

Kaminski 2015). A functional seed biomass is calculated by reducing raw density biomass for each seed species by a level equal to the proportional reduction in total raw biomass resulting from subtracting the foraging threshold from the total raw biomass. I applied foraging thresholds in two ways, similar to Pearse and Stafford (2014): 1) using the mean-subtraction method, where a foraging threshold is subtracted from the pooled (i.e., overall) mean food biomass, and 2) using the patch-level method, where each seed taxon within a field with a raw biomass value less than the foraging threshold was included in the dataset as a zero. For the mean-subtraction method, the foraging threshold is subtracted from the mean, which is equivalent to subtracting that constant from each observation and then averaging the resulting values (Pearse and Stafford 2014). Additionally, if subtracting the foraging threshold value from the estimated seed biomass of fields resulted in a value less than zero, the negative value was included in the dataset. However, when using the patch-level method, I truncated negative seed biomasses to zero because negative quantities of food are illogical (Pearse and Stafford 2014). Both methods yield identical results if all sampled fields contained food biomasses at or above the foraging threshold. However, when a portion of the sampled patches contained less seed than the foraging threshold, the patch-level method will yield a greater mean food biomass than the mean-subtraction method (Pearse and Stafford 2014).

After subtracting foraging thresholds using the mean-subtraction and patch-level methods, I used PROC MEANS to calculate a mean seed biomass for each seed taxon within each field classification for production and idled fields. Field classifications included: 1) fields harvested in August and again in November for a ratoon crop (harvested ratoon, HR); 2) fields in which a second crop was grown but not harvested and left standing, generally for crawfish aquaculture or waterfowl habitat (standing ratoon, SR); and 3) idle fields (standing or disked). Importantly, application of these field classifications were not mutually exclusive. For example, all production rice fields were harvested July–August, but each was then subjected to one of several unique practices (e.g., classifications 2–4) that impacted land use and vegetation conditions during autumn. Thus, some fields maybe viewed as a combination of farming activity and sampling period. Current GCJV models do not separate idled fields into disked and standing vegetation classifications, thus I pooled seed-biomass estimates from these two field classifications to derive estimates for a singular idled field classification. Additionally, GCJV models do not include energetic estimates for the field classification of no ratoon (i.e., fields harvested in July–August but with no ratoon crop grown), therefore I did not include any of my data from no ratoon fields. I developed two separate data sets containing functional seed biomasses for each field classification; one was derived using the mean-subtraction method and the other using the patch-level method.

Furthermore, for each of the two aforementioned data sets, I calculated an available metabolizable energy estimate per hectare for each field classification by: 1) using average TME values,

$$\sum \mathrm{FFD}_i \times \mathrm{TME}_i \tag{4.2}$$

where FFD_i was the available functional food biomass of each specific seed (g/ha), and TME_i was the TME value of rice (3.34 kcal/g; Kaminski et al. 2003) or the average value for natural seeds (2.47 kcal/g; Kaminski et al. 2003); and 2) using species-specific TME values, where FFD_i was the available functional food biomass of a specific seed (g/ha), and TME_i was the species-specific TME value which corresponded with FFD_i. If a species-specific TME value was not available for a natural seed species, I applied the average TME for natural seeds. Thus, I calculated four estimates of AME for each field classification derived using: 1) The mean-subtraction method and species-specific TME values (SMS), 2) the mean-subtraction method and the average TME value for natural seeds (AMS), 3) the patch-level method and species-specific TME values (SPL), and 4) the patch-level method and the average TME value for natural seeds (APL). I repeated all statistical analyses while varying the raw baseline seed biomass of each seed species within in each field ±10–50% (Miller and Newton 1999, Miller and Eadie 2006).

Estimating Habitat Requirements

I used calculation frameworks from existing GCJV bioenergetics habitat carrying capacity models to estimate area of flooded ricelands needed to support GCJV Louisiana Chenier Plain (LCP) waterfowl populations from August–March. Furthermore, I converted all metabolizable energy estimates from my study to kcal/ac estimates to align with GCJV methods and models. I converted habitat estimates back to hectares upon completion of statistical analyses. Current GCJV LCP models assume a 5-year mean riceland area of 129,553 ha, and a normal rice-idle field rotation of 2 years, where 10% of riceland area are idled, 40% of rice is ratooned, and 50% of ratooned rice is harvested. Additionally, GCJV bioenergetics models incorporate species-specific daily energy demands as calculated from equations in Miller and Eadie (2006). I used GCJV energy demands (kcal) for ducks and geese in LCP agricultural regions from August–March, derived from species-specific population objectives for the agricultural region. I considered LCP waterfowl energy demands baseline if unaltered (i.e., not varied ±10–50%; Table 4.2). Furthermore, I substituted GCJV energetic estimates (kcal/ac) from each field classification (i.e., I, HR, and SR) with energetic estimates derived from my study. Thus, I estimated required riceland habitat necessary to support GCJV waterfowl population from August–March (i.e., ~137days), while varying raw baseline seed biomass and LCP waterfowl energy demands ±10–50% (Miller and Newton 1999, Esslinger, and Wilson 2001, Miller and Eadie 2006).

Results

Mean-Subtraction and Patch-Level Seed-biomass estimates

November seed biomass (i.e., waste rice and natural seeds combined) in GCP ricelands was 262.8 kg/ha, 396.5 kg/ha, and 1,088.6 kg/ha for field classifications of I, HR, and SR, respectively (Table 4.3). After applying a foraging threshold using the mean-subtraction method (i.e., 170 kg/ha for I fields [Hagy and Kaminski 2015] and 50 kg/ha for HR and SR [Greer et al. 2010]), seed biomass estimated available to waterfowl averaged 92.8 kg/ha, 346.5 kg/ha, and 1,038.6 kg/ha for the aforementioned field classifications (Table 4.3). When correcting for a foraging threshold using the patch-level method, seed potentially available to waterfowl averaged 146.7 kg/ha, 347.6 kg/ha, and 1,038.6 kg/ha for the same field classifications, (Table 4.3). Only three HR fields (3%) and zero SR fields contained a seed biomass below the foraging threshold of 50 kg/ha. As a result, in HR fields, seed biomass differed by only 1.1 kg/ha between mean-subtraction and patch-level methods, and did not differ in SR fields (Table 4.3). However, 27% (*n* = 200) of idled fields contained a seed biomass below the foraging threshold of 170 kg/ha. Thus, when applying foraging thresholds using the patch-level method, the overall estimate of seed availability for idled fields increased 58% from 92.2 kg/ha to 146.7 kg/ha (Table 4.3).

Estimates of Available Metabolizable Energy

When using baseline (unaltered; not varied ±10–50%) seed biomass estimates from my study, AME was 239,733 kcal/ha, 1,020,346 kcal/ha, and 3,264,533 kcal/ha for I, HR, and SR, respectively when using AMS methodology (Table 4.4). When using SMS methods, AME was 3% (232,591 kcal/ha), 5% (973,830 kcal/ha), and 1% (3,236,405 kcal/ha) less than AMS methods for I, HR, and SR classifications, respectively (Table 4.4). When using APL methodology, AME was 374,485 kcal/ha, 1,023,206 kcal/ha, and 3,264,533 kcal/ha for field classifications of I, HR, and SR, respectively (Table 4.4). When using SPL methods, AME was 6% (357,204 kcal/ha), 5% (976,453 kcal/ha), and 1% (3,236,405 kcal/ha) less than APL methods for I, HR, and SR classifications, respectively (Table 4.4).

When reducing seed biomass estimates from my study by 50%, AME was -93,459 kcal/ha, 437,254 kcal/ha, and 1,554,670 kcal/ha for I, HR, and SR, respectively when using AMS methodology (Table 4.5). When using SMS methods, AME was 12% greater (-82,405 kcal/ha), 5% (417,180 kcal/ha) less, and 1% less (1,541,350 kcal/ha) than AMS methods for I, HR, and SR classifications, respectively (Table 4.5). When using APL methodology, AME was 96,740 kcal/ha, 443,133 kcal/ha, and 1,555,781 kcal/ha for field classifications of I, HR, and SR, respectively (Table 4.5). When using SPL methods, AME was 3% (93,538 kcal/ha), 5% (422,614 kcal/ha), and 1% (1,542,450 kcal/ha) less than APL methods for I, HR, and SR classifications, respectively (Table 4.5).

When increasing seed biomass estimates from my study 50%, AME was 572,920 kcal/ha, 1,603,442 kcal/ha, and 4,974,399 kcal/ha for I, HR, and SR, respectively when using AMS methodology (Table 4.6). When using SMS methods, AME was 4% (547,583 kcal/ha), 5% (1,530,484 kcal/ha), and 1% (4,931,465 kcal/ha) than AMS methods for I, HR, and SR classifications, respectively (Table 4.6). When using APL methodology, AME was 676,234 kcal/ha, 1,606,233 kcal/ha, and 4,974,399 kcal/ha for field classifications of I, HR, and SR, respectively (Table 4.6). When using SPL methods, AME was 5% (643,301 kcal/ha), 5% (1,533,032 kcal/ha), and 1% (4,931,465 kcal/ha) less than APL methods for I, HR, and SR classifications, respectively (Table 4.6).

Estimation of Habitat Requirements

Regardless of methods used to calculate AME estimates, and while holding waterfowl energy requirements constant, estimated area of ricelands required to support LCP waterfowl populations from August–March decreased exponentially as seed biomass increased up to 50% in production and idled ricelands (Figures 4.2–4.5). Regardless of methods used to calculate available metabolizable energy estimates, and while holding seed biomass constant, estimated area of ricelands required to support LCP waterfowl populations from August–March increased linearly as waterfowl energy demands increased up to 50% (Tables 4.7–4.10; Figures 4.6–4.9).

When holding seed biomass and energy demand estimates at baseline levels, regardless of using the mean-subtraction or patch-level methods, approximately 475 additional hectares of ricelands were necessary to support targeted baseline LCP waterfowl populations when using species-specific TME values instead of the average natural seed value (Figure 4.10). This trend increased to an additional 866 ha hectares as seed biomass was reduced up to 50% and decreased to an additional 314 ha as seed biomass was increased 50% (Figure 4.10). Lastly, the GCJV currently estimates that 16,305 ha of production and idled rice fields are required to support current LCP waterfowl population objectives. Estimates from my study were 10,029 ha (38%) greater than GCJV estimates, and indicated that as many as 26,334 ha of flooded production and idled rice fields could be required to support current LCP waterfowl population objectives from August–March when holding seed biomass constant.

Discussion

Estimates of Seed Biomass and Available Metabolizable Energy

Similar to simulations by Pearse and Stafford (2014), when I applied foraging thresholds at the patch (i.e., field) level as opposed to using the mean subtraction method, estimates of seed biomass in idled rice fields increased 58%. Seed biomass estimates in HR and SR fields rarely fell below the rice field foraging threshold of 50 kg/ha; thus, estimates for HR (347.7 kg/ha) and SR fields (1038.6 kg/ha) remained unchanged regardless of the scale of which foraging thresholds were applied. My results and those of Pearse and Stafford (2014) indicate the importance of applying foraging thresholds at the proper scale. Similar to results and recommendations from Pearse and Stafford (2014), foraging thresholds used in this study were derived at the patch level (Greer et al. 2009; Hagy and Kaminski 2015). Inaccurate habitat objectives may result from adjusting food biomass across an entire landscape, which is common practice in some bioenergetics models (Esslinger and Wilson 2001, Wilson and Esslinger 2002). Nonetheless, our understanding of waterfowl foraging is limited, and foraging patches in reality may be smaller or larger than individual production or idled rice fields. Research investigating how, and at what spatial scale waterfowl perceive, evaluate, select, exploit foraging patches and derive nutrient rewards from them would be beneficial for developing reliable carrying capacity models. Regardless of how foraging thresholds are integrated into bioenergetics models, if there is not consistency across conservation planning regions, landscape-scape scale conservation planning efforts may be tenuous (Pearse and Stafford 2014, Williams et al. 2014).

If seeds with TME values less or greater than the average account for a large proportion of biomass estimates, available energy in each field or across an entire landscape may be over- or under-estimated. Although my results were not terribly sensitive to this issue, the magnitude of sensitivity depends on the composition the natural seed community. True metabolizable energy values for some seeds included in models for this study were as much as 80% less (i.e., *Eleocharis* spp.; 0.50 kcal/g; Dugger et al. 2006) and 22% greater (*Sagittaria* spp.; 3.04 kcal/g; Hoffman and Bookhout 1985) than the average estimate of 2.47 kcal/g (Kaminski et al. 2003). In my study, AME in production and idled rice fields was 1–4.6% less when using species-specific TME values than when using the average TME value for both mean subtraction and patch level methods among all field classifications. Prior to this study, Gulf Coast Joint Venture planners, and likely other JV planners, did not have access to species-specific seed-biomass estimates to incorporate into bioenergetics models. When using species-specific TME values, seed biomass must be known for each individual seed species. This is a potential drawback and limiting factor, because deriving species-specific biomass estimates is costly and labor intensive.

Estimation of Habitat Requirements

The amount of flooded ricelands required to satisfy energy demands of GCJV LCP waterfowl population objectives decreased exponentially as seed biomass (waste rice and natural seeds combined) increased from -50% to +50%, regardless of estimation methods (i.e., SMS, AMS, SPL, and APL). This result occurred because, as seed biomass increased from -50% up to +50%, the number of production and idled rice fields with seed biomass less than foraging thresholds was reduced. As seed biomass increased up to 50%, nearly all production and idled rice fields contained a biomass above foraging thresholds, and the mean subtraction and patch level methods indicated that required riceland habitat was practically identical. When population objectives and seed biomass were held at baseline levels, the amount of necessary habitat was approximately 350 ha greater when using mean-subtraction than patch-level methods, similar to results by Pearse and Stafford (2014). Thus, current bioenergetics models are likely overestimating habitat objectives by subtracting foraging thresholds from landscape scale estimates of seed biomass. Holding population objectives and seed biomass constant, approximately 475 fewer hectares of habitat was required when using an average TME value for natural seeds, than when using species-specific TME values, suggesting that use of an average TME value in carrying capacity models may underestimate habitat requirements.

Results emphasized, that reducing seed biomass and increasing population objectives of waterfowl would have major consequences for waterfowl energy demands and habitat requirements. For example, approximately 97,000 ha of flooded ricelands would be required to support LCP waterfowl populations if a 50% reduction in seed biomass and a 50% increase in energy demands occurred. Indicating an additional

71,000 ha of flooded ricelands would be required when compared to baseline habitat requirements from this study; and an additional 81,000 ha would be required when compared to current GCJV habitat requirements. Miller and Newton (1999) reported similar results for California, where decreasing rice biomass 50% and doubling northern pintail populations resulted in a required area of ricelands 4–5 times greater than if all estimates remained at baseline levels. A future increase in harvester efficiency or development of farming practices better adept to controlling natural seed growth and production may be cause for concern among conservation planners. This may be especially true if the coastal marsh loss crisis along the Gulf Coast continues, habitat fragmentation occurs, area of planted rice declines, or water restrictions are implemented in the LCP regions. Thus, conservation planners should endeavor to find strategies to incentivize agricultural producers for flooding of idled and post-harvest production rice fields, as was done through the Migratory Bird Habitat Initiative.

Estimated SPL flooded riceland habitat required to support baseline waterfowl energy demands in the LCP was ~26,000 ha, which was approximately 10,000 ha greater than that currently estimated by GCJV estimate. I observed this outcome, because seed-biomass estimates derived from my study were less than those currently used in GCJV bioenergetics models (Chapter I). Habitat estimates from my study indicated that approximately 20% of all production and idled rice field hectarage would need be flooded to support waterfowl populations from August–March annually. This estimate would even be larger, because other birds (e.g., American coots [*Fulica americana*], blackbirds [*Agelaius* spp.], doves [*Zenaida* spp.], and sandhill cranes [*Grus canadensis*]) often forage in these habitats (Crase and DeHaven 1978, Day and Colwell 1998, Eadie et al. 2008).

Management Implications

Current GCJV bioenergetics models may underestimate area of flooded riceland habitat necessary to support target waterfowl populations from August–March annually. Differences between contemporary seed biomass estimates (Chapter I) and estimates currently employed in GCJV bioenergetics models are the driving force behind differences in estimates habitat requirements. Other factors having a weaker affect include the utilization of species-specific TME values, and the use of patch-level methods to subtract foraging thresholds. Current methods employed in daily ration carrying capacity models that subtract foraging thresholds from landscape scale seed-biomass estimates and apply an average TME value to natural seed biomass may output biased results. Thus, I recommend that GCJV conservation planners adopt the patch-level method for applying foraging thresholds, because seed-biomass estimates were calculated at the field level, and it is presumably the ecological scale at which waterfowl forage (Pearse and Stafford 2014). Additionally, using average TME values may underestimate habitat objectives. Therefore, I recommend that GCJV conservation planners only use an average TME value for natural seeds when seed-specific values are not available. I recommend use of species-specific TME values in bioenergetics models. Thus, use of species-specific TME values in combination with patch-level methods of subtracting foraging thresholds will result in increasingly accurate estimates of required habitat in GCJV bioenergetics carrying capacity models.

The cost associated with flooding wetland habitats is variable and dependent on flooding techniques (e.g., electric vs. diesel pumps, rain water, etc.), depth, and environmental conditions (temperature, rainfall, etc.). Manley et al. (2008) estimated the cost of flooding one hectare of rice to be $12.72–25.45/ha (2016 USD). Therefore, the cost of providing enough flooded riceland habitat to support annual LCP waterfowl energetic needs would be approximately $330,720–635,700. Thus, conservation organizations would need to dedicate an additional $127,200–244,500 annually toward the flooding of LCP ricelands.

Additional factors that will likely need to be addressed if conservation planners wish to develop more accurate carrying capacity models include the difficulty and error associated with estimating area of wetland habitats correctly, in addition to the potential impacts of human developments and disturbance which may cause waterfowl to avoid wetlands, thus reducing carrying capacity (Williams et al. 2014). Furthermore, I recommend continuity in carrying capacity models among habitat regions. This will serve to help conservation planners understand conservation issues and priorities on a broader scale going forward into the future.

The use of agent-based models for waterfowl and wetland conservation, a technique that links behavior of individuals with population- or community-level processes (a bottom-up approach), are potential alternatives to current daily-ration models (Miller et al. 2013, Williams et al. 2014). A spatially explicit waterbird agent-based model (SWAMP), developed by Miller et al. (2013) in the Central Valley of California provides the framework for an attempt to use agent-based models as a decision support tool to guide waterfowl conservation and management in North America. I recommend the GCJV conservation planers investigate the use of agent-based models as an alternative to daily-ration bioenergetics models for estimating habitat carrying capacity, and for wetlands conservation. Regardless of methods used (daily-ration or agent-based models), an update to current GCJV conservation models seems justified and would provide conservation planers with contemporary and seemingly more accurate estimates of agricultural (and coastal marsh) habitat requirements.

Table 4.1 True metabolizable energy values for seed taxa consumed by dabbling ducks.

Common name	Taxon	TME value[a]	Reference[b]
Sedge (seeds)	*Cyperus* spp.	1.96	2
Sedge (tubers)	*Cyperus* spp.	2.50	4
Crabgrass	*Digitaria* spp.	3.10	2
Virginia buttonweed	*Diodia virginiana*	2.47	10
Barnyardgrass	*Echinochloa* spp.	2.67	Mean from 3,4,6,7,8
Spikerush	*Eleocharis* spp.	0.50	3
Morningglory	*Ipomoea* spp.	2.47	10
Sprangletop	*Leptochloa* spp.	2.47	10
Rice	*Oryza sativa*	3.34	6
Panicgrass	*Panicum* spp.	2.65	Mean from 2,7
Dallisgrass	*Paspalum* spp.	1.57	2
Swamp smartweed	*Polygonum hydropiperoides*	1.59	1
Pennsylvania smartweed	*P. pensylvanicum*	1.24	Mean from 4,8
Beaksedge	*Rhynchospora corniculata*	1.86	2
Curly Dock	*Rumex crispus*	2.47	10
Arrowhead	*Sagittaria* spp.	3.06	4
Foxtail grass	*Setaria* spp.	2.88	2
Signal grass	*Urochloa* spp.	2.73	9

[a] TME, True metabolizable energy

[b] 1 – Ballard et al. (2004), 2 – Checkett et al. (2002), 3 – Dugger et al. (2006), 4 – Hoffman and Bookhout (1985), 5 – Petrie unpublished report, 6 – Reinecke et al. (1989), 7 – Sherfy (1999), 8 – Sherfy et al. (2001), 9 – mean for Poaceae, Straub (2008), 10 – mean TME for natural seeds, Kaminski et al (2003).

Table 4.2 Baseline energy demands (kcal) for ducks and geese in agricultural regions of the Louisiana Chenier Plain from August–March, derived from Gulf Coast Joint Venture bioenergetics carrying capacity models.

	late Aug	early Sep	late Sep	early Oct	late Oct	early Nov	late Nov	early Dec	late Dec
Ducks	3.92E+08	1.53E+09	1.75E+09	2.2E+09	3.19E+09	4.33E+09	4.79E+09	5.44E+09	5.42E+09
Geese	0	525905.5	525905.5	7.55E+08	8.06E+08	3.2E+09	3.2E+09	2.68E+09	2.86E+09
Total	3.92E+08	1.53E+09	1.75E+09	2.96E+09	4E+09	7.53E+09	7.98E+09	8.12E+09	8.27E+09
Cumulative	3.92E+08	1.92E+09	3.67E+09	6.63E+09	1.06E+10	1.82E+10	2.61E+10	3.43E+10	4.25E+10

	early Jan	late Jan	early Feb	late Feb	early Mar	late Mar
Ducks	5.77E+09	5.18E+09	2.83E+09	2.6E+09	3.09E+09	1.74E+09
Geese	4.12E+09	4.4E+09	3.24E+09	2.86E+09	1.55E+08	1.65E+08
Total	9.89E+09	9.58E+09	6.07E+09	5.47E+09	3.25E+09	1.9E+09
Cumulative	5.24E+10	6.2E+10	6.81E+10	7.35E+10	7.68E+10	7.87E+10

Table 4.3 Baseline seed-biomass estimates before and after subtraction of foraging thresholds in Gulf Coast Prairie ricelands.

Foraging threshold method[a]	Field classification[b]	Seed biomass[c] $\bar{x}$
None	I	262.82
	HR	396.52
	SR	1088.57
Mean-subtraction method	I	92.82
	HR	346.52
	SR	1038.57
Patch-level method	I	146.72
	HR	347.65
	SR	1038.58

Foraging threshold calculation methods, field classifications, and seed-biomass estimates (kg[dry]/ha) before and after subtraction of foraging thresholds in production (50 kg/ha) and idled rice fields (170 kg/ha) in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

[a] Blanks denote same field classification.

[b] I, Idled rice; HR, harvested ratoon; SR, standing ratoon.

[c] Waste rice and natural seeds combined (kg/ha).

Table 4.4 Baseline available metabolizable energy estimates in Gulf Coast Prairie ricelands.

AME estimation method[a,b]	Field classification[c]	AME[d]
AMS	I	239,733
	HR	1,020,346
	SR	3,264,533
SMS	I	232,591
	HR	973,830
	SR	3,236,405
APL	I	374,485
	HR	1,023,206
	SR	3,264,533
SPL	I	357,204
	HR	976,453
	SR	3,236,405

Available metabolizable energy estimation methods, field classifications, and baseline available metabolizable energy estimates (kcal/ha) before and after subtraction of foraging thresholds in production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November, 2010–2013.

[a] Blanks denote same available metabolizable energy (AME) estimation method.

[b] AMS–average true metabolizable energy (TME) values with means subtraction methodology; SMS–species-specific TME values with mean-subtraction methodology; APL–average TME values with patch-level methodology; SPL–species-specific TME values with patch-level methodology.

[c] I, Idled rice; HR, harvested ratoon; SR, standing ratoon.

[d] Available metabolizable energy (kcal/ha).

Table 4.5 Available metabolizable energy estimates in Gulf Coast Prairie ricelands after a 50% reduction in raw seed biomass.

AME estimation method[a,b]	Field classification[c]	AME[d]
AMS	I	-93,459
	HR	437,254
	SR	1,554,670
SMS	I	-82,405
	HR	417,180
	SR	1,541,350
APL	I	96,740
	HR	443,133
	SR	1,555,781
SPL	I	93,538
	HR	422,614
	SR	1,542,450

Available metabolizable energy estimation methods, field classifications, and baseline available metabolizable energy estimates (kcal/ha) before and after subtraction of foraging thresholds in production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November 2010–2013.

[a] Blanks denote same available metabolizable energy (AME) estimation method.

[b] AMS–average true metabolizable energy (TME) values with means subtraction methodology; SMS–species-specific TME values with mean subtraction methodology; APL–average TME values with patch level methodology; SPL–species-specific TME values with patch level methodology.

[c] I, Idled rice; HR, harvested ratoon; SR, standing ratoon.

[d] Available metabolizable energy (kcal/ha).

Table 4.6 Available metabolizable energy estimates in Gulf Coast Prairie ricelands after a 50% increase in raw seed biomass.

AME estimation method[a,b]	Field classification[c]	AME[d]
AMS	I	572,920
	HR	1,603,442
	SR	4,974,399
SMS	I	547,583
	HR	1,530,484
	SR	4,931,465
APL	I	676,234
	HR	1,606,233
	SR	4,974,399
SPL	I	643,301
	HR	1,533,032
	SR	4,931,465

Available metabolizable energy estimation methods, field classifications, and baseline available metabolizable energy estimates (kcal/ha) before and after subtraction of foraging thresholds in production and idled rice fields in the Gulf Coast Prairies of Louisiana and Texas, August–November 2010–2013.

[a] Blanks denote same available metabolizable energy (AME) estimation method.

[b] AMS–average true metabolizable energy (TME) values with means subtraction methodology; SMS–species-specific TME values with mean subtraction methodology; APL–average TME values with patch level methodology; SPL–species-specific TME values with patch level methodology.

[c] I, Idled rice; HR, harvested ratoon; SR, standing ratoon.

[d] Available metabolizable energy (kcal/ha).

Table 4.7 AMS[a] estimated habitat required (thousands; ha) to support Louisiana Chenier Plain waterfowl energy demands from August–March.

		% change in raw seed biomass										
		-50%	-40%	-30%	-20%	-10%	0%	10%	20%	30%	40%	50%
% change in waterfowl energy demand	-50%	25.94	20.73	17.27	14.79	12.94	11.50	10.35	9.40	8.62	7.95	7.39
	-40%	31.83	25.44	21.19	18.16	15.88	14.11	12.70	11.54	10.58	9.77	9.07
	-30%	37.97	30.36	25.29	21.67	18.96	16.85	15.16	13.78	12.63	11.66	10.83
	-20%	44.40	35.51	29.58	25.35	22.18	19.72	17.74	16.13	14.79	13.65	12.67
	-10%	51.12	40.89	34.08	29.21	25.56	22.72	20.45	18.59	17.04	15.73	14.61
	0%	58.16	46.54	38.79	33.25	29.10	25.87	23.28	21.17	19.40	17.91	16.63
	10%	65.53	52.46	43.73	37.49	32.81	29.17	26.26	23.87	21.89	20.20	18.76
	20%	73.27	58.66	48.91	41.94	36.71	32.64	29.38	26.72	24.49	22.61	21.00
	30%	81.39	65.19	54.36	46.62	40.81	36.29	32.67	29.70	27.23	25.14	23.20
	40%	89.92	72.04	60.09	51.54	45.12	40.13	36.12	32.85	30.12	27.81	25.82
	50%	98.90	79.26	66.12	56.72	49.66	44.17	39.77	36.16	33.16	30.61	28.43

[a] AMS – average true metabolizable energy values with mean-subtraction methodology.

Table 4.8 SMS[a] estimated habitat required (thousands; ha) to support Louisiana Chenier Plain waterfowl energy demands from August–March.

% change in waterfowl energy demand	% change in raw seed biomass										
	-50%	-40%	-30%	-20%	-10%	0%	10%	20%	30%	40%	50%
-50%	26.33	21.06	17.56	15.05	13.17	11.71	10.53	9.58	8.78	8.10	7.53
-40%	32.30	25.85	21.54	18.47	16.16	14.37	12.93	11.76	10.78	9.95	9.24
-30%	38.53	30.85	25.72	22.05	19.30	17.16	15.44	14.04	12.87	11.88	11.03
-20%	45.05	36.07	30.08	25.79	22.58	20.07	18.07	16.43	15.06	13.90	12.91
-10%	51.88	41.55	34.65	29.72	26.01	23.13	20.82	18.93	17.36	16.02	14.88
0%	59.02	47.28	39.44	33.83	29.61	26.33	23.71	21.56	19.77	18.25	16.95
10%	66.51	53.30	44.46	38.14	33.39	29.70	26.74	24.31	22.29	20.58	19.12
20%	74.37	59.61	49.74	42.67	37.36	33.23	29.92	27.21	24.95	23.04	21.39
30%	82.61	66.23	55.28	47.43	41.53	36.94	33.26	30.25	27.74	25.61	23.79
40%	91.28	73.20	61.10	52.44	45.92	40.85	36.78	33.46	30.68	28.33	26.31
50%	100.39	80.54	67.24	57.71	50.54	44.96	40.49	36.83	33.77	31.19	28.97

[a] SMS – species-specific true metabolizable energy values with mean-subtraction methodology.

Table 4.9 APL[a] estimated habitat required (thousands; ha) to support Louisiana Chenier Plain waterfowl energy demands from August–March.

		% change in raw seed biomass										
		-50%	-40%	-30%	-20%	-10%	0%	10%	20%	30%	40%	50%
% change in waterfowl energy demand	-50%	24.73	20.03	16.81	14.48	12.71	11.33	10.22	9.30	8.54	7.89	7.33
	-40%	30.36	24.58	20.64	17.78	15.61	13.91	12.54	11.42	10.48	9.69	9.00
	-30%	36.24	29.35	24.65	21.23	18.64	16.61	14.98	13.64	12.52	11.57	10.75
	-20%	42.40	34.35	28.84	24.84	21.81	19.44	17.53	15.96	14.65	13.54	12.58
	-10%	48.86	39.58	33.23	28.63	25.14	22.40	20.20	18.40	16.89	15.61	14.50
	0%	55.63	45.06	37.84	32.60	28.63	25.51	23.01	20.95	19.23	17.77	16.52
	10%	62.73	50.82	42.68	36.77	32.29	28.78	25.96	23.64	21.70	20.05	18.63
	20%	70.18	56.87	47.76	41.15	36.14	32.21	29.05	26.45	24.28	22.44	20.86
	30%	78.02	63.22	53.10	45.75	40.18	35.82	32.30	29.42	27.01	24.96	23.20
	40%	86.27	69.91	58.72	50.60	44.44	39.61	35.73	32.54	29.87	27.61	25.66
	50%	94.95	76.96	64.65	55.70	48.93	43.62	39.34	35.83	32.89	30.40	28.25

[a] APL – average true metabolizable energy values with patch-level methodology.

Table 4.10 SPL[a] estimated habitat required (thousands; ha) to support Louisiana Chenier Plain waterfowl energy demands from August–March.

		% change in raw seed biomass										
		-50%	-40%	-30%	-20%	-10%	0%	10%	20%	30%	40%	50%
% change in waterfowl energy demand	-50%	25.17	20.39	17.12	14.75	12.95	11.54	10.41	9.48	8.70	8.04	7.47
	-40%	30.89	25.03	21.02	18.11	15.90	14.17	12.78	11.64	10.68	9.87	9.18
	-30%	36.88	29.88	25.10	21.62	18.99	16.92	15.26	13.90	12.76	11.79	10.96
	-20%	43.15	34.96	29.37	25.30	22.22	19.81	17.86	16.27	14.93	13.80	12.83
	-10%	49.72	40.29	33.84	29.16	25.61	22.83	20.59	18.75	17.21	15.91	14.78
	0%	56.60	45.87	38.53	33.20	29.16	25.99	23.45	21.35	19.60	18.12	16.84
	10%	63.83	51.73	43.46	37.45	32.89	29.32	26.45	24.09	22.11	20.43	18.99
	20%	71.42	57.89	48.63	41.91	36.81	32.82	29.60	26.96	24.75	22.87	21.26
	30%	79.39	64.36	54.07	46.59	40.93	36.49	32.92	29.98	27.52	25.44	23.64
	40%	87.78	71.17	59.79	51.53	45.27	40.36	36.41	33.16	30.44	28.13	26.15
	50%	96.62	78.34	65.82	56.73	49.84	44.43	40.08	36.51	33.52	30.98	28.80

[a] SPL – species-specific metabolizable energy values with patch-level methodology.

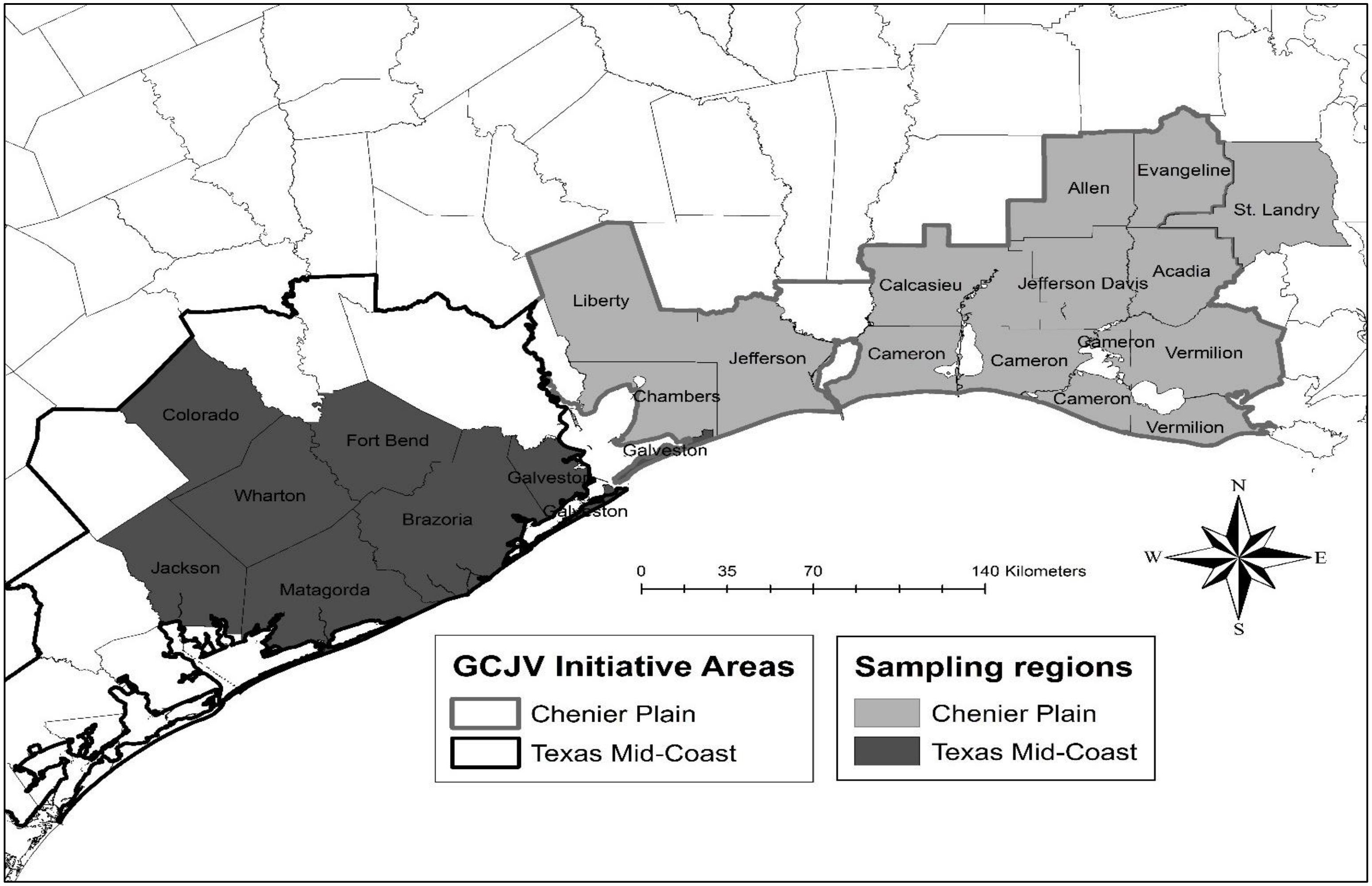

Figure 4.1 Gulf Coast Joint Venture Initiative Areas and sampling regions where soil cores were collected in August, October, and November, 2010–2013.

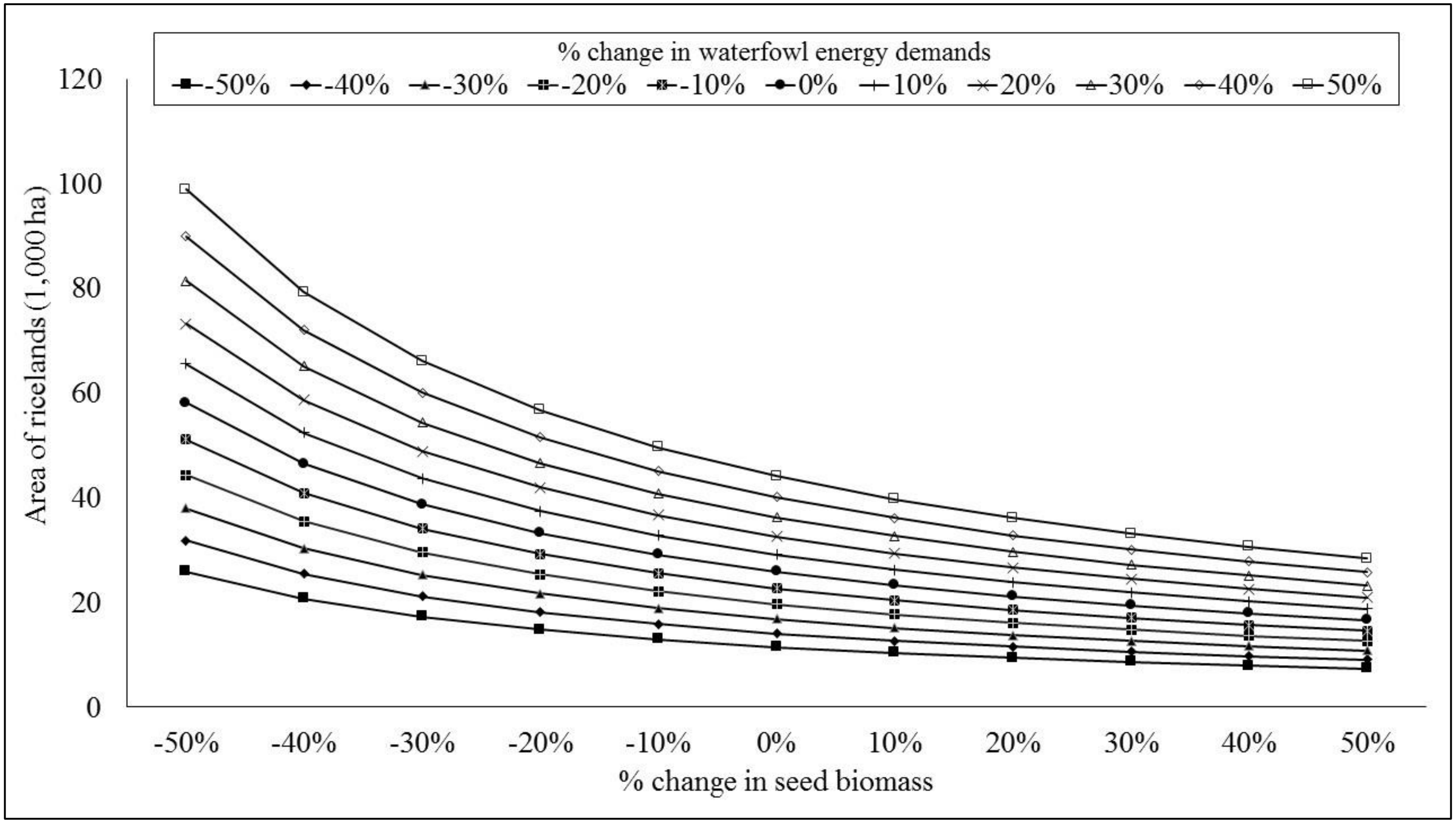

Figure 4.2 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using AMS[a] methods to calculate available metabolizable energy, when energy demand was held constant at ±10–50%, and seed biomass was varied ±10–50%.

[a] AMS – average true metabolizable energy values with mean-subtraction methodology.

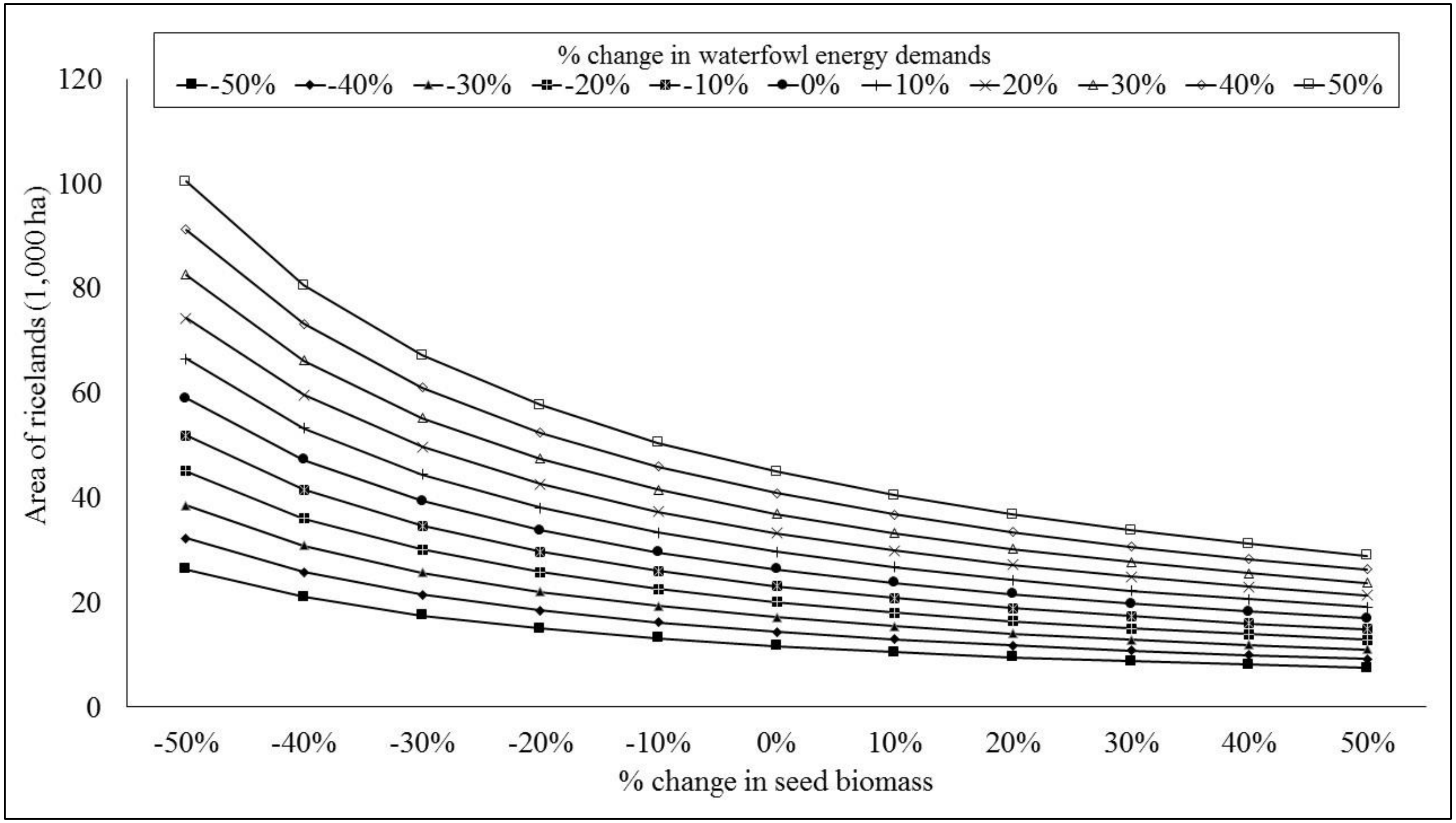

Figure 4.3 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain of Louisiana and Texas, using SMS[a] methods to calculate available metabolizable energy, energy demand was held constant at ±10–50%, and seed biomass was varied ±10–50%.

[a] SMS – species-specific true metabolizable energy values with mean-subtraction methodology.

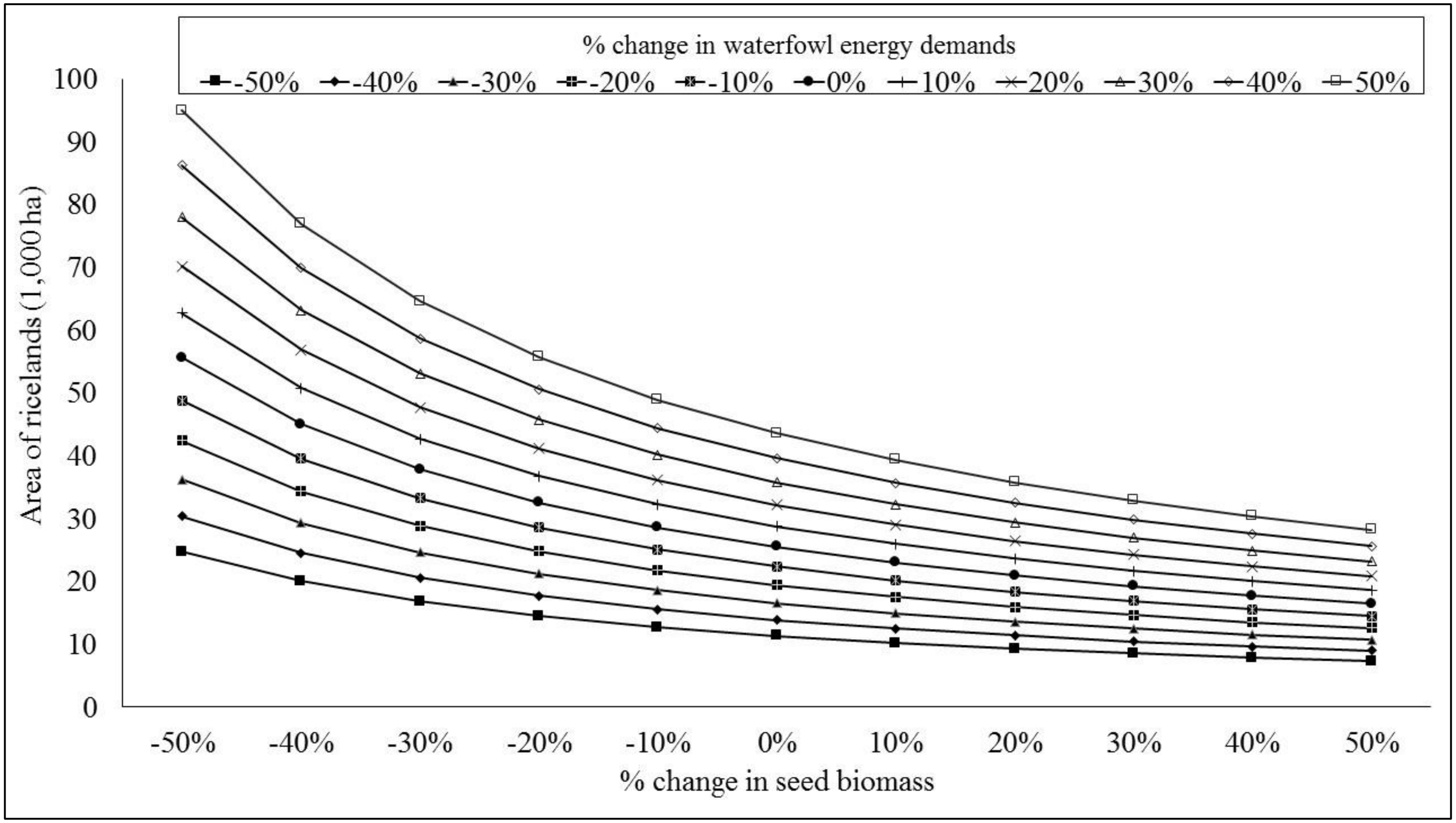

Figure 4.4 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using APL[a] methods to calculate available metabolizable energy, when energy demand was held constant at ±10–50%, and seed biomass was varied ±10–50%.

[a] APL – average true metabolizable energy values with patch-level methodology.

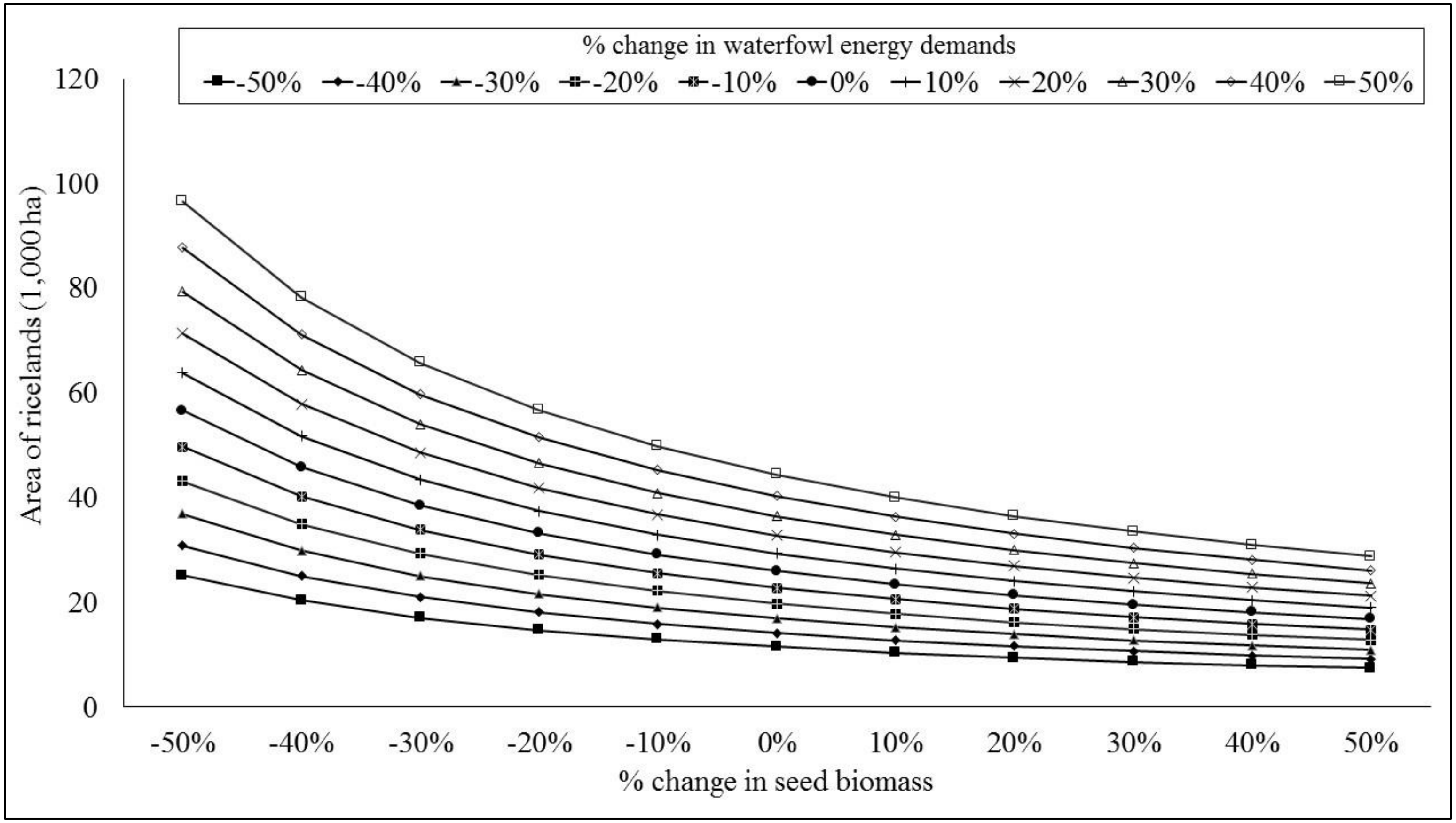

Figure 4.5 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using SPL[a] methods to calculate available metabolizable energy, when energy demand was held constant at ±10–50%, and seed biomass was varied ±10–50%.

[a] SPL – species-specific metabolizable energy values with patch-level methodology.

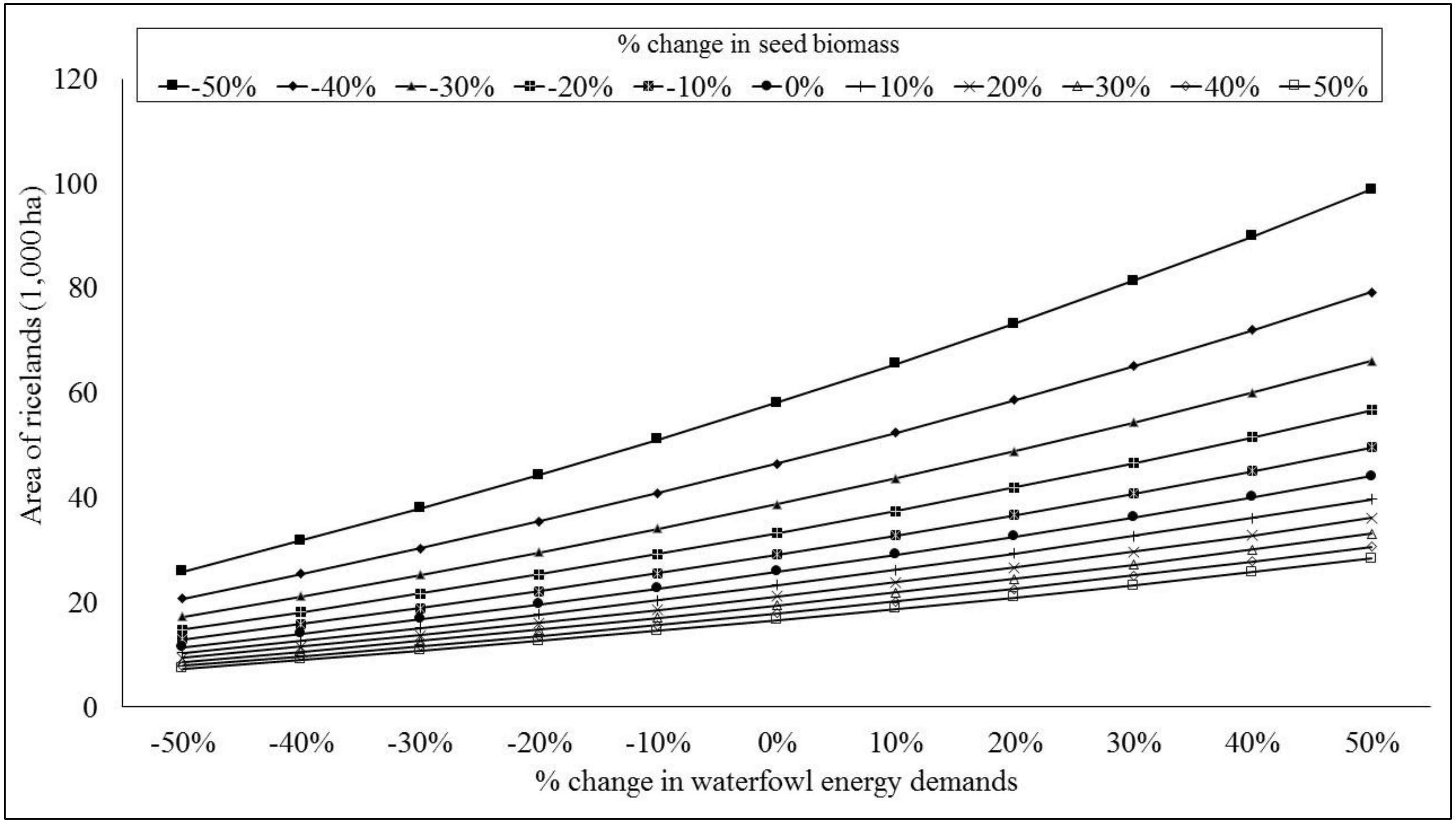

Figure 4.6 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using AMS[a] methods to calculate available metabolizable energy, when seed biomass was held constant at ±10–50%, and energy demand was varied ±10–50%.

[a] AMS – average true metabolizable energy values with mean-subtraction methodology.

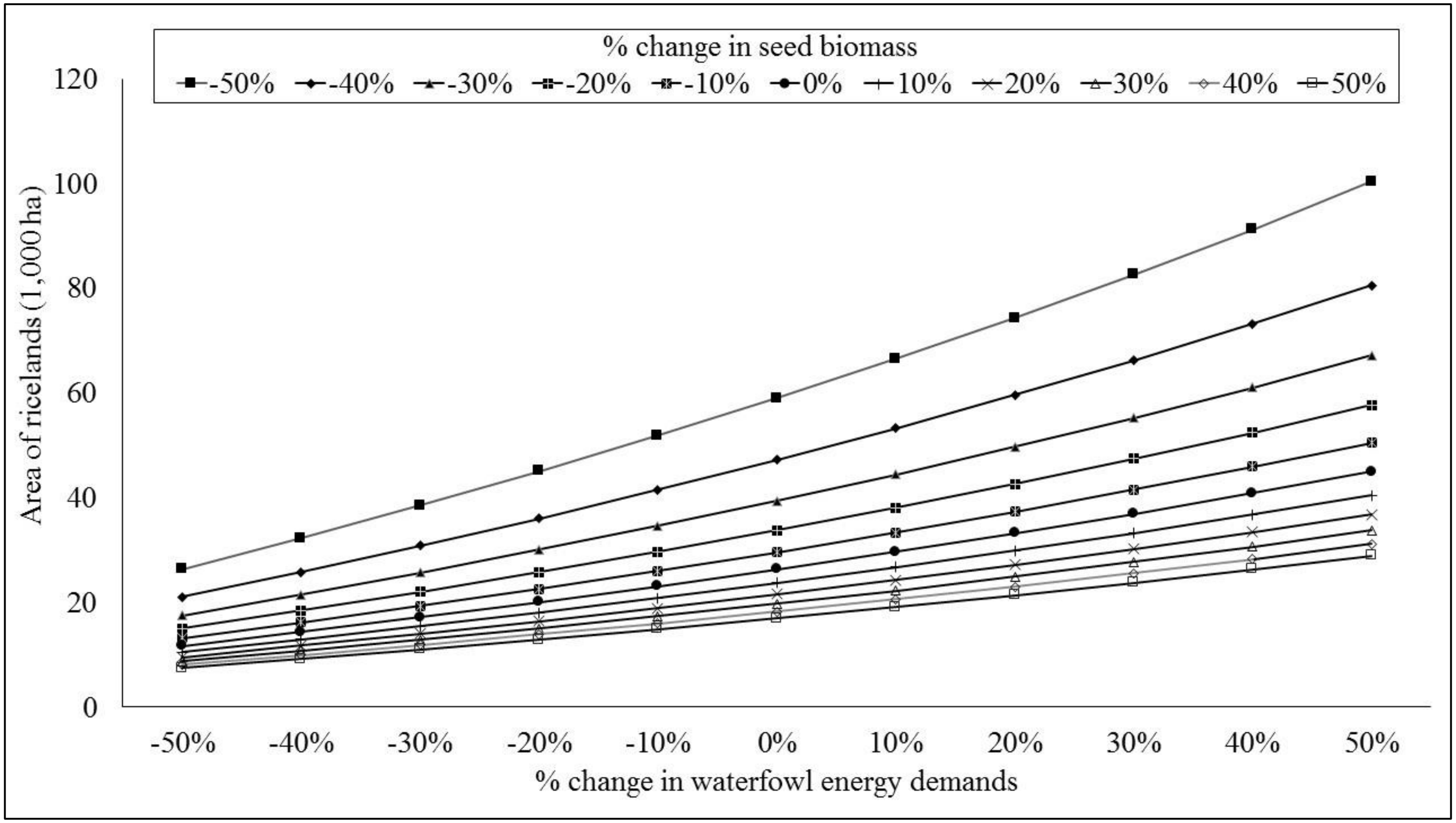

Figure 4.7 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using SMS[a] methods to calculate available metabolizable energy, when seed biomass was held constant at ±10–50%, and energy demand was varied ±10–50%.

[a] SMS – species-specific true metabolizable energy values with mean-subtraction methodology.

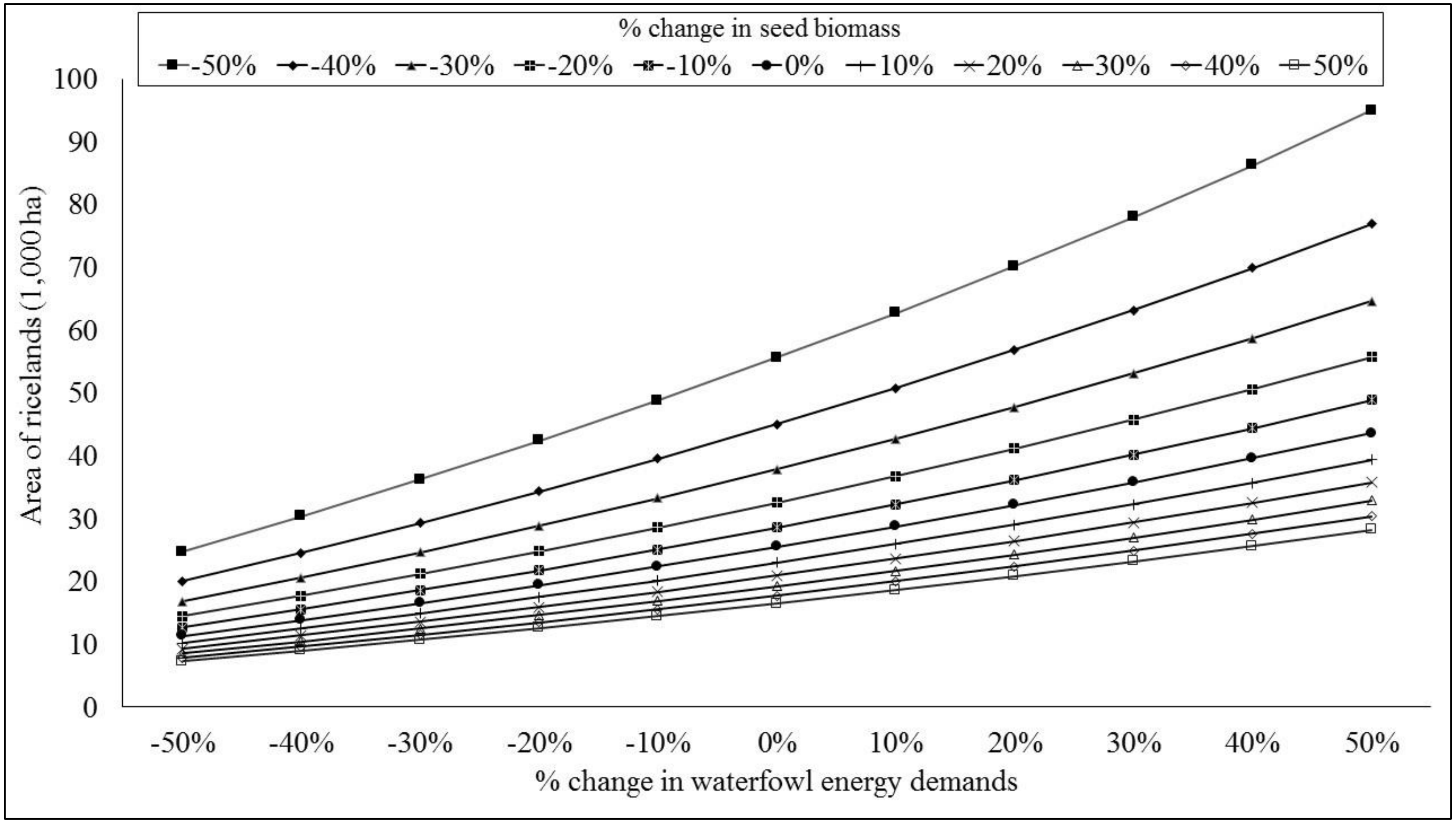

Figure 4.8 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using APL[a] methods to calculate available metabolizable energy, when seed biomass was held constant at ±10–50%, and energy demand was varied ±10–50%.

[a] APL – average true metabolizable energy values with patch-level methodology.

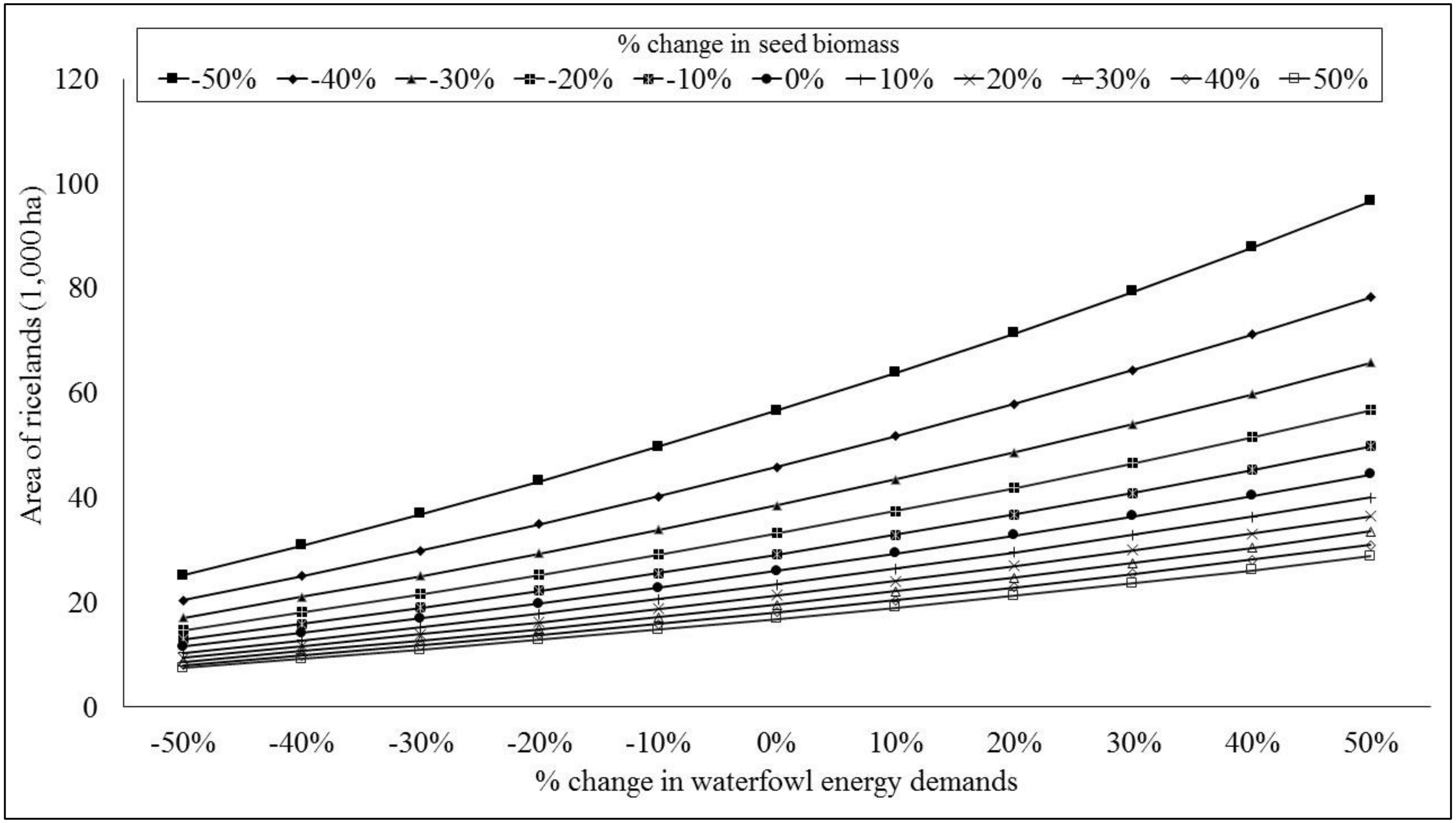

Figure 4.9 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, using SPL[a] methods to calculate available metabolizable energy, when seed biomass was held constant at ±10–50%, and energy demand was varied ±10–50%.

[a] SPL – species-specific metabolizable energy values with patch-level methodology.

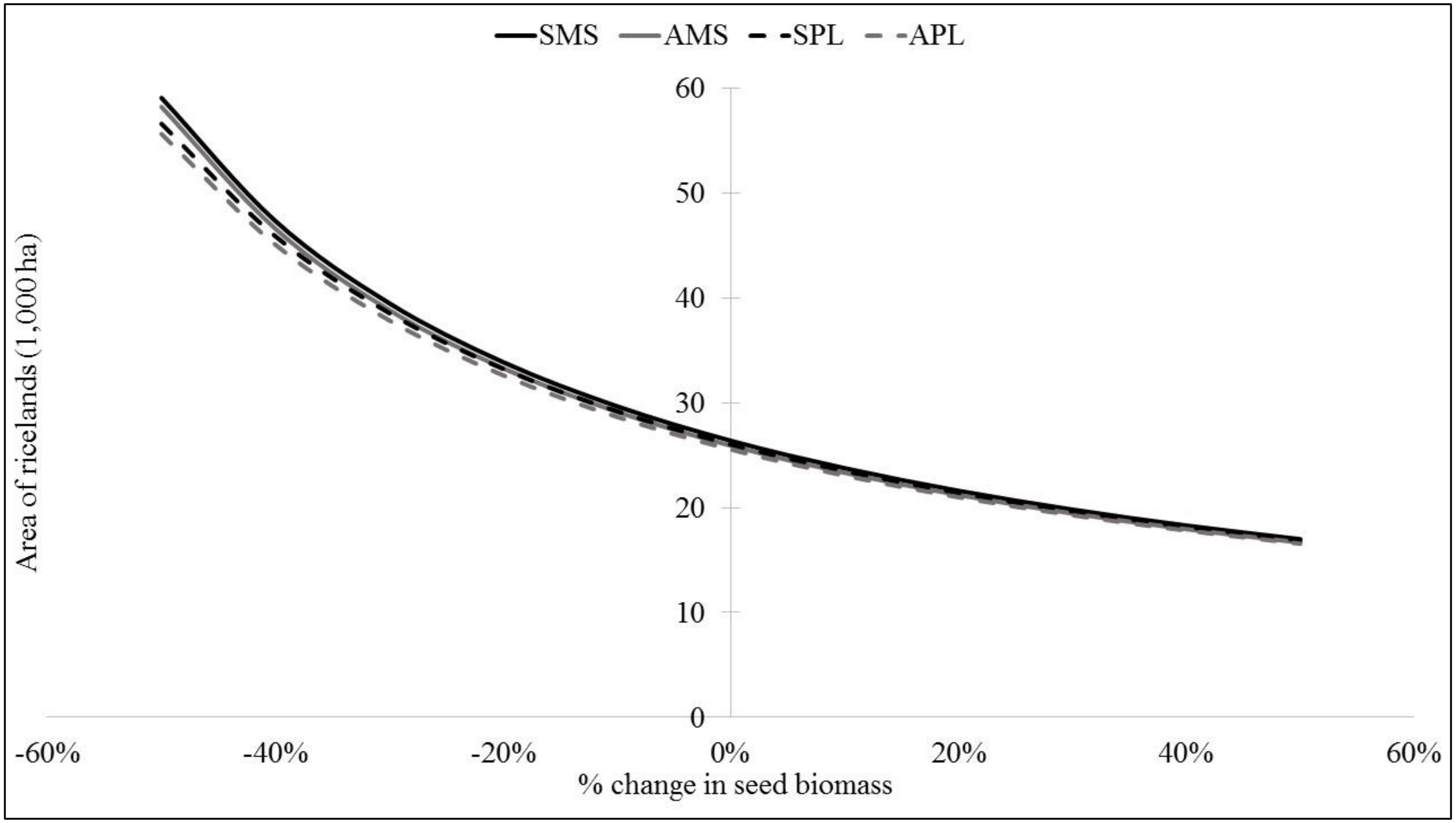

Figure 4.10 Area (ha) of ricelands required to support waterfowl energy demands in agricultural habitats from August–March in the Louisiana Chenier Plain, for each method of calculating available metabolizable energy, while holding population objectives constant at baseline levels, and varying seed biomass ±10–50%.

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APPENDIX A

ESTIMATES OF WASTE-RICE AND NATURAL SEED BIOMASS IN PRODUCTION, SEED-, AND IDLED RICE FIELDS IN THE LOUISIANA CHENIER PLAIN, TEXAS CHENIER PLAIN, AND TEXAS MID-COAST

Table A.1 Bias corrected estimates of mean waste-rice and natural seed biomass in production and idled rice fields in the Louisiana Chenier Plain, August–November, 2010–2013.

			Rice biomass			Natural seed biomass		
Sample period	Field classification[a,b]	*n* cores	$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
Aug	FH	1,215	259.14	31.5	0.12	138.03	19.8	0.14
Nov	SR	318	823.98	134.9	0.16	260.82	73.5	0.28
	HR	519	181.51	46.7	0.26	204.95	34.6	0.17
	NR	414	125.85	24.1	0.19	102.88	19.9	0.19
Aug	SI	606	17.21	14.3	0.83	182.79	24.1	0.13
Oct		149	0.03	0.0	1.11	265.46	79.3	0.30
Nov		408	9.73	4.4	0.45	287.90	58.1	0.20
Aug	DI	550	1.89	1.0	0.51	153.12	46.6	0.30
Oct		235	0.00	0.0	.	521.28	116.2	0.22
Nov		769	30.73	22.0	0.72	215.52	50.5	0.23

Sample periods, field classifications, *n* cores, and gross bias corrected estimates[c] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), standard errors (SE), and coefficients of variation (CV; %) for production and idled rice fields in the Louisiana Chenier Plain, August–November, 2010–2013.

[a] FH, first harvest; HR, harvested ratoon; SR, standing ratoon; NR, no ratoon; SI, standing idle; DI, disked idle.

[b] Blanks denote same field classification.

[c] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians.

Table A.2 Bias corrected estimates of mean waste-rice and natural seed biomass in production and idled rice fields in the Texas Chenier Plain, August–November, 2010–2013.

Sample period	Field classification[a,b]	*n* cores	Rice biomass			Natural seed biomass		
			$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
Aug	FH	180	249.66	64.1	0.26	118.40	35.4	0.30
Nov	HR	113	348.05	187.7	0.54	54.98	26.8	0.49
	NR	70	32.50	27.2	0.84	114.21	20.0	0.18
Aug	SI	78	1.55	1.2	0.76	253.36	91.8	0.36
Oct		18	0.00	0.0	.	250.46	98.7	0.39
Nov		80	0.58	0.1	0.14	238.13	31.1	0.13
Aug	DI	90	3.06	1.5	0.50	129.59	9.0	0.07
Oct		30	0.00	0.0	0.00	215.96	199.9	0.93
Nov		90	5.17	3.5	0.68	130.15	61.5	0.47

Sample periods, field classifications, *n* cores, and gross bias corrected estimates[a] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), standard errors (SE), and coefficients of variation (CV; %) for production and idled rice fields in the Texas Chenier Plain, August–November, 2010–2013.

[a] FH, first harvest; HR, harvested ratoon; SR, standing ratoon; NR, no ratoon; SI, standing idle; DI, disked idle.

[b] Blanks denote same field classification.

[c] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians.

Table A.3 Bias corrected estimates of mean waste-rice and natural seed biomass in production and idled rice fields in the Texas Mid-Coast, August–November, 2010–2013.

			Rice biomass			Natural seed biomass		
Sample period	Field classification[a,b]	*n* cores	$\bar{x}$	SE	CV	$\bar{x}$	SE	CV
Aug	FH	552	209.16	48.6	0.23	160.16	45.6	0.28
Nov	SR	50	1022.45	780.0	0.76	89.07	45.6	0.51
	HR	437	314.01	123.0	0.39	118.64	38.3	0.32
	NR	45	97.87	22.6	0.23	96.90	36.4	0.38
Aug	SI	332	4.85	2.9	0.60	209.75	65.9	0.31
Oct		112	2.09	2.1	1.00	291.74	50.2	0.17
Nov		268	5.64	5.3	0.93	406.47	60.7	0.15
Aug	DI	210	11.21	10.4	0.93	244.02	81.5	0.33
Oct		66	2.15	1.7	0.81	490.08	337.6	0.69
Nov		259	3.99	2.0	0.49	238.35	119.8	0.50

Sample periods, field classifications, *n* cores, and gross bias corrected estimates[a] of mean ($\bar{x}$) waste-rice and natural seed biomass (kg[dry]/ha), standard errors (SE), and coefficients of variation (CV; %) for production and idled rice fields in the Texas Mid-Coast, August–November, 2010–2013.

[a] FH, first harvest; HR, harvested ratoon; SR, standing ratoon; NR, no ratoon; SI, standing idle; DI, disked idle.

[b] Blanks denote same field classification.

[c] Estimates corrected for seed loss during sieving and non-detection or non-recovery of seeds by technicians.