

AMPHIBIANS AS WETLAND RESTORATION INDICATORS ON WETLANDS
RESERVE PROGRAM SITES IN LOWER GRAND RIVER BASIN, MISSOURI

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by

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The undersigned, appointed by the dean of the Graduate School, have examined the thesis entitled

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RESERVE PROGRAM SITES IN LOWER GRAND RIVER BASIN, MISSOURI

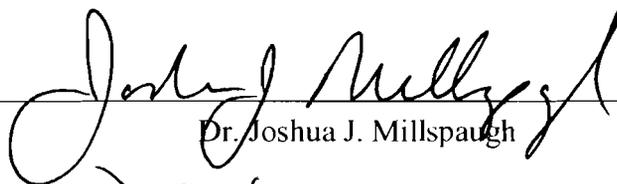
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ABSTRACT

Globally, amphibians have suffered dramatic population declines in the past twenty years with habitat destruction implicated as the primary threat. The Natural Resources Conservation Service's Wetlands Reserve Program (WRP) restores wetlands on marginal agricultural land and is a means to restore the spatio-temporal wetland habitat required by amphibians to prevent, reverse, or stabilize declining population trends. The goal of WRP is "to achieve the greatest wetland functions and values, along with optimum wildlife habitat, on every acre enrolled in the program." Functions and values are defined as the hydrological and biological characteristics of wetlands. A key unanswered question is to what extent is this goal being achieved? Amphibians enable quantifying the WRP goal due to their life-history requirements and explicit incorporation of their habitat needs into WRP plans. My research goal was to determine if hydrological and biological wetland characteristics had been restored to WRP sites in the Lower Grand River basin, north-central Missouri, based on distribution, recruitment success, and relative species richness estimates for members of a regional species pool. I identified three design strategies applied to WRP sites over time: walk-away, maximize hydrology, and naturalistic; the latter emphasizing restoring process as well as structure; and evaluated if design strategy was a useful covariate for restoration efforts. I encountered 10 amphibian species representing 59% of the regional species pool. Design strategy was not a predictive site-level covariate as sites within all three design strategies

had varying hydrological wetland conditions resulting in greater habitat heterogeneity than anticipated on maximize hydrology and walk-away sites and less than anticipated on naturalistic sites. Amphibian detections occurred across all sites resulting in no difference among design strategy as the degree of heterogeneity in habitat conditions at the within site-scale demonstrated that amphibians were responding to ecological conditions that occur at a finer resolution than site. Results, irrespective of design strategy, indicate seven of the detected species or groups were widely- distributed, two were moderately- distributed, and two were sparsely distributed on WRP sites indicating hydrological wetland characteristics have been restored to sites given the moderate- to wide-distribution of species associated with both seasonal and permanent wetlands. Although species were successfully recruiting young into adult populations, only leopard frogs had high estimates of recruitment success whereas the remaining species had moderately high to moderate to low recruitment estimates indicating biological wetland characteristics are somewhat lacking to lacking for these species. Results from the relative species richness assessment indicate that, whereas 74% of the sites provided some degree of wetland habitat for members of the regional species pool over the course of the field season (7 March – 19 September), 52% of the sites lacked suitable habitat conditions during the peak of amphibian breeding and larval development (May through July). Targeting management actions that result in suitable seasonal wetland habitat conditions (shallow, vegetated wetlands that gradually dry by mid-to late-summer) throughout the time needed for species to complete their life history requirements is one method to increase the biological wetland value of restored WRP sites. Results show the value of WRP at conserving and restoring river-floodplain amphibians; however, achieving optimum wildlife habitat on every enrolled acre will be difficult at a site-level

scale as habitat requirements, although overlapping, vary widely for the full range of species. Providing for all species in the regional species pool requires sites that transverse both the longitudinal and lateral floodplain gradient. If WRP is to realize its full potential, there must be recognition that optimum wildlife habitat can be defined at multiple spatial and temporal scales that match the landscape setting. Optimum wildlife habitat at a wetland scale is not the same as optimum wildlife habitat at the floodplain scale. The intent of WRP is to convert marginal, flood-prone agricultural lands back into wetlands so enrollment of lands located outside the active floodplain may be impracticable or unrealistic. Whereas attaining optimum wildlife habitat on every acre enrolled in the program may not be an achievable objective, providing optimum wildlife habitat for members of a regional species pool within an appropriately defined geography that includes both a longitudinal and lateral gradient represents an objective that is both desirable and attainable.

INTRODUCTION

Plight of Amphibians

Amphibians have received close attention during the past 20 years as reports of population declines began to surface around the world in the mid-1980s (Blaustein and Wake 1990, Wyman 1990). Reported declines and extinctions from protected locations such as Yosemite National Park, the Monteverde Cloud Forest Reserve in Costa Rica, and rainforests in Australia were of particular concern (Drost and Fellars 1996, Laurance et al. 1996, Pounds et al. 1997). Confusion reigned among the herpetological community as the first reports surfaced because very few long-term data sets existed and the boom-bust nature of amphibian population dynamics made it difficult to distinguish natural variation from real reductions (Wake 2003). Numerous monitoring projects were initiated as a result of this alarm (Wake 1998, Corn 2002) with habitat alteration and destruction, disease and pathogens, global climate change, invasive species, chemical contamination, and commercial trade all emerging as potential explanations for the declines (Semlitsch 2000, Collins and Storfer 2003, Bradford 2005).

Most amphibian biologists agree that habitat degradation continues to be the primary threat to amphibian populations (Wake 1991, Blaustein and Wake 1995, Semlitsch 2002); however, there is a growing body of evidence that the influence of human-induced environmental stressors such as pollution combined with “natural” biotic and abiotic factors such as competition, predation, and seasonal pool drying may be interacting to create a “threshold point” whereby amphibians are more susceptible to endemic diseases and other pathogens (Boone and Semlitsch 2001, Blaustein and Kiesecker 2002, Collins and Storfer 2003, Storfer 2003). Wetland restoration efforts that

address the biological issues associated with amphibian declines by replacing habitat elements on a landscape scale may buffer an existing population from additional stressors and prevent, reverse, or stabilize downward population trends (Semlitsch and Bodie 1998, Collins and Storfer 2003). The full suite of amphibian species exhibit life history events that exploit the gradient of wetland conditions ranging from ephemeral to permanent based on the animal's ability to survive pond drying and to coexist with predators, primarily fish and aquatic insects (Wellborn et al. 1996, Hecnar and M'Closkey 1997, Skelly et al. 1999). A key element in amphibian conservation, therefore, involves not only protecting existing habitat but restoring the density and spatial configuration of habitat across a hydrological gradient to support and maintain amphibian population dynamics (Semlitsch 2005).

Amphibians usually have both an aquatic larval and a terrestrial adult life stage, thus, their habitat requirement not only include wet areas that exhibit spatiotemporal variation for breeding activities and larval development, but also include terrestrial habitats for foraging, overwintering, and refugia (Wilbur 1980, Stebbins and Cohen 1995, Semlitsch 2000, Gibbons 2003). Terrestrial habitat is that portion of an area not covered by water, so it may include the moist edge around a wetland and also typically includes leaf litter, soil, small mammal or invertebrate burrows, and coarse woody debris (Stebbins and Cohen 1995, Semlitsch and Bodie 2003). Their biphasic life history dictates amphibian dependency on abundant wetlands interspersed among terrestrial habitat that collectively are configured to facilitate dispersal and recolonization of populations that may go extinct due to stochastic events (Semlitsch 2000, Trenham et al. 2003, Trenham and Shaffer 2005). These types of small, shallow freshwater wetlands, historically, have

been the most imperiled as they are the easiest to convert into other land uses such as agriculture or housing developments (Dahl 2000). Although, for the first time, recent trends indicate the rate of wetland acreage gained exceeded the rate of wetland acreage loss in the conterminous United States, wetland gains would not have been greater than wetland losses without a 12.6% increase in freshwater ponds (Dahl 2006). During this same timeframe (1998-2004) freshwater vegetated wetlands (i.e., emergent, forested, and scrub-scrub wetlands) declined by 4.3% (Dahl 2006). The increase in ponds was due primarily to golf course developments although creation of freshwater fishing ponds and ponds associated with aquaculture production and housing developments also contributed to the increase (Dahl 2006, 2007). These artificially created ponds are not equivalent replacement for vegetated wetlands (Dahl 2006). Freshwater emergent wetlands have declined by the greatest percent of all freshwater wetland types since the 1950s with approximately 21% of those remaining lost in the past 50 years (Dahl 2006).

Wetland Restoration through the Wetlands Reserve Program

The Wetlands Reserve Program (WRP), a U. S. Department of Agriculture (USDA) program established in the 1990 Farm Bill and re-authorized in the 2002 and 2008 Farm Bills, is a voluntary, incentive-based wetland restoration program intended to convert marginal, flood-prone agricultural lands back into wetlands (NRCS 2005). The goal of WRP is to protect, restore, and enhance the functions and values of wetland ecosystems (NRCS 2005). This is accomplished by providing habitat for migratory birds and wetland dependent wildlife, including threatened and endangered species; protecting and improving water quality; lessening water flows due to flooding; recharging ground water; protecting and enhancing open space and aesthetic quality; protecting native flora

and fauna contributing to the Nation's natural heritage; and contributing toward educational and scientific scholarship (NRCS 2005). Three enrollment options available through WRP include: 1) a permanent easement, 2) a 30-year easement, or 3) a restoration agreement. The first two options place a conservation easement on an accepted property resulting in a WRP easement area whereas the restoration agreement results in a cost-share agreement. A conservation easement transfers most property rights to the federal government to maximize wetland functions and values on the property in exchange for monetary benefit with the landowner retaining four basic rights: 1) right to sell the property and pay taxes, 2) right to private access, 3) right to quiet enjoyment and recreational use on the property, and 4) right to subsurface resources as long as no drilling occurs within the easement area (NRCS 2005). A restoration agreement does not place an easement on the property but instead is a cost-share agreement in which USDA pays up to 75% of the cost of the restoration activity to re-establish lost or degraded wetland habitat. The landowner, in return, agrees to protect the restored habitats for the life of the agreement, usually a minimum of 10 years (NRCS 2007). Land eligible for WRP includes agricultural land; adjacent lands that contribute significantly to wetland functions and values; previously restored wetlands that need long-term protection; upland areas needed to buffer the wetlands or to simplify the boundary; drained wooded wetlands; existing or restorable riparian habitat corridors that connect protected wetlands, and lands substantially altered by flooding. The land must be both restorable and suitable for providing wildlife benefits (NRCS 2007). WRP provides a means to restore wetlands across the landscape; as of fiscal year 2008, over 2.0 million acres have been enrolled nationwide (NRCS 2008).

Wetlands Reserve Program in Missouri

Missouri was one of nine states that participated in a WRP pilot program in 1992 (NRCS 2003). That first year, the Natural Resources Conservation Service (NRCS) established 19 landowner contracts to restore 1,696 wetland acres (NRCS 2003); as of September 2006, 787 contracts have been completed, or are pending, to restore 115,583 acres in Missouri (Frazier and Galat 2009). Missouri identified the greatest wetland restoration need along the Missouri River and its major tributaries and in the Mississippi Alluvial Valley (NRCS 1999). Reasons identified for the importance of restoring wetlands in north Missouri watersheds included flood attenuation, water quality improvement, and wetland habitat for migratory wildlife (NRCS 1999). The Grand River, a major tributary of the Missouri River located in north-central Missouri, is the largest watershed in Missouri, north of the Missouri River (Pitchford and Kerns 1994). The lower Grand River sub-basin, located south of Chillicothe, Missouri, was selected by NRCS as one of three WRP emphasis areas (Figure 1). Emphasis areas, since replaced by eco-regions, were selected based on three criteria (1) locations where historical presence of wetlands existed, (2) areas identified in the Missouri Department of Conservation Wetland Management Plan (MDC 1989) and referenced in the North American Waterfowl Management Plan (USDI and CWS 1986), and (3) areas of concentrated, present-day waterfowl use (Kevin Dacey, Missouri Department of Conservation, personal communication). Offered properties located within emphasis areas received preferential points in the WRP ranking process. Additionally, candidate properties located within 5 miles of state, federal or private wetland management areas scored higher than properties located farther away (Missouri Wetlands Reserve Ranking

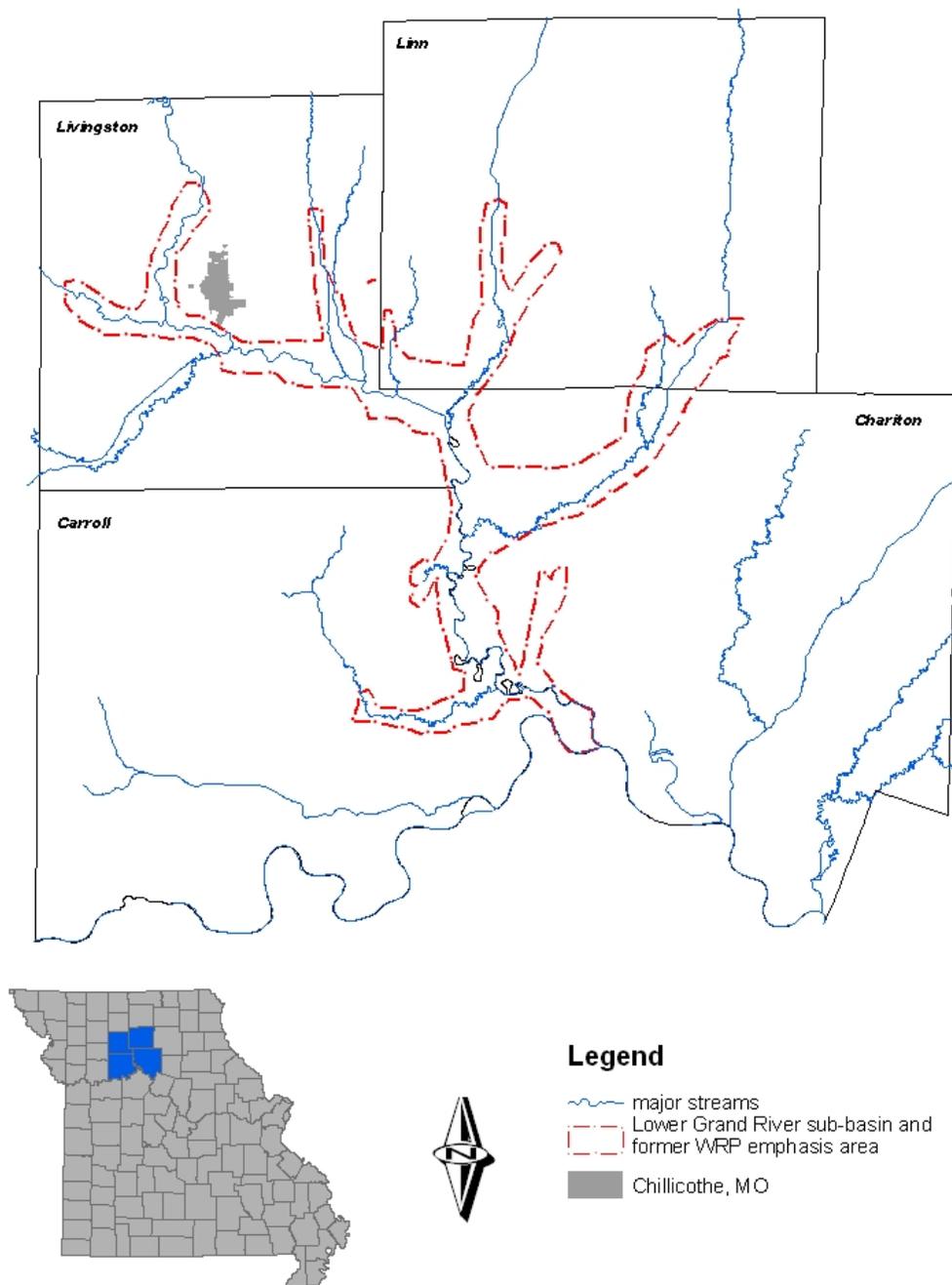


Figure 1. Location of Lower Grand River sub-basin which served as an emphasis area for the Wetlands Reserve Program in north-central Missouri.

System 2001, unpublished memo). The former lower Grand River emphasis area included the Missouri Department of Conservation's Fountain Grove Conservation Area and the U. S. Fish and Wildlife Service's Swan Lake National Wildlife Refuge, two intensively managed wetland areas. Waterfowl hunting has a long history within the Lower Grand River basin. The frequency of flood events and the strategic location of Fountain Grove and Swan Lake, two traditional stopping points for migratory waterfowl, made WRP an attractive option to landowners, particularly after the extreme flood years of the mid- to late-1990s (Galat et al. 1998).

Evolution of Riverine Wetland Restoration Strategies in Missouri

Practices commonly employed to convert wetlands for agricultural uses generally involved alterations to both hydrological and biological site characteristics. Methods used to alter hydrological characteristics included stream channelization to decrease the time required to drain water from adjacent floodplain fields, construction of flood-protection levees parallel to major streams to keep flood-waters off adjacent floodplain fields, and enhancing internal field drainage by leveling fields and constructing surface ditches and/or installing tiles, or subsurface permeable pipe, to remove excess water from poorly drained fields (Busman and Sands 2002). Alterations to biological site characteristics generally involved removing existing vegetation and, thus, reducing habitat diversity, to enable agricultural crop production. Wetland restoration, in the context of WRP, means rehabilitating degraded or lost habitats such that the original hydrology and vegetative community are, to the extent feasible, re-established (NRCS 1996). This is accomplished by first identifying the site as a wetland based on soil characteristics, cessation of farming activities, and restoring hydrological function by

reversing the agricultural practices designed to dry the site. The means by which these steps have been accomplished through WRP forms the basis for my story.

Wetland restoration efforts implemented through WRP over the past decade reflects an evolution in the thought-process of NRCS biologists and engineers as they learned and applied knowledge based on increased experience with riverine floodplains (D. Helmers, Natural Resources Conservation Service, personal communication). Early restoration efforts took a minimalist approach with projects generally referred to as “*walk-aways*.” Here emphasis was placed on restoring biological site characteristics through natural vegetative regeneration with little focus on hydrological restoration (Heard et al. 2005). The walk-away strategy reflected agency uncertainty with a new program and its potential appeal to landowners. Accepted properties tended to be relatively small (<30 ha) with small ditch plugs the only practice used to restore hydrology. Ditch plugs are small earthen berms constructed to block or slow down the flow of water in a ditch, thus, causing the water to back up the ditch and overflow into the field, creating small, shallow pools of water. These sites are generally dominated by early successional tree species.

By the mid-1990s, as the program matured and landowner interest increased, program focus shifted toward enhancing habitat for migratory birds by *maximizing hydrology* (Heard et al. 2005). This design strategy reflected the state-of-the-art knowledge at that time regarding wetlands and wetland management. Restoration efforts were focused on restoring the hydrological characteristics of a site by constructing low-profile perimeter levees (1-2 m tall) with narrow tops (3 m width) and 3:1 side slopes around each site, and installing a water control structure at the lowest end of the restored

pool. A pool is the resulting shallow, wetland impoundment constructed on a WRP property in which water levels can be manipulated due to placement of a water control structure. Borrow areas; i.e., areas from which soil was taken to construct levees; were typically located adjacent to the perimeter levee. This location, which resulted in relatively deep water areas along the periphery of the wetland pool, ultimately caused issues as borrows were difficult to re-flood if the pool was totally drained, taking a considerable amount of water from what was generally a scarce supply and leaving a limited amount to flood the remainder of the pool. Water elevation, or depth, within a pool is dictated by the topography of the pool and height of the water control structure; the maximum water depth is achieved when a structure is closed and the pool is flooded whereas the minimum water depth occurs when a structure is open and the pool is drained. Manipulating the extent and timing of when a structure is opened or closed enables one to change the water depth within a pool; generally, a structure is closed to increase water depth (flood-up event) and opened to decrease water depth (drawdown event). The design of maximize hydrology sites ensured the majority of the pool or pools could be flooded with at least 46 cm of water, the preferred foraging depth of most dabbling ducks (Krapu and Reinecke 1992). Maximize hydrology properties were designed to facilitate managed flooding and drawdowns with vegetative diversity dependent on water level manipulations and a premium placed on moist-soil vegetation management (Fredrickson and Taylor 1982). Relatively specific water management plans were provided to landowners; however, these plans were rarely followed due to complexity, logistics of accessing the property (many absentee landowners), and landowner desire to accomplish early drawdowns to facilitate food plot establishment to

attract wildlife, particularly waterfowl. Additionally, because the restored wetlands were located in the floodplain, even the low-profile levees created an impediment to water movement during flood events and the change in water heights as flood waters passed over the levees resulted in wide-spread scouring of levees and failures of water control structures. Damage to infrastructure occurs during a flood event when water depths are unequal on either side of a levee, creating either a difference in water pressure resulting in levee failure or a difference in water heights resulting in scouring. Damage can be minimized if water control structures are opened prior to the flood, allowing the water depths both inside and outside the wetland pool to rise at similar rates and depths on either side of the levees and equalizing the pressure gradient. However, damage then occurs as flood waters recede and inequality in water depths occurs because the flood waters outside the levee recedes faster than water levels within the pool because the amount of water within the pool exceeds the designed capacity of the water control structure. The difference in water heights results in levee scouring as the flood waters drop over the levee and scouring continues until the water level within the pool drops back to the designed pool elevation and, thus, the designed capacity of the water control structure. Animal burrowing was another source of infrastructure failure as the 3:1 slopes on levees proved attractive to muskrats. Maximize hydrology properties were typically dominated by herbaceous vegetation.

Program planning underwent another iteration as design emphasis shifted toward incorporating both the biological and hydrological characteristics of sites into the restoration scheme. Rather than relying on vegetative diversity as a by-product of hydrological restoration as occurred with the maximize hydrology design, this design

strategy enhanced micro- and macro-topographic features within the restored wetland pools, thus, creating varying water depths and habitats. Additionally, infrastructure modifications included low, broad levees constructed in serpentine patterns with wide tops (6 m width) and with side slopes ranging from 8:1 to 10:1 (NRCS 2002). This *naturalistic* design is a landscape approach that attempts to restore wetland function by emulating a more natural hydrologic regime through floodplain expansion and incorporating microtopography as matched appropriately to a given site (NRCS 2002). Naturalistic sites typically had water control structures designed with more spillway capacity than structures used on maximize hydrology properties, broad interior levees with wide tops, excavated wetlands constructed within wetland pools, and floodways, particularly if surrounded by flood-protection levees. *Excavated wetlands* are engineer-designed wetlands embedded within a pool and are created during construction with a tractor and scraper to restore micro-and macro-topography (Stratman 2000). The bottom elevation of an excavated wetland is generally lower than the bottom elevation of the water control structure so it cannot be drained when a drawdown is performed on the remainder of the pool. Sculpting excavated wetlands within wetland pools as was done with the naturalistic strategy not only provided soil needed for levee construction, thus, eliminating the need to locate borrows adjacent to perimeter levees but also increased the diversity of wetland habitats. Another characteristic of naturalistic sites was enhanced connectivity with streams by creating floodways on flood-protection levees. Flood-protection levees are usually adjacent to large streams, e.g., Grand River, and are built to exclude flood waters from agricultural fields; average height of flood-protection levees within the Lower Grand River basin is approximately 3-4 m. Floodways are

approximately 30 m wide sections of a flood-protection levee that are lowered to within 0.51 cm to 1.3 cm of the management pool elevation to allow inflow of water during a flood event and outflow of water as the flood waters recede. This scenario limits damage during flood events as water more quickly reaches equilibrium on either side of levees, thus limiting the scouring damage that occurred in maximize hydrology properties, and the amount of water flowing through water control structures stays within design capacity of the structure. Naturalistic designs are intended to be “flood-friendly” with broad, wide infrastructure that is less inclined to scour during flood events and that is not attractive to burrowing animals. Water management plans are general and simply provide guidance on rotating high or low water management regimes among pools. Pools are designed so it is impossible to totally drain a site and vegetative diversity both in plant species and structure results due to designing wetland pools that vary in depth and size. Naturalistic sites were generally designed to take advantage of landscape features and mimic remnant wetland scars resulting in properties that are primarily dominated by herbaceous, aquatic vegetation. These three design strategies represent an adaptive learning process; however, a key unanswered question is to what extent are program management objectives being met? (MacKenzie et al. 2006) (Table 1).

Amphibians as Indicators for Assessing Wetland Restoration Efforts

The goal of WRP is to “achieve the greatest wetland functions and values, along with optimum wildlife habitat, on every acre enrolled in the program” (NRCS 2005a). Functions and values are defined as the “hydrological and biological characteristics of wetlands and the socioeconomic value placed upon these characteristics” (NRCS 2005b). One of the functions and values of wetlands receiving primary emphasis by the WRP is

Table 1. Management objective for Wetlands Reserve Program and design strategy categories used to classify Wetlands Reserve Program properties in the Lower Grand River basin, north-central Missouri. Evolution of methods employed within each design strategy to restore both the biological and hydrological site characteristics to attain program objective are identified as well as the resulting benefits and issues associated with each design strategy.

Management object	Design strategy	Biological site practices	Hydrological site practices	Benefits	Issues
Restore wetland functions and values on former or degraded wetlands in the agricultural landscape	Walk-away	natural vegetation regeneration	minimal; ditch plugs	low cost	no hydrologic restoration; sites reverted to early successional tree species
	Maximize hydrology	intensive water level management plans intended to maximize vegetative diversity by promoting moist soil vegetation management	low level perimeter levees built with 3:1 side slopes and narrow tops (3 m); borrow taken from area adjacent to levee; water control structures; designed to maximize amount of pool covered by 46 cm of water depth	relative ease of flood-up and drawdown; increased wetland habitat	infrastructure failures due to scouring during flood events and animal damage; did not enhance connectivity with streams; early drawdowns to facilitate foodplot establishment compromised wetland and program objectives; complicated water level management plans
	Naturalistic	excavated wetlands created by borrows located in pool; less intensive water level management plans; incorporated floodways-permitted more passive management approach	low broad levees with side slopes ranging from 8:1 to 10:1 and wide tops (6 m) constructed in serpentine patterns; water control structures with increased spillway capacity	reduced infrastructure damage; increased wetland diversity; simplified water level management recommendations	sedimentation due to enhanced connectivity with streams; early efforts at excavated wetlands narrow, steep slopes, and deep

providing habitat for wetland-dependent wildlife including amphibians. A key uncertainty toward meeting this objective is associated with partial controllability; i.e., NRCS biologists and engineers do not have total management control to obtain a desired biological response (e.g., successful amphibian recruitment on WRP sites) from a given action (e.g., hydrologic restoration on WRP sites) because they do not have all the information required to exactly recreate a functioning wetland complex that varies both spatially and temporally (Humburg et al. 2006). Such questions can generally be informed by a reference condition; however, given the degree of alteration to wetlands and surrounding landscapes in Missouri, there is no remaining, intact historical condition with which to compare restoration efforts. An alternative method to gain insight into the intricacies of a dynamic wetland system is provided by the historical and current distribution of wetland-dependent wildlife as species assemblages can provide distinct information about system structure and function (Tockner et al. 1999).

Using communities or species assemblages as indicators of ecological conditions has been proposed by a number of investigators. Karr (1981) introduced the concept by suggesting fish communities could be used to assess the biotic integrity of rivers and streams. O'Connell et al. (1998) developed a similar biotic integrity index based on songbird community composition. Amphibians have been used as indicators of environmental degradation (Hammer et al. 2004), of habitat quality (Sheridan and Olsen 2003), and of ecosystem restoration success (Rice et al. 2006, Waddle 2006). Waddle (2006) determined amphibians are suitable ecosystem indicators because they are abundant and cost-effective to survey, are sensitive to stresses on the system and respond in a predictable manner, display responses to local changes that are anticipatory of

change to the whole system, integrate a response across the whole system, and are useful indicators of both short- and long-term changes. Amphibians can also serve as indicators of wetland permanency, or wetland hydroperiod, due to species-specific differences in the time required to complete larval development (Babbitt et al. 2003). Wetland hydroperiod is defined as the length of time and portion of year a wetland holds ponded water (Tarr and Babbitt 2010). Babbitt (2003) identified three categories of wetland hydroperiod that are applicable to my study: 1) *ephemeral* in which water is present < 4 months after ice out (dry by July 1); 2) *seasonal* in which water is present > 4 months after ice out but < 12 months (water present after July 1 but generally dry by late summer); and 3) *permanent* in which water is present > 12 months. Amphibians distribute themselves along these gradations and, although overlap in habitat use occurs among different species, most species are generally more strongly associated with one hydroperiod versus another. For example, bullfrogs may be found in seasonal wetlands but are more strongly associated with permanent wetlands as bullfrogs generally require more than one year to complete larval development. For my study, amphibians serve as indirect indicators of wetland restoration efforts as representation by members of the local amphibian assemblage associated with the different categories of wetland hydroperiod would indicate restoration of hydrological wetland characteristics. Evidence of successful recruitment by members of the local amphibian assemblage would indicate restoration of biological wetland characteristics. Therefore, representation by all members of the local amphibian assemblage on WRP sites combined with evidence of successful recruitment would imply ideal evidence of WRP having met the goal of “restoring hydrological and biological characteristics of wetlands.”

Design Strategy and Restoration of Wetland Characteristics

Amphibians use the entire wetland continuum by opportunistically exploiting the duration, magnitude, and frequency of wetland flooding and drying that varies spatially and temporally both within and among years (Pechmann et al. 1989, Semlitsch 2000). The three design strategies, i.e., walk-away, maximize hydrology, and naturalistic, applied to WRP properties in the Lower Grand River basin represent three wetland restoration models that attempt to mimic this spatiotemporal variability. Walk-aways represent the dry end of the hydrological scale; the hands-off approach taken toward hydrologic restoration assumes the system will re-establish itself with little to no input from management actions. Maximize hydrology reflects a “wetter-is-better” philosophy that relies on active management intervention to attain desired results and represents the wetter end of the hydrological scale. The naturalistic approach strikes a balance between the walk-away and maximize hydrology approaches as this design strategy attempts to restore both structure and process assuming a resilient system that will function in a more passive management scenario requiring only minor adjustments to correct system imbalance (e.g., actions required to control invasive species). The naturalistic strategy represents the intermediate portion of the hydrological scale; although ephemeral and permanent wetlands are likely represented on naturalistic sites; seasonal wetlands should be the dominant hydrological feature. Wetland-breeding amphibians distribute themselves across the hydrological gradient during the breeding and larval development portions of their life history (Babbitt 2003). Presence of members of a local amphibian assemblage associated with a specific wetland hydroperiod provides a means to compare the extent to which each design strategy is attaining program objectives related to

hydrological restoration. Presence of metamorphosed individuals provides the measure as to whether WRP restoration practices are successfully restoring biological wetland characteristics as metamorphs represent recruitment and recruitment represents species success at the population level (Semlitsch 2000). If WRP restoration efforts are to “prevent, reverse, or stabilize downward population trends,” then species must successfully recruit. Restored wetlands that attract breeding adults may have restored hydrological wetland characteristics but, if they do not provide suitable wetland conditions through larval metamorphosis, then they lack biological wetland characteristics and may function as a sink, resulting in a decline of local adult amphibian populations (Semlitsch 2000). Alternatively, restored wetlands that provide suitable breeding and larval development habitat for the suite of species expected to inhabit the Lower Grand River basin would indicate restoration of both hydrological and biological wetland characteristics. This should result in wetlands that represent a source and that contribute toward stable to increasing local adult amphibian populations (Semlitsch 2000).

The primary factor likely to influence presence of amphibians associated with each WRP wetland restoration model is timing and availability of suitable aquatic habitat (Pechmann et al. 1989, Babbitt 2003). If duration of water is too long (> 12 months), then a predator community, particularly predatory fish, develops resulting in reduced amphibian abundance and species diversity, whereas if duration of water is not long enough (< 4 months), wetlands dry up before larval amphibians metamorphose and amphibians are unable to successfully recruit (Pechmann et al. 1989, Wellborn et al. 1996, Hecnar and M’Closkey 1997, Babbitt 2003). As a result, amphibian species

richness is generally low on ephemeral wetlands due to risk from desiccation and on permanent wetlands due to predation risk. Seasonal wetlands (inundated > 4 months but < 12 months) that occur along the intermediate portion of the hydrological gradient usually have higher species richness estimates due to reduced risk from desiccation or predation (Wellborn et al. 1996, Babbitt 2003). Walk-away sites, due to lack of hydrological restoration, are assumed to result in dry to ephemeral wetlands that do not retain water of sufficient duration to ensure successful amphibian recruitment (Figure 2). This should result in restoration of neither hydrological nor biological wetland characteristics. Maximize hydrology sites, due to hydrological restoration efforts that attempted to ensure the majority of the pools were flooded to a depth of 46 cm, are assumed to result in seasonal to permanent wetlands. However, landowner interest in establishing food plots combined with the ability to more completely drain water off maximize hydrology sites, likely result in ephemeral wetlands that do not retain water of sufficient duration for successful amphibian recruitment (Figure 2). This should result in restoration of hydrological but not biological wetland characteristics. Naturalistic sites, due to incorporation of excavated wetlands that make it difficult to totally drain water from these sites, are assumed to result in seasonal to permanent wetlands somewhat similar to maximize hydrology sites. However, the inability to completely drain naturalistic sites is likely to ensure some water is present through the time required for amphibians to successfully recruit (Figure 2). This should result in restoration of both hydrological and biological wetland characteristics.

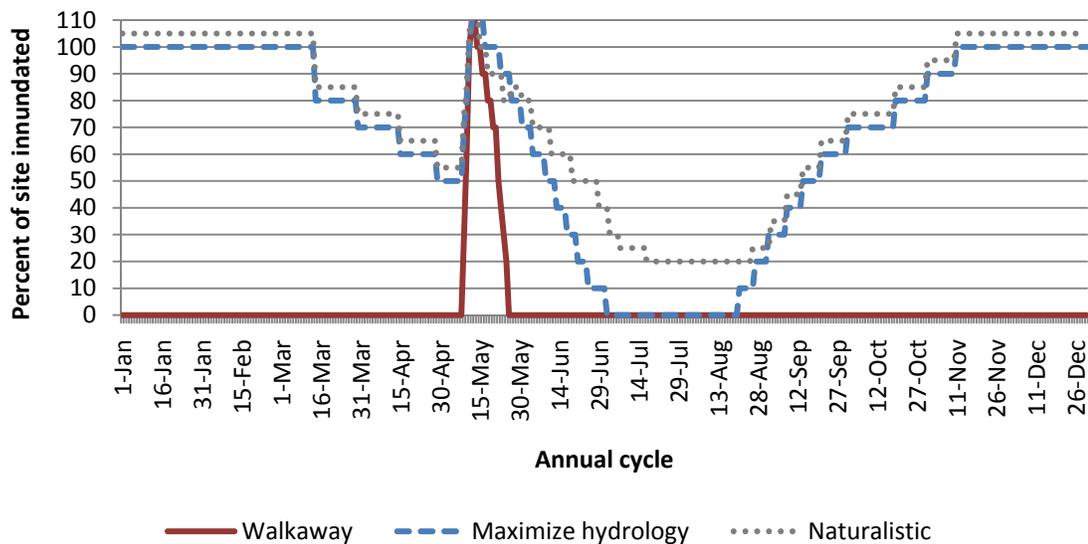


Figure 2. Conceptualized annual hydrological cycle on Wetlands Reserve Program sites classified by design strategy as either walk-away, maximize hydrology, or naturalistic for amphibian occupancy and species richness study conducted during 2007 in Lower Grand River basin, north-central Missouri. Walk-away sites were assumed dry except during flood events when duration of water was short term. Restoration efforts should have resulted in dry to ephemeral wetlands. Maximize hydrology sites were assumed flooded to full pool by late October, remained at full pool through fall, winter, and early spring, and were drawdown to minimum pool elevation by early summer and remained dry during the summer except during flood events. Restoration efforts should have resulted in seasonal to permanent wetlands but the early drawdowns were assumed to result in ephemeral wetlands that were not inundated of sufficient duration (< 4 months after ice off and dry before or by July 1) to provide amphibian recruitment habitat. Naturalistic sites were assumed flooded to full pool by late October, remained at full pool through fall, winter, and early spring and were drawdown to approximately 20% of the site area by early summer except during flood events. Restoration efforts should result in seasonal to permanent wetlands of sufficient duration (> 4 months after ice off and retain water after July 1 but < 12 months for seasonal and > 12 months for permanent) to provide amphibian recruitment habitat. Percent of site flooded 110 represents a flood event in which the entire floodplain is inundated. The lines representing maximize hydrology and naturalistic sites are off-set to prevent overlap from approximately 12 September through 30 April.

State Variable Selection for Assessing Wetland Restoration Efforts

State variables are one or more measurements of a population or community that characterize the system of interest (MacKenzie et al. 2004). State variables typically used in animal population sampling include population abundance, species richness, and occupancy. Population abundance, or number of individuals in a population, is often the state variable of interest when dealing with individual species; however, the time and effort required to attain sufficient sample size, i.e., the number of individuals observed or caught, at a landscape scale can be cost-prohibitive, particularly when dealing with rare or elusive species (MacKenzie et al. 2005). *Species richness*, or the number of species within a predefined area, is a community-level variable based on presence/absence data and conveys information on community structure and biodiversity (Boulinier et al. 1998). Species richness as a state variable can be problematic when dealing with rare species as the difficulty in attaining a suitable sample size hinders one's ability to make unbiased and accurate estimates (MacKenzie et al. 2005). *Occupancy*, defined as the proportion of area, patches, or sample units occupied by a single species (MacKenzie et al. 2005, 2006), is another useful state variable for population and community studies. Occupancy estimation also relies on presence/absence data, but exploits repeat surveys to attain unbiased estimates rather than recapture histories as are typically used to estimate abundance. Studies designed to estimate occupancy generally require less effort than studies designed to estimate abundance and, in the case of rare species, occupancy estimation can still be accomplished even when it is almost impossible to estimate abundance (MacKenzie et al. 2005). For these reasons, occupancy estimation is sometimes viewed as a surrogate for abundance (MacKenzie et al. 2006); however,

occupancy is the natural state variable for studies in which the primary focus is a species' distribution and range or in which the focus is metapopulation ecology (MacKenzie et al. 2006). Occupancy applied at the community-level may be viewed as a combination of multiple single-species studies whereby the same group of species is sampled at the same group of sites, thus, enabling inference to be made to the larger collection of all possible sampling sites (MacKenzie et al. 2005). It may also be viewed in a similar fashion to species richness if the quantity of interest is the fraction of members of an identified species list that are present in an area of interest (MacKenzie et al. 2005, 2006).

All three estimation approaches; abundance, species richness, and occupancy; must account for the variation introduced by detectability to provide accurate estimates (Williams et al. 2002, MacKenzie et al. 2005). Detectability is defined as the probability of detecting at least one individual of a given species in a particular sampling effort, given that individuals of that species are present in the area of interest during the sampling effort (Boulinier et al. 1998). Detectability has long been a recognized issue associated with count data (Fisher et al. 1943, Preston 1948, Nichols & Conroy 1996). It is very rare for all species or individuals to be counted when an area is sampled; therefore, estimates based on raw count data are generally negatively biased and lead to biased estimates if comparative assessments are made between two areas (Nichols et al. 1998).

Occupancy as a State Variable

Occupancy modeling is a probability-based approach that allows one to estimate occupancy rates when the probability of detecting a species is less than perfect ($p < 1$) (MacKenzie et al. 2002, 2006). MacKenzie et al. (2002) developed a single-season,

single-species occupancy model that allows estimation of proportion of area occupied as a function of measured variables while accounting for imperfect detection. Estimating the occupancy, or the proportion of an area of interest where the species is present during sampling, allows a community-level approach when comparing the occupancy rates of species included on a regional species list and how those rates are influenced by measured variables such as site size or site habitat characteristics. Occupancy modeling estimates the proportion of the landscape occupied by the species of interest while allowing one to incorporate measured variables enabling comparison of multiple sites (MacKenzie et al 2002, 2006). Occupancy is often the state variable of choice when attempting to determine range and distribution of a target species (MacKenzie et al 2006).

Species Richness as a State Variable

Assessing restoration efforts requires placing the efforts in the context of what is possible with what is realized (Palmer et al. 2005). As an example, *species richness* is a community-level state variable often used to assess community completeness; i.e., fewer species at a site is interpreted as community impoverishment (Cam et al. 2000). However, unless the number of species is placed in some context with the number of species possible at that site, fewer species may simply be a reflection of a smaller species pool (Cam et al. 2000). *Relative species richness* is defined as the ratio between observed richness at a site and the number of species potentially available for that site; i.e., the regional species pool (Cam et al. 2000). Identifying a regional species pool places the richness index within a realistic context of species that may occur on a site and should be greater at sites with favorable ecological conditions (i.e., high quality habitat resulting in recruitment) than at sites without favorable conditions (i.e., poor quality

habitat resulting in no recruitment) (Cam et al. 2000, Van Horne 2002, Dorazio and Royle 2005, MacKenzie et al. 2006).

Occupancy Method

Occupancy modeling provides a means to directly estimate the relative species richness parameter of Cam et al. (2000) if a regional species pool has been compiled for a site under investigation as an analogy can be drawn between estimating the proportion of sites occupied by a single species and estimating the proportion of species on a list that occupy a single site (MacKenzie et al. 2006). Using this approach, each species is considered a “site” and the proportional occupancy of a site, i.e., the probability that a member of the regional species pool is present at a site, is the Cam et al. (2000) relative species richness parameter. Occupancy modeling enables one to estimate relative species richness on one site at a time with comparisons possible through the use of means and other summary statistics (MacKenzie et al. 2005).

Capture/Recapture Method

Another method to estimate species richness is through the use of models developed for closed population, capture/recapture purposes (Nichols and Conroy 1976, Burnham and Overton 1979, Boulinier et al. 1998). These models have explicitly incorporated detectability by accounting for the fact that capture probabilities are not equal for all animals in a population being trapped (Nichols and Conroy 1976, Nichols et al. 1998, Boulinier et al. 1998). This species richness method estimates the number of species not detected at a site, based on the number of species detected at least once and places no limit on the number of species that may inhabit a community (Burnham and Overton 1979, MacKenzie et al. 2006). Again, use of a regional species pool enables

estimation of the Cam et al. (2000) relative species richness index by calculating the proportion of the regional species pool represented by the species richness estimate.

Research Goal and Objectives

My research goal is to evaluate if wetland restoration efforts on WRP properties within the Lower Grand River basin are restoring the hydrological and biological characteristics of wetlands as indicated by amphibian distribution, or proportion of sites occupied, amphibian recruitment success, and a relative species richness index.

Restoration of hydrological characteristics of wetlands will be indicated by the proportion of sites occupied by amphibians in the regional species pool. Estimation of the proportion of area, or sites, occupied by each member of the regional species pool will provide the means to determine distribution of amphibians on WRP sites in the Lower Grand River basin. The distribution of amphibians will serve as an indirect means to determine if hydrological wetland characteristics as represented by species associated with different wetland hydroperiods have been restored on maximize hydrology, naturalistic, and walk-away WRP sites in the Lower Grand River basin. Amphibian species exhibit strong seasonal movements based on breeding chronology and may exhibit varying degrees of vulnerability to sampling methods. These potential sources of heterogeneity in detection probabilities will be accounted for by including them as covariates in the detection analysis. Restoration of biological wetland characteristics will be indicated by recruitment success of detected amphibians. Multi-state occupancy probability estimation of each member of the regional species pool will provide the means to determine if amphibians are successfully recruiting young into the adult population.

The use of an identified regional species pool throughout this study enables an overall assessment of wetland restoration efforts, an assessment by design strategy, and an assessment of each individual sampled WRP site based on the relative species richness estimate. The definition of *relative species richness* for this project follows that of Cam et al. (2000) and is the ratio of estimated species richness to the number of species in the regional species pool. Therefore, a relative species richness estimate that represents a low proportion of the regional species pool will result in a poor assessment for wetland restoration efforts and a relative species richness estimate that represents a high proportion of the regional species pool will result in an excellent assessment for wetland restoration efforts. This rating metric reflects that the more amphibian species on a site, the higher the probability that species representing multiple hydroperiods are present indicating a hydrological gradient has been restored to that site. Also, the higher the species richness estimate, the higher probability that successful recruitment occurred, thus, indicating restoration of biological wetland characteristics. Ideally, restoration efforts should result in wetland conditions that span a hydrological gradient ranging from ephemeral to permanent and that provide suitable hydrological and biological wetland conditions enabling successful amphibian recruitment. My objectives are:

Objective 1. Evaluate if hydrological wetland characteristics have been restored on WRP properties in the Lower Grand River basin as indicated by amphibian distribution based on proportion of area occupied estimates on walk-away, maximize hydrology, and naturalistic sites.

Objective 2. Evaluate if amphibian detection probability varied by seasonal response to environmental conditions or sampling method.

Objective 3. Evaluate if biological wetland characteristics have been restored on WRP properties in the Lower Grand River basin as indicated by amphibian recruitment based on estimation of multi-state occupancy probability.

Objective 4. Assess wetland restoration efforts as indicated by the relative species richness metric achieved on WRP properties in the Lower Grand River basin.

Objective 5. Assess whether design strategy serves as a criterion of a functional ecological attribute created by management actions on WRP properties in the Lower Grand River basin through use of a cluster analysis.

STUDY AREA

The study area was located in north-central Missouri within the lower Grand River sub-basin below Chillicothe, Missouri in Livingston, Linn, Carroll, and Chariton counties. Agriculture is the dominant landuse within the study area which falls entirely within the Central Dissected Till Plains section in Missouri (Nigh and Schroeder 2002). This location was targeted because of its former status as a WRP emphasis area which provided a previously described geographic boundary that served as a starting point for narrowing down a study area and because of the concentration of WRP properties within the described boundary of the former emphasis area (approximately 5,000 ha [12,300 ac] of WRP properties located within the approximately 71,000 ha [175,000 ac] Lower Grand River emphasis area). The study area for this project was defined as a *modified* Lower Grand River emphasis area, hereafter referred to as the ***Lower Grand River basin*** (Figure 3). The original Lower Grand River emphasis area was modified to include that portion of the Lower Grand River floodplain protected by the Garden of Eden levee. The

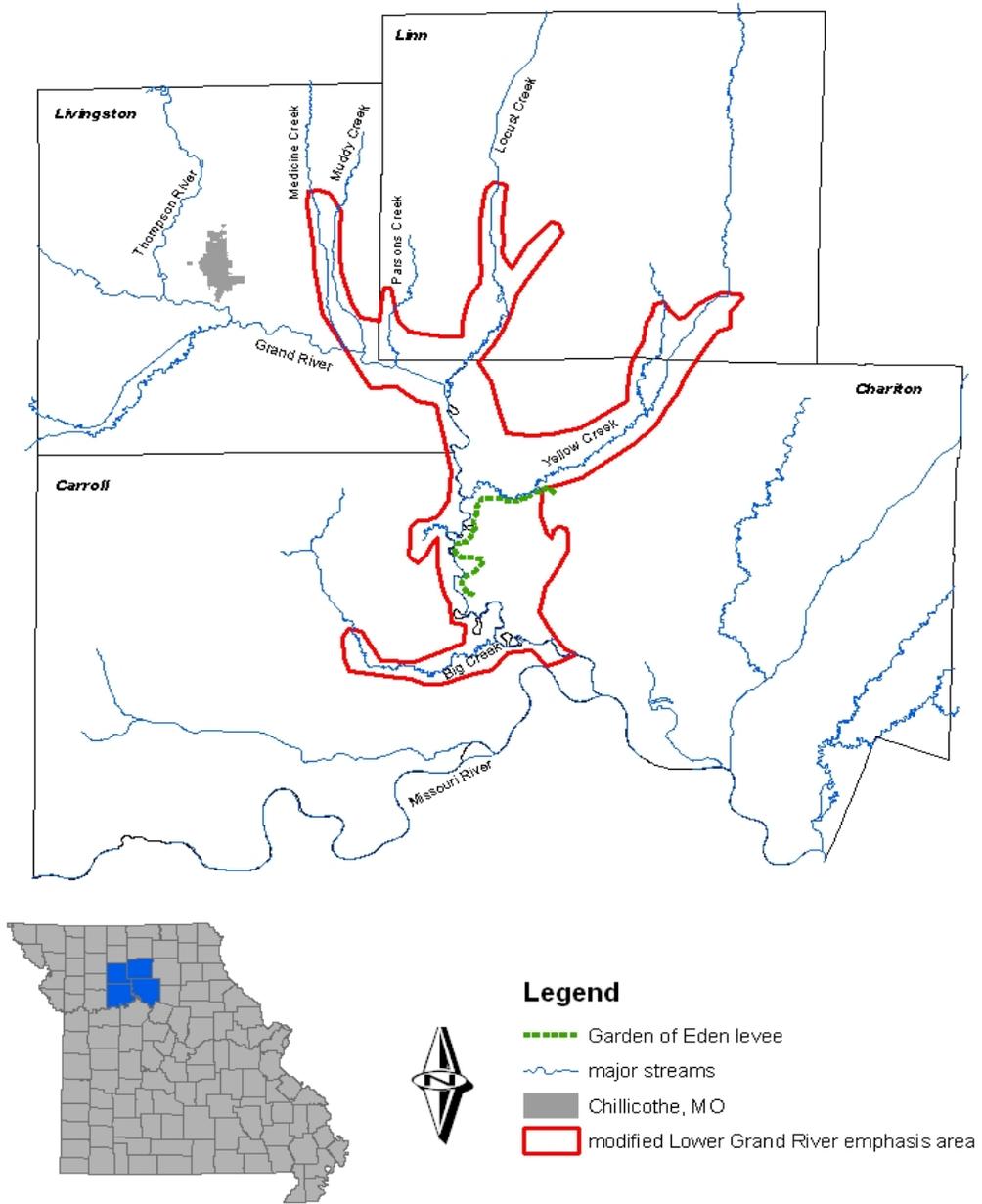


Figure 3. Location of Lower Grand River basin, defined as a modified Lower Grand River emphasis area, that served as study area for amphibian occupancy and species richness project conducted during the summer of 2007 in north-central Missouri.

Garden of Eden levee is a significant agricultural levee providing flood protection to a portion of the Lower Grand River floodplain located in Chariton County. Additionally, the Thompson River and the Lower Grand River west of Medicine Creek were excluded due to logistical constraints and the extreme southern portion of the original Lower Grand River emphasis area was excluded due to Missouri River's influence. Approximately 135 WRP properties were located within the Lower Grand River basin during my field season; of these, 10 had been accepted into the program but were not yet developed and 17 were scheduled for enhancement activities so these 27 properties were not included in the pool of potential sample sites. The number of WRP properties represented by each design strategy of the remaining 108 properties included 22 walk-away, 51 maximize hydrology, and 35 naturalistic sites. A few exceptions occurred within each group of design strategies that varied from the "typical model;" however, they retained the primary characteristics that defined each strategy (Table 1). Four of the sampled walk-away properties had previously developed wetlands including one 80 acre property with a developed wetland on the northern 35 acres and willows (*Salix nigra*) and cottonwoods (*Populus deltoids*) interspersed with ponds on the southern 45 acres. Exceptions among the maximize hydrology included 2 properties dominated by pin oaks (*Quercus palustris*) but with water control structures and 1 former fish farm/tree farm with ponds, planted trees, and an upland component dominated by food plots. Three properties within the naturalistic group had previously developed pools that were enhanced with the addition of excavated wetlands.

METHODS

Identification of a *regional species pool*, defined as the wetland-breeding amphibian assemblage for the Lower Grand River basin, was the first step required to accomplish this study and was applicable to all objectives (Table 2). Members of the regional species pool were determined by consulting amphibian distributional maps for Missouri (Johnson 2000, Lannoo 2005) and consulting with two amphibian experts (Dr. Jeff Briggler, Missouri Department of Conservation, Herpetologist and Dr. Raymond Semlitsch, University of Missouri, Curators Professor of Biological Sciences). Additionally, with input from Briggler and Semlitsch, I classified a predicted abundance and potential likelihood of detecting each member of the regional species pool during the 2007 field season so that I would have an a priori supposition with which to compare results. Definition of abundance and likelihood terms include: *abundant and likely*: very numerous with widespread habitats, certain to be detected if present; *common and likely*: numerous, habitats fairly wide-spread, expected to be detected if present; *infrequent but likely*: less numerous with more restricted habitats or range distribution but expected to be detected if present; *rare but likely*: few numbers with restricted habitat requirements but expected to be detected if present; *rare and unlikely*: few numbers with restricted habitat requirements and/or includes portion of Lower Grand River basin as part of its range; detection is possible but not probable (Table 2). Each member of the regional species pool was also classified according to its association with ephemeral, seasonal, or permanent hydroperiods (Table 2).

Table 2. Amphibian species list representing regional species pool for Lower Grand River basin compiled for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. Also included is category of wetland condition associated with each species and a predicted abundance and likelihood of species detection for each species in the regional species pool (Johnson 2000, Lannoo 2005, J. Briggler, Missouri Department of Conservation and R. Semlitsch, University of Missouri, personal communication). The predicted abundance and likelihood of detection^a for each species is indicated by a 1 in the appropriate column.

Scientific name	Common name	Wetland condition association	Estimated abundance and likelihood of detection				
			Abundant and likely	Common and likely	Infrequent but likely	Rare but likely	Rare and unlikely
<i>Notophthalmus viridescens</i>	Central newt	permanent	0	0	0	0	1
<i>Ambystoma texanum</i>	Small-mouthed salamander	seasonal	0	1	0	0	0
<i>Ambystoma tigrinum</i>	Eastern tiger salamander	seasonal	0	0	1	0	0
<i>Anaxyrus americanus</i>	American toad	seasonal	0	1	0	0	0
<i>Anaxyrus cognatus</i>	Great Plains toad	seasonal	0	0	0	0	1
<i>Anaxyrus woodhousii</i>	Woodhouse's toad	seasonal	0	0	0	0	1
<i>Acris crepitans</i>	Northern cricket frog	seasonal to permanent	1	0	0	0	0
<i>Hyla chrysoscelis- Hyla versicolor complex</i>	Grey treefrogs, Eastern and Cope's	seasonal	0	1	0	0	0
<i>Pseudacris crucifer</i>	Spring peeper	seasonal	0	0	1	0	0
<i>Pseudacris maculata</i>	Boreal chorus frog	seasonal	0	1	0	0	0
<i>Gastrophryne olivacea</i>	Western narrow-mouthed toad	seasonal	0	0	0	0	1
<i>Lithobates areolatus</i>	Crawfish frog	seasonal	0	0	0	1	0
<i>Lithobates blairi</i>	Plains leopard frog	seasonal to permanent	1	0	0	0	0
<i>Lithobates catesbeianus</i>	American bullfrog	permanent	1	0	0	0	0
<i>Lithobates clamitans</i>	Green frog	permanent	0	0	1	0	0
<i>Lithobates sphenocephalus</i>	Southern leopard frog	seasonal to permanent	0	1	0	0	0
<i>Lithobates sylvaticus</i>	Wood frog	ephemeral to seasonal	0	0	0	1	0

^a Definition of abundance and likelihood terms include: abundant and likely: very numerous with widespread habitats, certain to be detected if present; common and likely: numerous, habitats fairly wide-spread, expected to be detected if present; infrequent but likely: less numerous with more restricted habitats or range distribution but expected to be detected if present; rare but likely: few numbers with restricted habitat requirements but expected to be detected if present; rare and unlikely: few numbers with restricted habitat requirements and/or includes portion of Lower Grand River basin as part of its range; detection is possible but not probable.

Study Design

Occupancy modeling provided a means to determine amphibian distribution based on proportion of area occupied estimates, to determine detection probability estimates, and to determine amphibian recruitment success based on multi-state occupancy probability estimates for each member of the regional species pool (Objectives 1, 2, and 3). An occupancy study design was devised by following guidelines provided by MacKenzie and Royle (2005). Key aspects of designing an occupancy study include defining a site, determining an appropriate balance between number of sites and number of repeat surveys to conduct, selecting sites, defining a season, and identifying site characteristics that influence the probability of a site being occupied. This study design was also appropriate for calculating species richness estimates which were used to develop the relative species richness metric (Objective 4). Additionally, information collected using this study design enabled an assessment of design strategy as a useful criterion for ecological attributes of WRP sites (Objective 5). The following terms and resulting study design are applicable throughout this study and apply to all objectives.

Site is the term generally used in occupancy analysis to represent the sampling units of the population, or area, of interest. The population of interest for this project is WRP properties with an easement or cost-share agreement located within the Lower Grand River basin, Missouri. A ***WRP easement or cost-share agreement*** is a parcel of private property accepted into the WRP and is that portion of the property upon which the restoration practices or conservation practices are required (NRCS 2005b). My definition of site is that portion of the WRP easement or cost-share agreement that was either agriculturally cropped before its enrollment into the program or that portion of the WRP

easement or cost-share agreement that had a previously restored wetland and, thus, received restoration or enhancement efforts. This definition excludes forest stands of large trees present within the WRP property prior to restoration efforts and existing, remnant wetlands within these forest stands. I excluded this portion of the WRP properties from my assessment due to time and man-power constraints. Throughout the remainder of my thesis, I use *site* to designate that portion of a selected WRP property within the Lower Grand River basin that received restoration efforts and served as the sampling unit.

Season is the sampling period which should be defined such that the target species is either always present or always absent from the sites; i.e., the sites are closed to changes in occupancy; or changes occur at random (MacKenzie et al. 2006). Multiple, or repeat, surveys are then conducted within the season. I had three primary sampling periods: 1) 7 March 2007-4 May 2007, inclusively; 2) 14 May 2007-9 July 2007, inclusively; and 3) 23 July 2007-19 September 2007, inclusively, designed to encompass the breeding and larval development periods of all members of the regional species pool. Three detection methods were used, described below in Detection Methods, *Sampling techniques*, which served as secondary repeat samples within the primary sample period.

Number of sites vs number of repeat surveys: Allocating resources between the number of sites to survey and the number of repeat surveys involves balancing improvement gained in precision of the occupancy estimate by surveying an increased number of sites with the increased variance component related to detection probability resulting from not conducting enough repeat surveys (MacKenzie et al. 2006) MacKenzie et al. (2006) demonstrated that the minimum number of sites required

[assuming perfect detection ($p = 1$), an occupancy probability of 0.8, and a standard error of 0.05] was 64. This seemed a realistic starting point for this study as it likely represented the maximum number of sites that could be surveyed during a defined sampling period. My goal was to randomly select 20-22 sites from each design strategy for a sample size of 60-66 sites. Additionally, three detection methods (see detection methods below) were used during site visits to increase the number of repeat surveys and to increase the probability of detection, given a site was occupied.

Site characteristics: Site characteristics, or site-specific covariates, are those attributes of a site that one believes will influence the probability of occupancy among the sites sampled. My project is designed to evaluate whether the three design strategies employed on WRP properties in the Lower Grand River basin influence the probability that a site is occupied by amphibians; therefore, design strategy was my only site-specific covariate.

Site Selection: I stratified WRP properties within the Lower Grand River basin by design strategy and numbered each property, or site, within a strategy: 1) walk-away 1-22, 2) maximize hydrology 1-51, and 3) naturalistic 1-35. I used a table of random numbers to select 21 sites from the maximize hydrology and from the naturalistic design strategies (Snedecor and Cochran 1980). There were only 22 walk-away sites so, given my goal of sampling 64 sites (20-22 in each design category), I included all 22 walk-away sites in my sample pool. The resulting sample size of 64 selected sites included 22 walk-away, 21 maximize hydrology, and 21 naturalistic. I contacted landowners via United States Postal Service letter, telephone, and email to get permission to access their property. By the start of my field season in 2007, I had permission to access 13 walk-

away, 18 maximize hydrology, and 19 naturalistic properties for a total of 50 sites. Decreasing the number of sites to survey from 64 to 50 reduced the precision of the occupancy estimate; however, nine repeat surveys (three detection methods times three primary survey periods) should have somewhat offset this effect. Recalculating the standard error using a more realistic detection probability of 0.4, an occupancy probability of 0.8, nine repeat surveys, and 50 sites, the estimated standard error is 0.06 rather than 0.05 as assumed previously.

Quadrat, or within site sample sub-unit: Previously, a sampling unit was defined as a site, i.e., that portion of a WRP easement within the Lower Grand River basin that received restoration efforts. Size of individual sites included in this study ranged from approximately 1.5 ha to 242 ha; given time and man-power constraints, it was necessary to further divide a site into smaller sample sub-units to feasibly accomplish amphibian sampling within a site. I required a method that randomly sampled site-specific characteristics across an entire site in order to make inferences to the site; however, I also required a method that balanced the need to detect all species with the need to survey many sites (Adams et al. 1997). Williams et al. (2002) offer a useful rule of thumb that plot size should be sufficient for one to expect greater than one-half of the plots to contain individuals of the species of interest. Adams et al. (1997) made reference to a 2500-m² area that, according to their guidance, would require 9-10 traps; this size seemed large enough to encompass multiple features and to ensure more than one-half contained individuals of the target species but small enough to sample. Therefore, I chose a 2500-m² or a 50 m X 50 m quadrat as the sample sub-unit within a site. The Fishnet tool in XTools 3.1 for ArcGIS 9.2 desktop (ESRI 1999-2006) was used to overlay a 50 m X 50

m grid on each WRP property. The number of 50 m X 50 m quadrats within a site was determined by using the “Select by Location” tool in ArcGIS 9.2 and by using a finite population correction formula to determine the number of quadrats to sample on each site (Lohr 1999), because the number of quadrats per site represents a finite number from which to sample.

The greatest challenge in selecting the number of quadrats to sample per site involved a tradeoff between precision of sample estimate and amount of time available to sample all 50 sites across the three design strategies. Using a sampling error of 0.10 resulted in number of quadrats to sample per site ranging from 37 to 89 with a total of 3,263 quadrats to sample from all 50 sites. Assuming an average of 15 minutes to sample each quadrat, it would take 815.75 hours or 102 8-hour days to sample; this did not take into account travel time among quadrats and among sites nor allowing time to check traps. As it was not feasible to sample this many quadrats per site within the estimated amphibian reproductive season, I evaluated increasing the sampling error to 0.20 or 0.30, and the number of quadrats to sample per site decreased to 13 – 24 and 9 – 11, respectively. Based on field tests of sampling methods, I found a sampling error of 0.30 to be logistically most appropriate. This level of sampling error resulted in a total of 516 quadrats across 50 sites. I anticipated that this would enable completion of three primary sampling periods during the March through August amphibian reproductive season; however, a primary sampling period took an average of 59 days to complete which extended the field season into September.

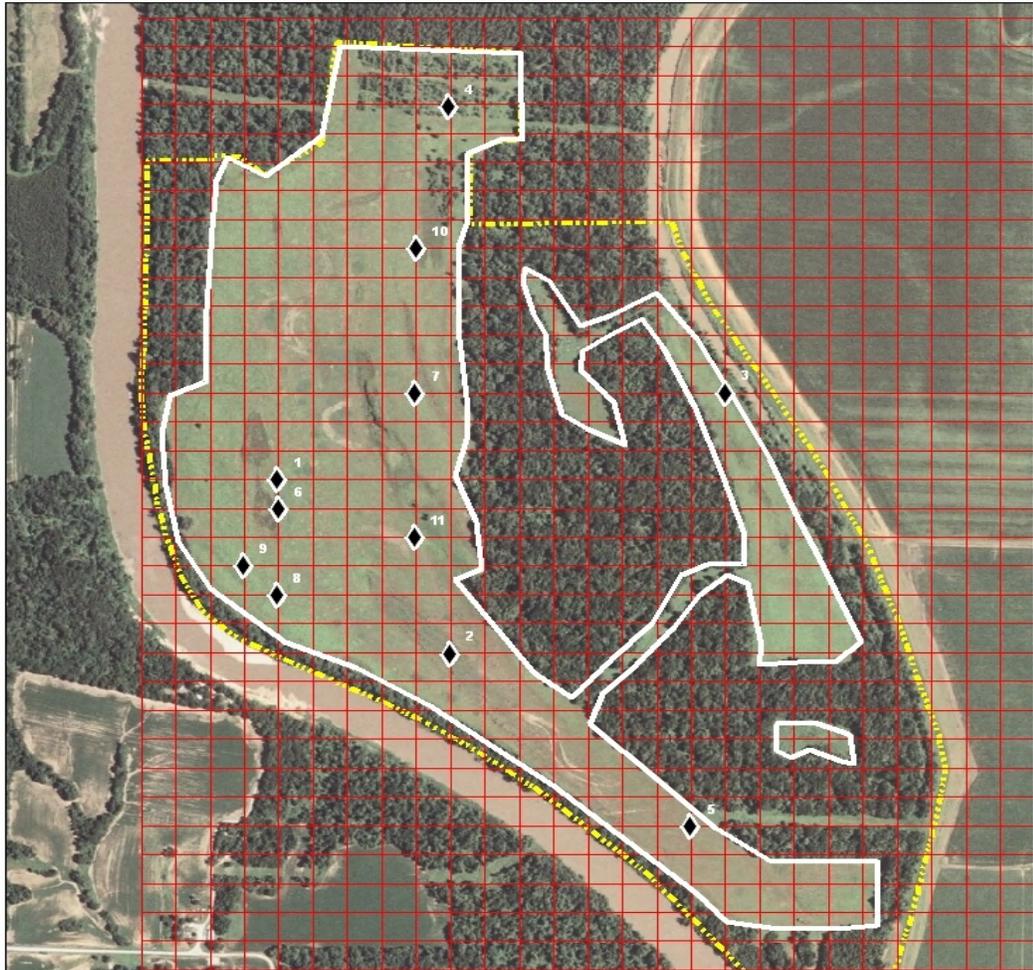
Hawth’s tool in ArcGIS 9.2 was used to randomly select the quadrats to sample on a site. The number of random points was determined by doubling the number of required

quadrats; e.g., if a site required sampling nine quadrats to meet the 0.30 sampling error requirement, I generated 18 random points with associated numbers. This accounted for situations when more than one number occurred in the same quadrat; in this instance, the smallest number was selected and the other number(s) in that quadrat deleted. I located the starting point for sampling to the northeast corner of each selected quadrat. This provided the coordinates necessary to locate the quadrat using a Global Positioning System (GPS) receiver. Quadrats selected to sample were those assigned the first 9-11 numbers; keeping any remaining quadrats in case a replacement quadrat was necessary based on field inspection. Replacement quadrats were used in the event that a selected sample quadrat included greater than 50% non-site area. Maps for each site, created in ArcGIS 9.2, included 2006 National Agricultural Imagery Project (NAIP) photography, the WRP property boundary, the restored site boundary, a 50 m X 50 m grid overlaid on the WRP property representing the pool of quadrats to sample, and selected sampling quadrats (Figure 4).

Detection Methods

It is very rare for a target species to always be detected when present at a sampled site; therefore, imperfect detection, i.e., $p < 1$, represents a component of variation that is strictly nuisance as it generally does not correspond to any ecologically relevant phenomenon; rather, it is a measure of how well individuals are counted (Royle and Dorazio 2008). However, not accounting for detectability can result in negatively biased estimates (Nichols et al. 1998, MacKenzie et al. 2002). In occupancy modeling, occupancy is the primary parameter of interest; nevertheless, detectability is an essential element due to the previously discussed issues with an investigator's inability to detect all

**MAXIMUM HYDROLOGY 38
11 Quadrats**



Township Range Section

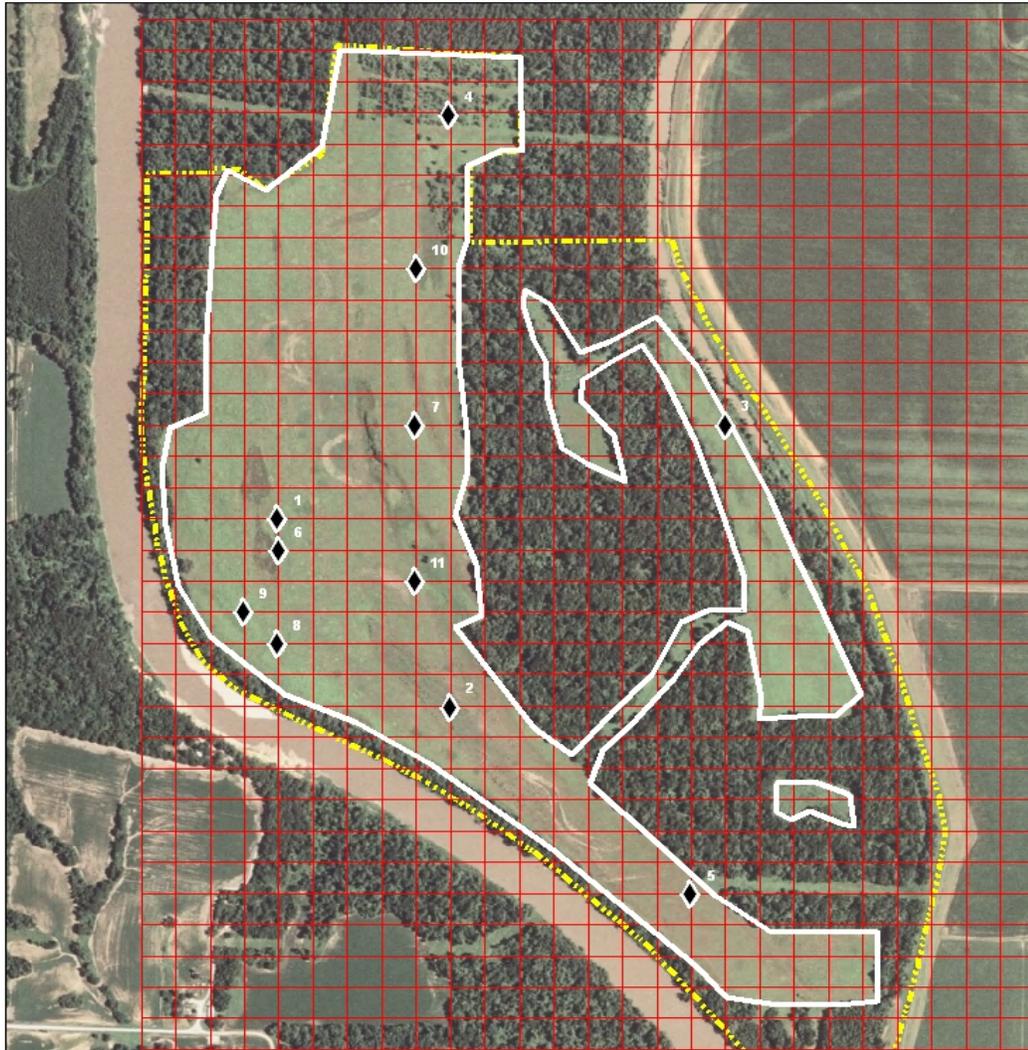
◆ Selected quadrats

Contact person:

Chariton county -



MAXIMUM HYDROLOGY 38
11 Quadrats



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◆ Selected quadrats

Contact person:

Chariton county - pg 39



Figure 4. Example of map used during 2007 field season for amphibian occupancy and species richness project in Lower Grand River basin, north-central Missouri. The photography is 2006 National Agricultural Imagery Project (NAIP), the yellow, dashed line is the WRP site boundary, the white, solid line is the restored site boundary, the squares represent a 50 m X 50 m grid delineating the within-site quadrats, and the diamonds are located on the northeast corner of each quadrat selected for sampling.

species present during a survey and the resulting biased estimates if detectability is not taken into account. For this study, covariates included to account for potential heterogeneity in detection probability estimates, i.e., seasonal response to environmental conditions and detection method, (Objective 2) also served to inform future studies. Amphibians exhibit strong seasonal movements so it is unlikely that detection probability remained constant across all primary survey periods; also, determining a season with the highest probability of detecting the most species while ensuring closure, i.e., species are present and available for detection throughout the defined season of the study, will prove useful to future amphibian investigations. Additionally, different amphibian species and different life stages likely exhibited varying degrees of vulnerability to the three detection methods (described below) used in this study; again, evaluating the efficacy of detection methods was not only informative for this study but provides useful information for future studies. Repeat surveys conducted over a defined season accounts for imperfect detection by reducing the probability that a nondetected species was present at a site but went undetected. Detection histories, composed of a series of detections (recorded as a 1) and nondetections (recorded as a 0) of the target species generated during repeat surveys using methods described below, were used in the occupancy, detection, multi-state occupancy, and species richness analyses to accomplish objectives 1-4.

Sampling techniques: Three sampling techniques were used in this study to serve as repeat surveys during a site visit and to increase the probability of detecting target species. Techniques included aquatic funnel-trap surveys, visual encounter surveys (VES), and dip-net surveys. The resulting idealized hierarchical study design yielded between 450 and 550 quadrats sampled per primary survey period (Figure 5). Aquatic

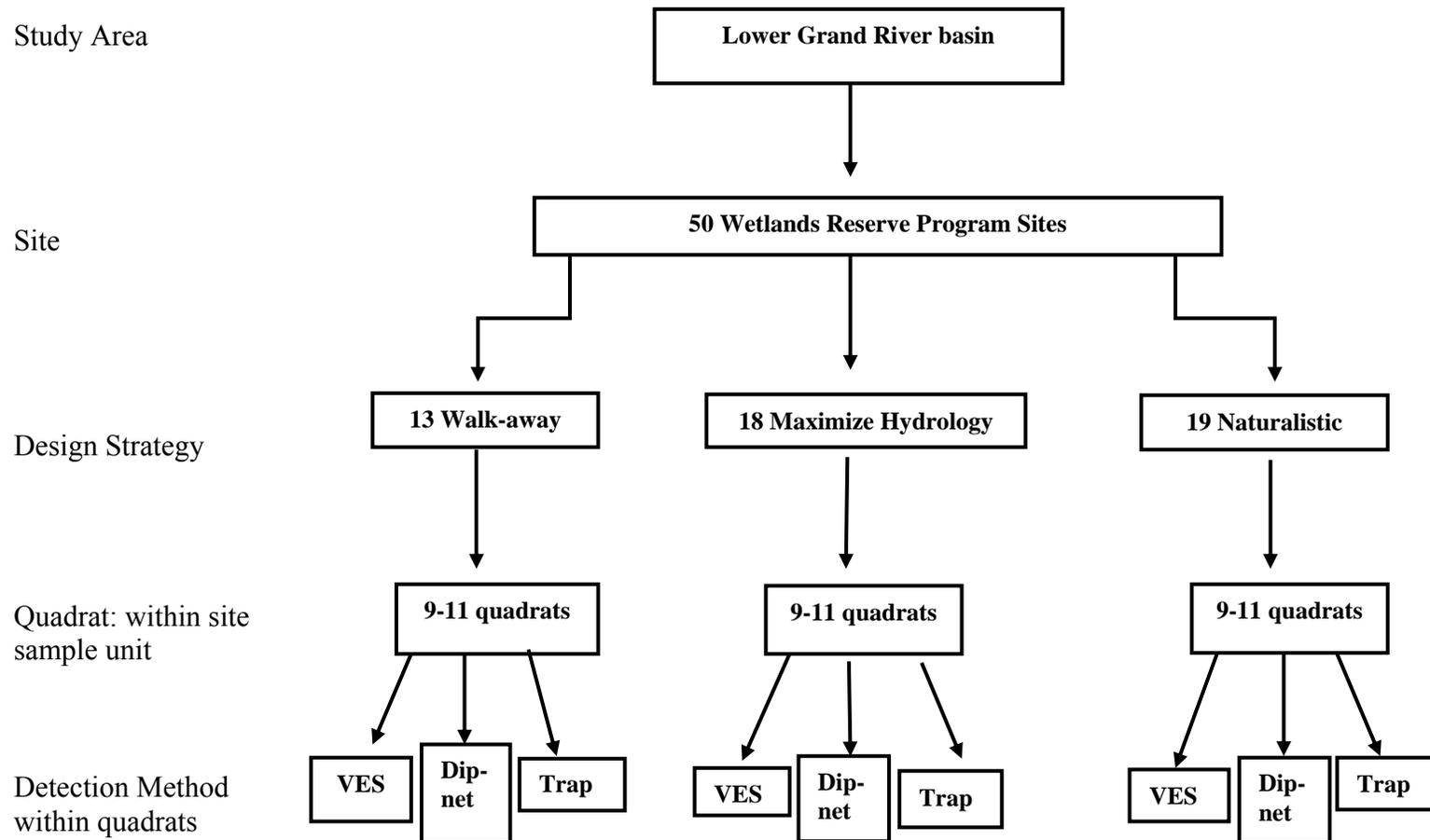


Figure 5. Idealized, hierarchical study design for amphibian occupancy and species richness project conducted during summer of 2007 in north-central Missouri. The study design included three primary survey periods during which each site was sampled using three detection methods on nine to eleven randomly selected quadrats within a site. Quadrats measured 50 m X 50 m. The three detection methods included visual encounter surveys (VES), dip-nets, and aquatic funnel traps. One VES was conducted per quadrat in both terrestrial and aquatic habitats whereas the number of dip-net surveys and traps deployed was conditional on the amount of aquatic habitat within a quadrat. See text for additional explanation on number of detection methods applied per quadrat.

funnel-trap and dip-net surveys were only deployed in aquatic habitats within a selected quadrat whereas VESs were conducted on both terrestrial and aquatic habitats within a selected quadrat. Sampling techniques generally followed guidelines in Heyer et al. (1994) and Olson et al. (1997). The exception occurred in deep water areas as described below. Cleaning and disinfecting protocols involved soaking all sampling equipment in a 10% bleach solution after use on one WRP site and before being used on another site.

Aquatic funnel trap surveys: Funnel traps reliably determine presence of most wetland-breeding amphibians and are capable of detecting small populations of cryptic larvae that may go undetected by other methods (Adams et al. 1997). Aquatic funnel trap surveys were conducted using Gee Exotic Minnow¹ traps, a commercially available minnow trap with small 0.3175-cm mesh and 2.54-cm openings on each end.

The optimal number of traps to deploy at a site involves a tradeoff between level of precision required, expense, and effort available as well as balancing the need to detect all species present with the need to survey many sites. The general deployment design followed Adams et al. (1997) with modifications relevant to hydrologic and biological features of my study sites. Water depths varied greatly spatially and temporally across sample quadrats with some quadrats completely dry and others covered with water deeper than trap height (>22 cm). My objective was to sample the entire water column within the constraints of sampling equipment, therefore, the following criteria were used to place traps within a quadrat. If a quadrat was 100% dry or the water depth was < 10 cm, then no traps were set as a minimum water depth of 10 cm was required for the water to reach the trap funnel. If water depth was ≥ 10 cm on a quadrat then traps were placed

¹ The use of trade, product, industry or firm names or products or software or models, whether commercially available or not, is for informative purposes only and does not constitute an endorsement by the U.S. Government or the U.S. Geological Survey.

according to the following criteria: 1) if water depth across the entire quadrat varied from 10 cm-41 cm, nine traps were placed in the quadrat in roughly a 3 X 3 grid, 2) if any portion of the quadrat had water depth >42 cm, traps were placed as described in 1) but one trap was replaced with a vertical stack of two traps for a total of 10 traps in the quadrat, and 3) if any portion of quadrat had water depth > 64 cm, traps were placed as described in 1) but one trap was replaced with a vertical stack of three traps for a total of 11 traps in the quadrat. Water depth was recorded in centimeters by placing the end of a meter stick on the pool's substrate and recording the reading at the water's surface for each trap location. The need for stacked traps was determined after the initial one through nine traps were placed in the quadrat, depth measurements were taken, and each trap had received a number designation ranging from one to nine. If more than one location within a quadrat had water depth sufficient for stacked traps, a number from one to nine, inclusively, was randomly selected and the stacked traps were placed at the randomly selected spot.

Stacked traps were installed by placing each on 1.1 m fiberglass electric fence posts with two sliding clips and one stationary clip. Height of traps on the sliding clips could be adjusted so placement of the bottom trap was just above the wetland substrate allowing the middle trap to be placed just above the bottom trap. The clip at the top of the post was not adjustable and was used only in water depth > 64 cm. Water deeper than 84 cm resulted in submersion of all three traps leaving the portion of the water column above the upper trap unsampled. Water depths greater than approximately 110 cm exceeded the height of our chest waders and were not sampled. This situation generally occurred in borrow areas that often had steep slopes; here we generally placed traps along

the edge of the levee associated with the borrow and, if appropriate, along the opposite side in the deepest water through which we could wade.

Traps were checked and collected the day following when set; set duration ranged from 16 to 22 hours. When checked, the contents of each trap was identified and enumerated. Size classes were recorded for any species that exhibited variation in size. Captured amphibians were identified to species when possible and life stage noted (adult, larva, metamorph, juvenile). Anurans were designated as larva if they were free-swimming tadpoles, as metamorphs when at least one forelimb had emerged (Gosner stages 42-46) (Gosner 1960) and as juveniles based first on whether the tail had been reabsorbed indicating metamorphosis was complete and then by size (froglets and toadlets were classified as juveniles). Salamanders were classified as larva if they were free-swimming with external gills and a dorsal fin, as metamorphs when gills and the dorsal fin had been reabsorbed, and as juveniles based on size. Leopard frog tadpoles were recorded as leopard frog complex as it was not possible to distinguish Plains leopard frogs (*Lithobates blairi*) from southern leopard frogs (*L. sphenoccephala*) until they reached the metamorph stage. Similarly, there was a timeframe when it was difficult to distinguish between boreal chorus frog (*Pseudacris maculata*) and spring peeper (*P. crucifer*) larva; we recorded the detection as *Pseudacris* spp. if a species determination could not be made. We made no distinction between grey treefrog (*Hyla chrysoscelis*-*Hyla versicolor* complex) species due to the similarity of their external appearance. Fish species were also identified to Family and counted; any fish that could not be identified to Family were classified as "other." Length was measured for species exhibiting size variation.

Visual encounter surveys (VES): Visual encounter surveys were conducted by walking through a predetermined area and visually searching for evidence of amphibians. The predetermined search area for this project was the area within a site sampling sub-unit, or quadrat. Visual encounter surveys were conducted after locating the northeast corner of a selected quadrat using coordinates determined during the quadrat selection process described earlier; VESs continued while locating the remaining three corners of a quadrat by walking the perimeter of the quadrat and walking the diagonal from the northeast corner to the southwest corner. Amphibian presence was quantified as adult amphibians, juvenile amphibians, metamorphs, larva, or egg masses. Additionally, adult or juvenile anurans seen but not identified to species were recorded as unknown jumpers. If a quadrat was 100 percent dry, no additional amphibian surveys took place after completing VESs within that quadrat and we moved on to sample the next quadrat.

Dip-net surveys: Dip-net surveys were conducted using a commercially available fish landing net, Ed Cumings² model 270F12, with an approximate opening of 38 cm X 23 cm X 30.5 cm deep and 0.3175-cm sized mesh. Aquatic habitats within each sample quadrat were dip-net surveyed with 15 sweeps if the habitat type composed more than 33% of a quadrat and with 10 sweeps if it composed less than 33% of a quadrat. All areas with water present within a quadrat were dipnetted using these criteria; however, occasionally, very small, shallow puddles only required one or two sweeps to survey the entire puddle. All individuals captured in a dip-net were identified and enumerated as described in the *Aquatic funnel trap surveys* section.

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Environmental characteristics: Environmental information collected at the quadrat, or within site sub-unit, level represented survey-specific covariates. Survey-specific covariates were applicable to detection probability estimates as they represented the environmental characteristics that I assumed would influence the probability of detecting a species (Objective 2). Additionally, this information, when used to calculate an average value for a site, provided a means to compare whether design strategy served as a criterion that reflected a functional ecological attribute of a site (Objective 5). Information recorded at each quadrat on both the day of initial visit and on the day of trap collection included date, begin time upon arrival at the quadrat, end time upon departure from the quadrat, and air temperature ($^{\circ}\text{C} \pm 0.1$). Additionally, water depth ($\text{cm} \pm 2$), surface water temperature ($^{\circ}\text{C} \pm 0.1$), and vegetation category (categories described below) were recorded for each trap if traps were deployed.

A visual assessment was made as to percent of quadrat that was terrestrial (dry) or aquatic (wet) and the proportion of vegetation covering the aquatic portion of the quadrat was noted. Vegetation was classified based on wetland type following Cowardin et al. (1979) (Figure 6). Our categorization diverged from this classification regime in the persistent subclass in which robust, erect, perennial vegetation was specified as one category and annual, grass, or grass-like vegetation as a separate category because these vegetative categories represent different wetland hydroperiods and amphibian habitats.

Survey timing and specimen collections: Sites were visited during daylight hours. All amphibian species and life history forms were identified in the field unless there was some uncertainty related to identification. In such cases, the specimen was collected and either preserved in a buffered 70% ethanol solution (McDiarmid and Altig 1999) and

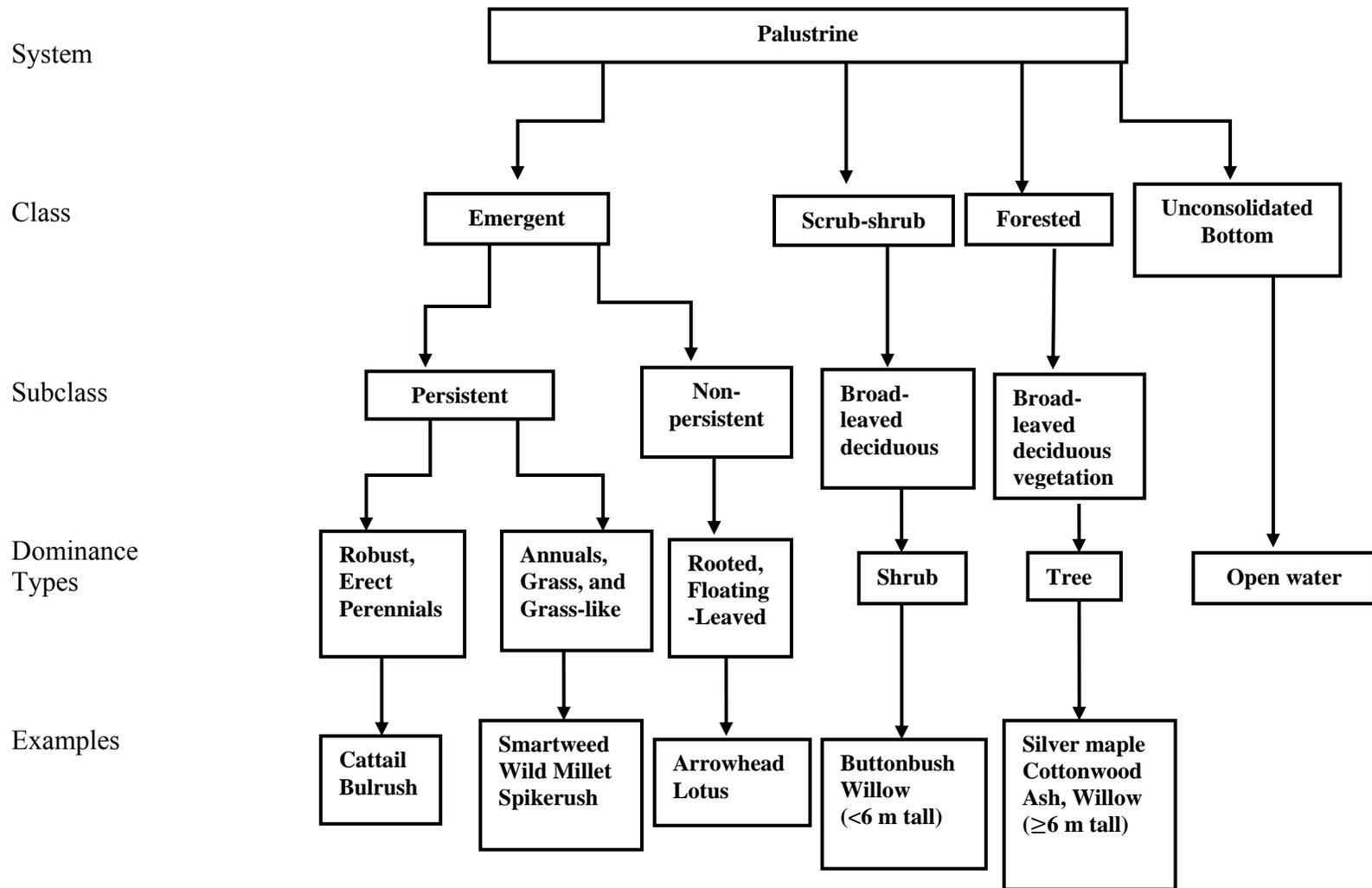


Figure 6. Vegetation classification scheme used for amphibian occupancy and species richness project conducted during the summer of 2007 in Lower Grand River basin, north-central Missouri. Classification followed Cowardin et al. (1979); however, dominance types, although similar to the Cowardin classification, are the categories recorded for this project. The primary divergence occurred in the persistent subclass in which two dominance types were differentiated. Examples of vegetation placed in each dominance type are listed in the last row.

identified in the laboratory with keys from Gosner (1960) and Altig (1970) or it was transferred to the laboratory and raised until it metamorphosed and species identification could be determined. Voucher specimens for each species were collected and preserved as described above and kept for future reference and verification.

DATA ANALYSIS

Occupancy Probability Estimation

Occupancy model: Occupancy modeling makes it possible to model the probability of a site being occupied as a function of measured variables, or covariates. Estimating the site occupancy of each species at randomly selected sites within a defined area produces an estimate that represents the proportion of sites occupied by a species (Rice et al. 2006). This also represents the species' distribution within the defined area. For my study, occupancy estimates served as a means to determine amphibian distribution which was used to evaluate if hydrological wetland characteristics had been restored on WRP sites in the Lower Grand River basin (Objective 1). I predicted amphibian occurrence would vary at the site scale based on design strategy. Specifically, I predicted the probability of amphibian occupancy, i.e., distribution, would be lowest on walk-away, intermediate on maximize hydrology, and highest on naturalistic sites due to the varying hydrological wetland characteristics restored within each design strategy. Model selection provided a process whereby I tested this hypothesis (Burnham and Anderson 2002). I used the single-species, single-season occupancy model described by MacKenzie et. al. (2002) that requires multiple visits, or sampling occasions, are made to n number of sites resulting in a series of detection histories composed of 0's and 1's where a 0 denotes no detection of the target species and 1 denotes detection of the target

species. No uncertainty is associated with detection of a species; if a site visit results in a 1 (species detected in any of the sampled quadrats), then the site is occupied (true state is 1). If a site visit results in a 0 (no detection of the species in any of the sampled quadrats), then two situations are possible; the site may be truly unoccupied or the site could be occupied but the species not detected (possible true states are 0, 1). Each of the three detection methods; aquatic funnel traps, visual encounter surveys, dip-nets; served as a repeat, secondary survey resulting in nine surveys over the course of the three primary sampling periods. An example detection history of 000 010 101 indicates the species was not detected by any of the three detection methods in the first primary sampling period, the species was detected by detection method two in the second primary sampling period, and the species was detected by detection methods one and three in the third primary sampling period. In occupancy modeling, the symbol ψ (psi) denotes the probability that a species is present at a site and p denotes the probability of detecting a species at a site, given the site is occupied. Data analysis was conducted using program PRESENCE 2.4 (Hines 2006). Amphibian abbreviations are defined in Appendix A and data used for the occupancy analysis is presented in a series of X-matrix tables in Appendix B.

Link function: Program PRESENCE uses the logit link as the default link when covariates are used in a model (Donovan and Hines 2007). The advantage of using a link function is it bounds the probability estimates between 0 and 1 (MacKenzie et al, 2006). However, when species are either not detected on any or only a few sites surveyed or when species are detected on most or all sites surveyed, the result is either a large negative or positive coefficient, i.e., beta, associated with the untransformed covariate

estimate (generally $-20 \leq$ or ≥ 20 , respectively). This indicates the real, or transformed, parameter is near the boundary of zero in the case of a large negative untransformed beta estimate or near the boundary of one in the case of a large positive untransformed beta estimate. In these situations, a nonsensical real parameter estimate results and the standard error cannot be estimated (Long et al. 2009). One recommended approach to address this situation in program PRESENCE is to delete the beta parameter associated with ψ , the occupancy parameter, from the design matrix and fix the parameter to 1 (Hines 2009 www.phidot.org). This allows estimation of detection to proceed. Using this method, in essence, reduces the model to the constant $\psi(\cdot)$ model.

Site occupancy: I included design strategy as a site-level covariate for the occupancy analysis. Design strategy was treated as a categorical variable where M represents maximize hydrology sites, N represents naturalistic sites, and W represents walk-away sites; W served as the reference covariate. A reference to design strategy using an upper case letter refers to all sites within a strategy and a lower case letter preceding a site number refers to that specific site. Model 1 $\psi(\cdot)$ is the null model that assumes no effect of covariates; i.e., there is one occupancy estimate that is constant across all sites. Model 2, $\psi(W M N)$, tests whether the species responded differently to each of the design strategies; i.e., there are three occupancy estimates, one for each design strategy. The remaining Models 3-5; $\psi(W \{MN\})$, $\psi(N \{WM\})$, and $\psi(M \{WN\})$; pool site information and evaluate whether a species responded to the design strategies in a more similar fashion than different, resulting in two occupancy estimates. For example, Model 3 results in one occupancy estimate for walk-away sites and a second occupancy estimate for maximize hydrology and naturalistic sites.

Detection probability: A two-stage analysis was conducted in which I first ran the occupancy models with detection probabilities constant, i.e., $p(\cdot)$ (Franklin et al. 2000, Washburn et al. 2004). Once the most supported occupancy model was determined, this result was paired with the detection models in the Detection section below.

Analysis: The single-species, single-season occupancy model assumes population closure such that there are no non-random changes to occupancy throughout the defined season. Given the length of my field season, I had several closure violations, particularly for the early- to mid-season breeding species (small-mouthed salamander (*Ambystoma texanum*), American toad (*Anaxyrus americanus*), grey treefrog complex, spring peeper, and boreal chorus frog). To satisfy the closure assumption, the data set for each species was truncated by only including the portion of data between the first and last detections, exclusively (MacKenzie et al. 2002, 2006); however, if a species was detected on either the first or last day of my field season, those dates for that species were retained. The candidate set of models included five occupancy and one detection parameterizations for a total of five models. An information-theoretic approach was applied to determine the most approximating model for each species using a second order Akaike Information Criterion which corrects for small sample sizes (AIC_c); an effective sample size of 50, i.e., number of sites surveyed, was used for all AIC_c calculations.

Using an information-theoretic approach based on AIC provides several quantitative means to examine the strength of evidence for models within a candidate set of models including AIC delta values, model probabilities, i.e., Akaike weights (w_i), and evidence ratios (ratios of the model likelihoods or model probabilities) (Burnham and Anderson 2002). The level of support for a particular model within a candidate set can

be estimated with AIC delta values (Δ_i); i.e., delta value of 1-2 provides substantial support, 4-7 provides considerably less support, and >10 provides essentially no support (Burnham and Anderson 2002). Akaike weights; i.e., w_i , provide a means to determine the probability that model i is the most approximating model in the candidate set in that it represents the least loss of information between truth and the model; $w_i \geq 0.90$ indicates strong support that the model is the most approximating model in the candidate set (Burnham and Anderson 2002). Evidence ratios provide a means to determine the strength of evidence between two models in the candidate set (w_i/w_j); generally, a value ~ 150 (equivalent to $\Delta_i \sim 10$) is necessary to provide strong evidence in favor of model i over model j (Mazerolle 2006, Anderson 2008).

Species in the occupancy analysis included small-mouthed salamander, American toad, northern cricket frog (*Acris crepitans*), grey treefrog complex, spring peeper, boreal chorus frog, Plains leopard frog, American bullfrog (*L. catesbeiana*), and southern leopard frog. Additionally, I included leopard frog complex (*L. blairi-sphenocephala* complex), which includes leopard frog detections that could not be differentiated to species. Given the low detections of spring peepers (13 detections representing 15 individuals) versus detections of boreal chorus frogs (45 detections representing 380 individuals) I combined the *Pseudacris* spp. and boreal chorus frog detections and hereafter refer to this group as *Pseudacris* spp. One detected species, the central newt (*Notophthalmus viridescens louisianensis*), occupied two sites with a total of five detections representing seven individuals so was not included in the analysis.

Model Fit: Goodness of fit was assessed on the most general model that provided adequate parameter estimates within each species' candidate set of models using the

method described by MacKenzie and Bailey (2004) and implemented in Program PRESENCE 2.4 (Hines 2006). This method is based on a Pearson chi-square statistic with the test statistic calculated as:

$$\chi^2 = \sum (O - E)^2/E$$

where O is the observed frequency of detection histories and E is the expected frequency of detection histories (MacKenzie and Bailey 2004). As many of the E values may be relatively small (<2), it is unlikely that χ^2 will exhibit a chi-square distribution. I followed the recommendation by MacKenzie and Bailey (2004) to use the parametric bootstrap method so it was not necessary to assume a chi-square distribution, thus, resulting in the test being less sensitive to the small expected values.

Count data often have overdispersion due to violation of the independent and identically distributed data assumption (Anderson 2008). The overdispersion parameter, \hat{c} , was estimated using

$$\hat{c} = \chi_{obs}^2 / \bar{X}_B^2$$

where \bar{X}_B^2 is the average of the test statistic obtained from the parametric bootstrap.

Overdispersion parameter values in the 1-3 range are not unusual and may be substantially higher than 4-5 when counting species with a high degree of dependency such as schooling fish, swarms of insects, or groups of tadpoles (Anderson 2008). I used 10,000 parametric bootstraps to calculate \hat{c} ; if lack of model fit was indicated by a value of $\hat{c} > 1$, I adjusted AIC_c values to quasi- AIC_c (QAIC_c) and inflated standard error estimates by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004) as overdispersed data indicate the expected structure of the model is adequate but the variance structure is inadequate

(MacKenzie et al. 2006). I also augmented the number of parameters, K , by one to account for the estimation of c (Burnham and Anderson 2002).

Effect of covariates: Odds and odds ratios provide a means to interpret the effect of covariate beta coefficients (MacKenzie et al. 2006). Odds are a measure of success and are calculated by dividing the probability of success by the probability of failure; e.g., in an occupancy framework, success is the probability of occupancy (ψ) and failure is the probability of non-occupancy ($1-\psi$), i.e., odds = $\psi/(1-\psi)$ which equates to the odds of a site being occupied. Odds of 2:1 indicate the site is twice as likely to be occupied than unoccupied and, as probability = odds/(1+ odds), then, in this example, $\psi = 2/1+2$ or 0.67 probability of occupancy. An odds ratio is the ratio of odds at 2 different types of places: Odds Ratio (OR) = odds₂ / odds₁. The logit link, used to model the probability of success as a function of covariates in logistic regression, can be rearranged in terms of the odds of success for sampling unit i :

$$\begin{aligned} \text{Logit}(\theta_i) &= \ln (\theta_i/1- \theta_i) = \beta_0 + \beta_1x_{i1} + \beta_2x_{i2} + \dots + \beta_u x_{iu} \\ &= \theta_i/1- \theta_i = \exp(\beta_0 + \beta_1x_{i1} + \beta_2x_{i2} + \dots + \beta_u x_{iu}) \\ &= \exp(\beta_0)\exp(\beta_1x_{i1})\exp(\beta_2x_{i2}) \dots \exp(\beta_u x_{iu}) \end{aligned}$$

where θ_i is the parameter of interest for the i th sampling unit. $\text{Exp}(\beta_u)$ can be interpreted as the odds ratio for a one-unit change in the covariate x_{iu} (MacKenzie et al. 2006). An odds ratio of 1.0 indicates the covariate has no effect, an odds ratio >1.0 indicates a positive effect, and an odds ratio <1.0 indicates a negative effect; therefore, if a 2-sided, 95% confidence interval (CI) for the odds ratio includes 1.0, the covariate has no effect, if the interval is <1.0 , the covariate has a negative effect, and if the interval is >1.0 , the covariate has a positive effect (MacKenzie et al. 2006). For my study, this is relevant in

models that evaluate the probability of occupancy on sites with different design strategies and provides a means to determine if a particular strategy has a positive, neutral, or negative effect on the occupancy estimate relative to another strategy.

Detection Probability Estimation

Site occupancy: The most supported occupancy model for each species as determined in the occupancy analysis was used in the detection probability analysis. My interest for this analysis was to evaluate if amphibian seasonal behavior patterns or if varying vulnerability by amphibian species to the three detection methods resulted in heterogeneous detection probabilities (Objective 2). Covariates included day of survey and detection method. If the most supported model included day of survey, then I determined that most of the variation in detection probability was explained by seasonal movements whereas if the most supported model included detection method, then I determined that most of the variation in detection probability was explained by the type of detection method used.

Detection probability: I used the single-species, single-season occupancy model described in the Data Analysis-Occupancy section. Each of the three detection methods; visual encounter surveys, dip-nets, and aquatic funnel traps; served as a repeat survey resulting in nine surveys over the course of the three primary sampling periods. Day of survey served as a surrogate to air and water temperature as these two variables exhibit a broadly predictable annual pattern of increasing from spring to summer, reaching a peak, and then decreasing into fall.

The detection intercept ($p(int)$) estimates detection probability whereas covariates influence the slope of the line, thus, giving some indication of the magnitude of effect. I

assumed that detection probability would exhibit temporal variation due to the seasonal behavioral patterns in amphibian breeding strategies and amphibian response to changing environmental conditions (Bailey et al. 2004). I also assumed that different amphibian species and amphibians at different life history stages would exhibit varying degrees of vulnerability to the three detection methods used in this project. To test these hypotheses, I included four detection parameterizations in my candidate set of models: 1) $p(\cdot)$ forced the detection probability intercepts to be constant across all three primary sampling occasions which implies detection probability is constant with no temporal variation and no effect of detection method; 2) $p(\text{day})$ included day of survey as a survey-specific covariate which should not only indicate that detection probability does reflect temporal variation but should also reflect both seasonal changes in species behavior pattern as well as response to changing environmental conditions; 3) $p(\text{day sq})$ included day of survey squared to test whether the effect of survey day was quadratic rather than linear; and 4) $p(\text{method})$ allowed detection estimates to vary by secondary sampling occasions which evaluated whether species detection varied by detection method. The DAY and DAY SQ covariates were continuous; using pooled data from the three primary sampling periods, I standardized the DAY and DAY SQUARE covariates into Z scores by using the STANDARDIZE function in Excel ($Z = (\text{observed data} - \text{mean observed data}) / \text{standard deviation}$) resulting in a mean of 0.0 and a standard deviation of 1.0 (Donovan and Hines 2007). Data analysis was conducted using program PRESENCE 2.4 (Hines 2006).

Analysis: I used the same truncated data set described in the Data Analysis-Occupancy section. Candidate models included one occupancy and four detection parameterizations for a total of four models. An information-theoretic approach was used

as described in the Data Analysis-Occupancy section. Species in this analysis were also the same as described in the Data Analysis-Occupancy section.

Model Fit: Goodness of fit was assessed on the most general model that provided adequate parameter estimates within each species' candidate set of models using the method described by MacKenzie and Bailey (2004) and implemented in Program PRESENCE 2.4 (Hines 2006) as described in the Data Analysis-Occupancy section.

Multi-State Occupancy Probability Estimation

Multi-state occupancy models: The multi-state occupancy model provided a means to evaluate if biological wetland characteristics had been restored to WRP sites based on amphibian recruitment success (Objective 3) using detection/nondetection of metamorphs. In the context of occupancy modeling, detection of metamorphs results in estimation of occupancy with multiple states: i.e., metamorphic and non-metamorphic individuals. The detection history is extended to deal with two states of occupancy with 0 denoting no detection of occupancy in any of the sampled quadrats within a site, 1 denoting detection of occupancy in at least one of the sampled quadrats within a site but no evidence of recruitment (no detection of metamorphs), and 2 denoting detection with evidence of recruitment (detection of metamorphs) in at least one of the sampled quadrats within a site (MacKenzie et al. 2006). I modeled this situation by extending the notation and modeling of the single-species, single-season approach described in the Data Analysis-Occupancy section using the multi-state occupancy estimation model in Program MARK 5.1 (White and Burnham 1999). The multi-state model in MARK implements a conditional binomial probability structure and parameterizes the model as described in Nichols et al. (2007); I used the same parameterization in my analysis.

Similar to the single-species, single-season occupancy model, there is uncertainty associated with detection only now it is associated with assignment of life history stage or species state. No uncertainty is associated with detection of metamorphs; if a site visit results in a 2 (metamorph detected), then the site successfully produced recruits (true state is 2). If a site visit results in a 1, then two situations are possible; the site was occupied with no successful recruitment or the site successfully produced recruitment (possible true states 1, 2). If a site visit results in a 0 (no detection of the species), then three situations are possible; the site may be truly unoccupied, the site could be occupied but the species not detected, or the site could be occupied with evidence of successful recruitment (possible true states are 0, 1, 2). Each of the three detection methods; visual encounter surveys, dip-nets, and aquatic funnel traps; served as a secondary repeat survey within three primary sampling periods resulting in nine surveys. As before, an example detection history of 000 010 002 indicates the species was not detected by any of the three detection methods in the first primary sampling period, the species was detected by detection method two in the second primary sampling period with no evidence of successful recruitment (no detection of metamorph), and the species was detected by detection method three with evidence of successful recruitment (detection of metamorph) in the third primary sampling period.

The parameters for the multi-state model and associated descriptions, modified to fit my project, include:

ψ_i^1 = probability that site i is occupied regardless of life history stage;

ψ_i^2 = probability that successful recruitment (detection of metamorphs) occurred, given that the site is occupied;

p_{it}^1 = probability that occupancy is detected for site i , period t , given that true state = 1;

p_{it}^2 = probability that occupancy is detected for site i , period t , given that true state = 2;

δ_{it} = probability that evidence of successful recruitment found, given detection of occupancy at site i , period t .

The overall, unconditional probability that a site successfully produced recruits is the product of $\psi_i^1 * \psi_i^2$, a derived parameter. Real parameters are estimated through the maximum likelihood function directly from the model whereas derived parameters are estimated from real parameters or beta coefficients. Real parameters included in this analysis are ψ_i^1 , ψ_i^2 , p_{it}^1 , p_{it}^2 , and δ_{it} . Data used for the multi-state occupancy analysis is presented in a series of X-matrix tables in Appendix C.

Analysis: The multi-state, single-species, single-season occupancy model assumes population closure such that there are no non-random changes to occupancy throughout the defined season. I used a truncated data set to satisfy the closure assumption as described in the Data Analysis-Occupancy section. My primary interest in this analysis was 1) determining the probability that successful recruitment (detection of metamorphs) occurred, given that the site is occupied (ψ_i^2), 2) determining if the probability of detecting successful recruitment (δ_i), varied by a) primary sampling period or b) detection method, and 3) overall probability that a site successfully produced recruits ($\psi_i^1 * \psi_i^2$). Due to the small sample size and sparse data I kept the models in the candidate set of models relatively simple by maintaining both occupancy parameters and both detection parameters as constant and did not use any covariates. As there were no site-specific covariates, i was dropped from the parameter descriptions as models are based on the assumption that sites have similar characteristics (MacKenzie et al. 2006).

The candidate set of models included three models with both occupancy parameters (ψ^1 and ψ^2) and both detection parameters (p^1 and p^2) as constant and distinct and three parameterizations for delta (δ), i.e., probability of observing evidence of recruitment given detection of occupancy. Each candidate set included three models with one of three parameterizations for delta: 1) delta (δ) constant, 2) delta varying by primary sampling period (δ_t ; $t = 1, 2, \dots, 9$ where survey period 1 = 1, 2, 3; survey period 2 = 4, 5, 6, and survey period 3 = 7, 8, 9), and 3) delta varying by detection method or secondary survey occasion (δ_m ; $m = 1, 2, \dots, 9$ where VES = 1, 4, 7; net = 2, 5, 8; and trap = 3, 6, 9).

An information-theoretic approach was again used to determine the most supported model for each species using a second order Akaike Information Criterion as described in the Data Analysis-Occupancy section. I used Program MARK to calculate a model-averaged estimate of ψ^1 , ψ^2 , and the overall probability that a site successfully produced recruits, $\psi^1 * \psi^2$, for each species' candidate set of models. Model-averaging is a means to make use of information from all models in a candidate set of models and is particularly useful if model selection uncertainty is high. A model set in which the most supported model receives high weight ($w_i > 0.90$) would see little difference in a model-averaged estimate as the remaining models have virtually no weight and contribute little to the average (Anderson 2008).

Species and groups in this analysis included small-mouthed salamander, northern cricket frog, grey treefrog complex, *Pseudacris* spp., Plains leopard frog, American bullfrog, southern leopard frog, and leopard frog complex. Again, a truncated data set as

described in the Data Analysis-Occupancy section was used to ensure the closure assumption was satisfied for all species or groups.

Model Fit: There is no established goodness-of-fit test for the multi-state occupancy model. Nichols et al. (2007) found the Pearson goodness-of-fit statistics could not be computed with their most general model as the cell pooling algorithm left fewer cells than estimated parameters. A common problem that arises in both capture/recapture data and occupancy data is very small expected values (<2) for capture or detection histories. This is generally corrected by pooling cells until the sum of expectations is >2 . However, as noted by Nichols et al. (2007), this method is better suited for large sample sizes and creates issues with small sample sizes and sparse data sets. I, therefore, took their alternate approach and used a deviance-based goodness-of-fit statistic to test for model fit or overdispersion. This method uses deviance, i.e., the difference between the $-2\log$ likelihood of the saturated model and the $-2\log$ likelihood of model_{*j*}, divided by degrees of freedom (df), i.e., the difference between the number of unique detection histories in the saturated model minus the number of estimable parameters in model_{*j*}, to calculate a variance inflation factor, \hat{c} . A saturated model, by definition, is a model in which the data fit the model perfectly as expected values for detection history probabilities are computed from the raw data by dividing the frequency of each history by the total number of sites. This then becomes the standard upon which the other models are measured (Donovan and Hines 2006). A small deviance value is better than a larger value as a small value indicates the model is closer to the saturated model and does a better job of explaining the data. The drawback to this method is, in capture/recapture situations, deviance does not follow the chi-squared distribution well enough to provide a

valid test of model fit and this likely is also true for occupancy models (Rotella 2008, MacKenzie and Bailey 2004). Therefore, I primarily used deviance to estimate the amount of overdispersion in the data through calculating \hat{c} .

If overdispersion was indicated by a value of $\hat{c} > 1$, I adjusted AIC_c values to quasi- AIC_c ($QAIC_c$) and inflated standard error estimates by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004). I also augmented the number of parameters, K , by one to account for the estimation of c (Burnham and Anderson 2002).

Species Richness Estimation

Use of a regional species pool enabled an assessment of wetland restoration efforts as indicated by the relative species richness metric (Objective 4). However, in order to define the relative species richness metric, it was necessary to first estimate species richness for each sampled WRP site. Williams et al. (2001) state that “the particular approach used to estimate species richness depends on the type of community sampling that is conducted.” My sample design followed Pollock’s (1982) robust design which involves sampling at two temporal scales (Figure 7) in which the primary sampling periods represent one temporal scale and the three detection methods represent the second temporal scale (Figure 8). This was an appropriate approach when conducting a comparative occupancy analysis among all 50 sites in time and space. This species richness analysis, however, is a site-by-site analysis with comparison among sites available through the use of summary statistics such as means (MacKenzie et al. 2005). By conducting a site-by-site analysis, my study design can then be viewed as a quadrat-based sampling approach that involved subdividing an area of interest, i. e., WRP sites, into smaller, within-site sampling sub-units, i.e., quadrats, and randomly selecting the

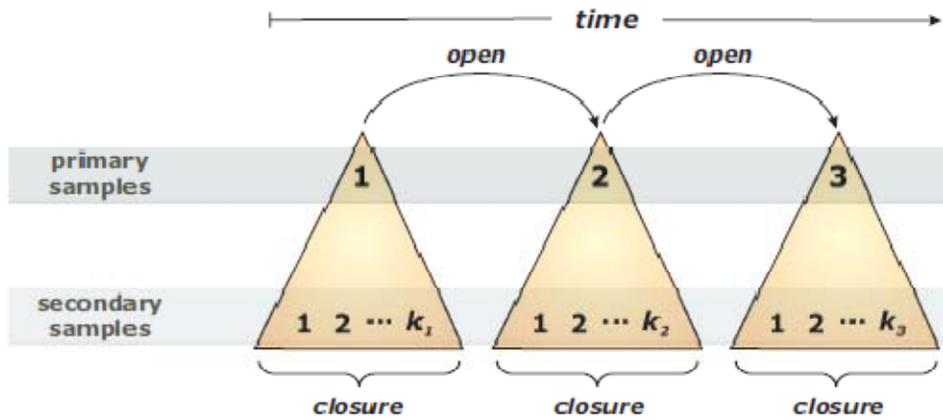


Figure 7. Schematic diagram illustrating Pollock's (1982) robust design. Populations are open between primary samples representing one temporal scale and are closed during secondary samples representing a second temporal scale. Taken from Cooch and White (2009) MARK: A Gentle Introduction Ch 15.

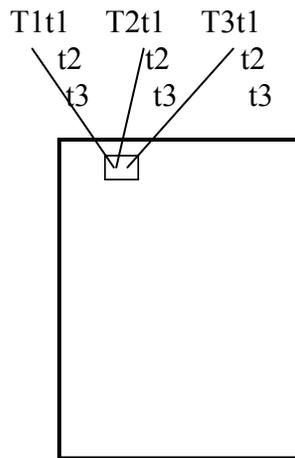


Figure 8. Stylized diagram illustrating how Pollock's (1982) robust design was applied to amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. The large rectangle represents a site and the smaller square represents a quadrat, the within site, sub-unit. Each quadrat was sampled in each of three primary survey periods (T1, T2, and T3) representing 1 temporal scale and, within each quadrat, three detection methods were used (t1, t2, and t3) representing a second temporal scale. Each site had nine to 11 quadrats; detection/nondetection information from all quadrats was collapsed by detection method to create a site-level detection history. Modified from Nichols et al. (1998).

quadrats for sampling. The study design then becomes a spatial analog of Pollock's (1982) robust design as it involves sampling at 2 spatial scales rather than 2 temporal scales (Figure 9) (Nichols et al. 1998). The WRP sites represent the primary samples and the within-site quadrats represent the secondary samples. As recommended by Conroy and Nichols (1996), equal sampling effort and methods were applied to each sampled quadrat as described previously in the Methods section. The result was a list of species detected on each site with closure assumed among the quadrats.

Two methods, the occupancy method and the capture/recapture method, were used to estimate species richness and are described below.

Occupancy Method

Occupancy modeling provides the means to estimate the proportion of species from a regional species pool that occupy a single site. The proportional occupancy of a site by a member of the regional species pool is the same as the relative species richness parameter of Cam et al. (2000) (MacKenzie et al. 2005). I used detection/nondetection data generated from all three primary sampling periods as described previously, thereby, retaining a temporal element in the analysis; however, I structured it for a species richness analysis. This was accomplished by listing each of the 17 species included in the regional species pool as a row in the input data tables, replacing sites. The columns in the input table then became the detection/nondetection event of each species in a sampled quadrat. For instance, a detection history of 10000110000 00011101010 11011000000 from 11 quadrats on Site 1 for Species A indicates Species A was detected on Site 1 in quadrats one, six, and seven during primary sampling period one, in quadrats four, five six, eight, and ten during primary sampling period two, and in quadrats one,

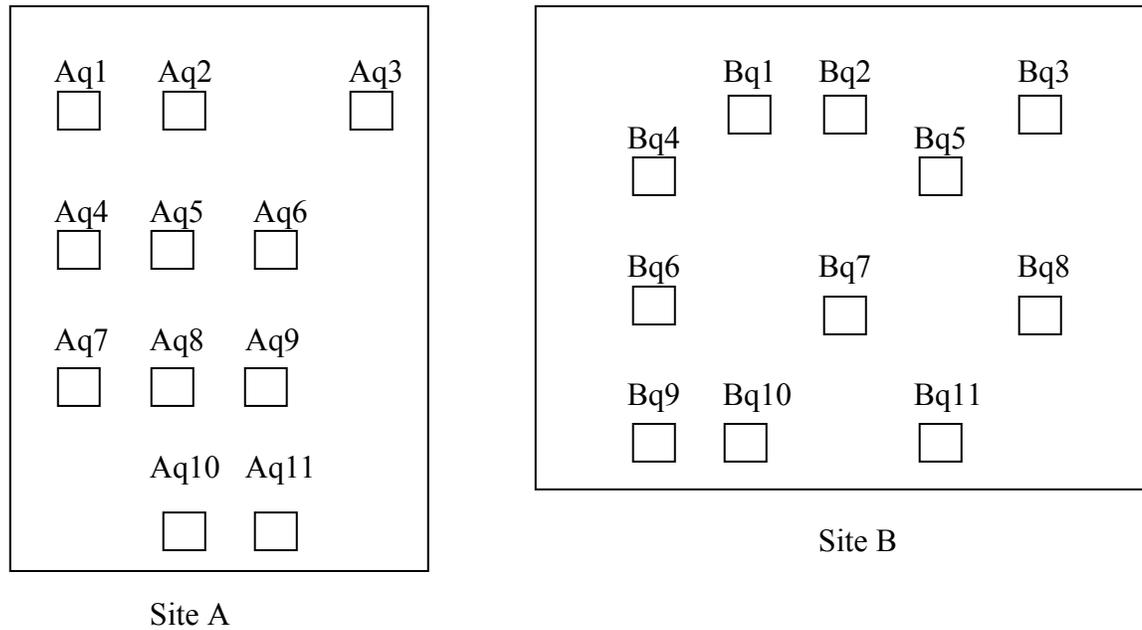


Figure 9. Stylized diagram illustrating how a spatial analog of Pollock's (1982) robust design was applied to amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. The large rectangles represent sites, the primary spatial scale, and the smaller squares represent quadrats, the secondary spatial scale. Sampling produced a species list from each quadrat and these lists were used to construct a detection/nondetection history of each detected species in each quadrat. The species list from the secondary samples (quadrats) was used to estimate species richness for the primary samples (site A and B in Figure 9). Each quadrat was sampled in each of three primary survey periods; however, as described in text, only primary survey period two was used for the capture/recapture method to estimate species richness whereas all three primary survey periods were used in the occupancy method to estimate relative species richness, thus, retaining a temporal element for the occupancy-based analysis. Modified from Nichols et al. (1998).

two, four, and five during primary sampling period three. A similar detection history is constructed for each species in the regional species pool for Site 1. The result is the proportional occupancy of a site or the probability that a member of the regional species pool is present at a site. Data used for the relative species richness analysis using the occupancy method is presented in a series of X-matrix tables in Appendix D. I used the single-species, single-season occupancy model described by MacKenzie et. al. (2002) and data analysis was conducted using program PRESENCE 2.4 (Hines 2006). The model set included one model in which the occupancy and detection parameters were constant ($\psi(\cdot)$, $p(\cdot)$). All 17 species from the regional species pool were included in the analysis for all 50 sampled sites. Given that, generally, each quadrat within a site was sampled in each of three primary sampling periods, the occupancy species richness method retains a temporal element in the analysis. Any exceptions to a quadrat being sampled three times are noted in Appendix D. Additionally, a truncated data set as described in the Data Analysis-Occupancy section was used to ensure community closure.

Capture/Recapture Method

For the species richness analysis using the capture/recapture method, I only used data generated during the second primary sampling period (14 May 2007-9 July 2007, inclusively) to ensure I met the community closure assumption (Conroy and Nichols 1996) and to avoid issues caused by missing values present in the truncated data set used for the previous occupancy analyses. I chose primary sampling period two as all species were present during this period. Again, this is a site-by-site analysis with a quadrat-based sample design; therefore, the study design can be viewed as a spatial analog of Pollock's

(1982) robust design as it involves sampling at two spatial scales rather than two temporal scales (Nichols et al. 1998) with WRP sites representing the primary samples and the within-site quadrats representing the secondary samples. Equal sampling effort and methods were applied to each sampled quadrat as described previously in the Methods section. The result was a list of species detected on each site with closure assumed among the quadrats. The data used to estimate species richness of each sampled WRP site using the capture/recapture method is presented in Appendix D and are the detection histories from primary sampling period two.

I used program CAPTURE to generate species richness estimates for each of the 50 sampled WRP sites based on detection/nondetection information from primary sampling period two (Rexstad and Burnham 1991, Nichols and Conroy 1996, Boulinier et al. 1998, Nichols et al. 1998). Relative species richness estimates were then calculated by computing the ratio of the species richness estimates to the number of species in the regional species pool. Eight models are available in program CAPTURE and vary based on their assumptions regarding sources of variation in detection probability (Table 3) (Boulinier et al. 1998, Kéry and Plattner 2007). The program includes a model selection procedure; based upon this procedure, I used species richness estimates from the most supported model that provided reasonable estimates.

Cluster Analysis

I hypothesized that design strategy would serve as a useful criterion of wetland restoration efforts with each category indicating if hydrological wetland characteristics had been restored to a site. Cluster analysis provided a means whereby I could test the similarity and dissimilarity of sampled WRP sites based on measured environmental

Table 3. A list of eight models available in Program CAPTURE to estimate species richness for a site and the associated assumptions regarding sources of variation in detection probability for each model.

Model	Detection probability
M_0	assumes all species have equal detection probability
M_t	assumes equal detection probability within a sample unit but that detection probability varies between sample units
M_b	assumes detection probability varies by behavioral response to capture, i.e., an individual develops either a positive response to capture (becomes trap-happy) or a negative response to capture (becomes trap-shy)
M_{tb}	assumes detection probability varies by both sampling unit and behavior
M_h	assumes detection probability varies among species but is constant within a sampling unit
M_{th}	assumes detection probabilities varies by species and sampling unit
M_{bh}	assumes detection probability varies by individual species and by behavior response to capture
M_{tbh}	assumes detection probability varies by species, sampling unit, and behavior response to capture

variables and determine if design strategy was a useful descriptor of the ecological attributes on WRP sites after completion of wetland restoration efforts (Objective 5). Cluster analysis is a descriptive, exploratory data analysis tool that moves objects between clusters while trying to minimize variability within clusters and maximize variability between clusters. I used a k-means cluster analysis which required the number of clusters be identified. Initially, I identified three clusters and used the overall averages from all three primary sampling periods using the following variables: 1) size of site (ha), 2) average proportion of sampled quadrats within a site that were dry, 3) average proportion of sampled quadrats within a site that were open water, 4) average proportion of sampled quadrats within a site that were wet with grass-like vegetation, and 5) average water depth within sampled quadrats on a site. I chose the environmental variables dry, wet with grass-like vegetation, and open water assuming they represented ephemeral, seasonal and permanent wetland conditions, respectively.

After the first analysis, I dropped size of site as a variable as six large sites were dominating the analysis and diluting the ability to discern if a difference existed based on the remaining variables. I then ran the analysis again using three clusters and the remaining four variables described previously. I also ran three additional analyses for each primary sampling period using site-level averages for the same four variables for each occasion and again identifying three clusters. I included 49 of the 50 sampled WRP sites in this analysis; the missing values associated with w18, the only site we did not visit in primary sampling period one, were problematic so w18 was dropped from the primary sampling period analysis. The analysis was performed in SPSS Statistics version 17 (SPSS, Inc Chicago, IL) that provides an ANOVA table with mean square error values

and F-tests for each variable. The mean square error is one metric used to determine if a variable is contributing toward differences between clusters as the higher the mean square error value, the less influential the variable. The F-tests also indicate how well each variable helps to discriminate between clusters. The F-tests cannot be used as significance tests as they are chosen to maximize the differences among cases in different clusters, thus, violating the assumptions for significance testing (SPSS, Inc Chicago, IL). Their usefulness is in a descriptive and exploratory fashion. The larger the F-test value, the more dominant the variable and the more it contributes toward differences between clusters. The smaller the F-test value, the less the variable is contributing.

RESULTS

A total of 30,995 individual amphibians were detected on the 50 sampled WRP sites in the Lower Grand River basin during the 2007 field season ranging from only seven central newts to nearly 16,000 individuals classified as leopard frog complex (Table 4). Ten of 17 species identified in the regional species pool (Table 2) were detected and included three of three species categorized as abundant and likely, five of five species categorized as common and likely, one of three species, (spring peeper), categorized as infrequent but likely, and one of four species, (central newt), categorized as rare and unlikely. Neither of the two species categorized as rare but likely was detected. This represents 59% of the regional species pool resulting in a naïve rating of good for wetland restoration efforts on WRP sites, collectively, within the Lower Grand River basin. A naïve rating is based on raw count data and does not account for detectability.

Table 4. Total number of amphibians detected by species and life history stage during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

Species	Adult	Larva	Metamorph	Juvenile	Total
Central newt	7	0	0	0	7
Small-mouthed salamander	80	588	24	5	697
American toad	2	178	54	24	258
Northern cricket frog	829	823	96	4,862	6,610
Grey treefrog	30	1,565	65	10	1,670
Spring peeper	3	2	8	2	15
<i>Pseudacris</i> spp.	31	1,009	49	9	1,098
Plains leopard frog	21	0	31	61	113
American bullfrog	102	3,150	49	1,130	4,431
Southern leopard frog	24	0	206	97	327
Leopard frog complex	185	8,106	1,927	5,551	15,769
Total	1,276	15,421	2,509	11,782	30,995

Metamorphs were detected for nine of the 10 species; no central newt metamorphs were detected during this study. Metamorphs were detected on 44 of 50 or 88% of surveyed WRP sites in the Lower Grand River basin; number of species with metamorphs detected per site ranged from zero to eight ($\bar{X} = 2.5$, $sd=1.7$). Metamorphs were detected on 16 of 18 or 89% of maximize hydrology sites, on 19 of 19 or 100% of naturalistic sites, and on 9 of 13 or 69% of walk-aways sites.

A total of 49,089 fishes were detected on the 50 sampled WRP sites in the Lower Grand River basin during the 2007 field season. Members of the Centrarchidae family dominated collections (28,676 collected), followed by Poeciliidae (11,595), Ictaluridae (4,058), Cyprinidae (3,549), Lepisosteidae (343), and 862 fishes classified as other (Figure 10). Ninety-nine percent of the Centrarchid detections were green sunfish (*Lepomis cyanellus*) with crappie (*Pomoxis* spp.), largemouth bass (*Micropterus salmoides*), and orange-spotted sunfish (*L. humilis*) detections comprising the remaining one percent. Ninety-one percent of green sunfish detections were young-of-year fish as indicated by size (Figure 11).

Approximately 74.6 cm of rainfall occurred throughout the 2007 field season (7 March-19 September) (Station 230980-Brookfield, MO) (Table 5). This compares to an average over the past 10 years (1997-2007) for the same time period (March-September) of 73.2 cm (SD=12.0). A rain event of 11.6 cm on 6 May 2007 resulted in a near-record flood event occurring in the Lower Grand River floodplain during the period 7 May through 14 May 2007 (USDI 2010) (Figure 12). A crest of 37.14 feet was recorded at the Grand River at Chillicothe gage on 10 May, the 3rd highest crest in recorded history at the

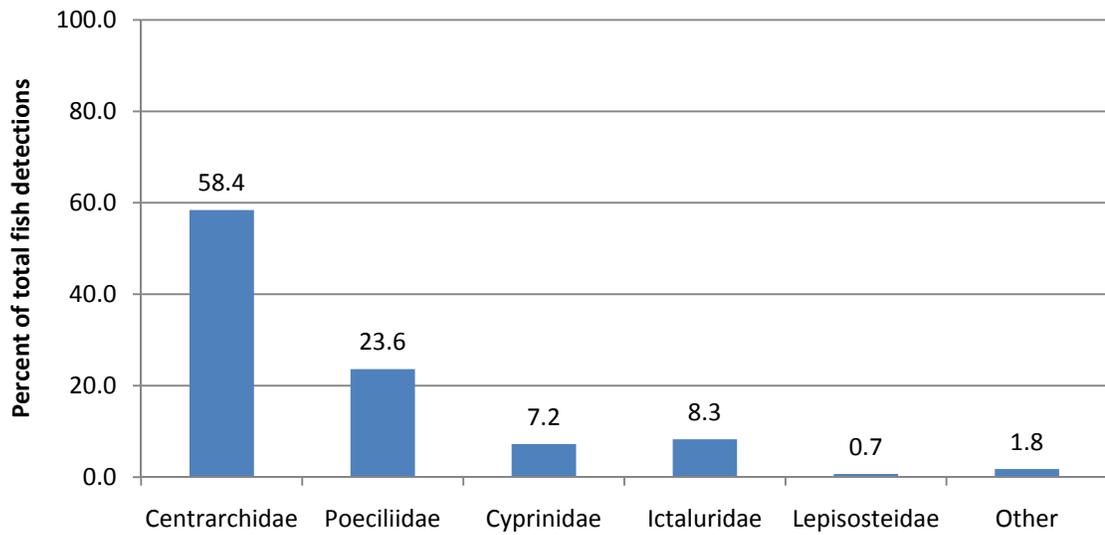


Figure 10. Percent of total fish detections by Family during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

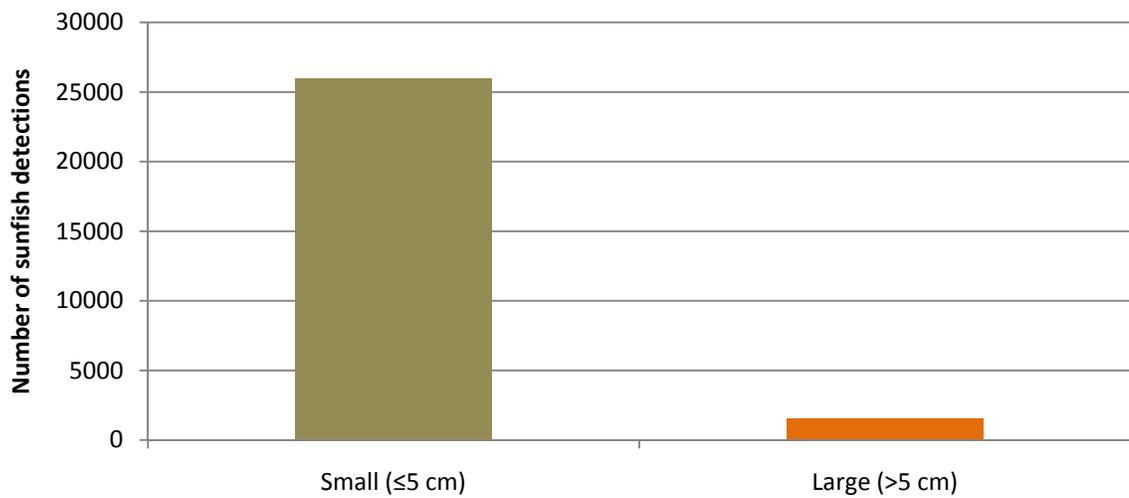


Figure 11. Number of green sunfish detections classified by size during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

Table 5. Monthly amounts of precipitation (cm) recorded at station 230980 Brookfield, Missouri for the time period March through September 1997-2007.

Year	March	April	May	June	July	August	September	Total
1997	5.1	16.2	9.2	8.6	3.1	11.7	6.9	61.8
1998	13.4	12.1	9.1	23.6	13.3	5.1	14.2	92.2
1999	6.7	14.9	12.4	7.7	3.9	3.5	16.6	66.6
2000	6.3	2.6	5.5	14.8	16.3	15.7	8.2	70.5
2001	8.0	10.3	14.6	19.2	9.8	13.1	10.2	86.3
2002	2.9	8.7	28.4	6.3	13.9	9.5	1.9	72.7
2003	2.9	14.5	9.8	8.2	4.5	10.2	17.7	68.9
2004	12.3	5.0	19.2	11.0	8.3	27.3	4.9	89.3
2005	2.7	6.1	5.9	14.4	3.2	10.6	8.6	52.2
2006	12.1	7.7	8.3	12.5	12.5	11.5	4.9	70.6
2007	7.3	9.3	15.7	17.2	2.3	16.4	5.3	74.6
Average	7.2	9.7	12.6	13.0	6.3	12.2	9.0	73.2

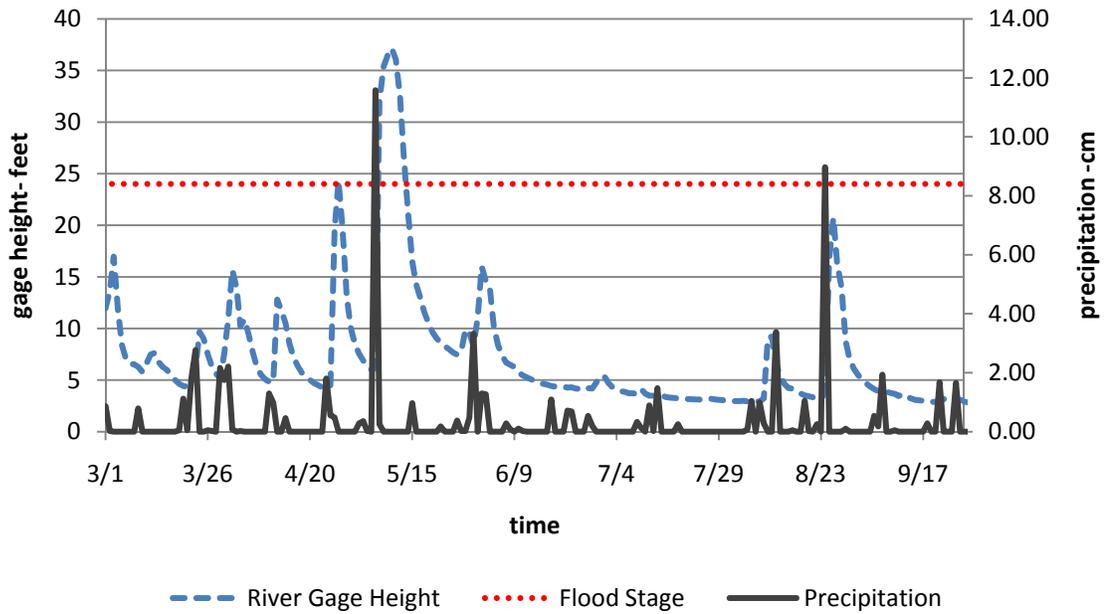


Figure 12. Daily gage heights recorded at Grand River at Chillicothe gage 1 March – 30 September 2007 and daily precipitation amounts recorded from 1 March – 30 September 2007 at station 230980 Brookfield, Missouri during the 2007 field season (March – September) of the amphibian occupancy and species richness study conducted in Lower Grand River basin, north-central Missouri.

Chillicothe gage (Figure 12) (NWS 2010). The 2007 field season included a warmer than average March followed by much cooler temperatures in early April (NWS 2009); otherwise, air temperatures followed the expected general trend of average monthly temperatures that were lower early in the season, gradually increased through the middle of the season, and gradually decreased late in the season (Figure 13).

Amphibian Distribution

The first objective of my study was to evaluate if a range of hydrological wetland characteristics as indicated by amphibian distribution had been restored to maximum hydrology, naturalistic, and walk-away sites in the Lower Grand River basin. This was accomplished by determining occupancy probability estimates using design strategy as a site covariate for each species from the regional species pool detected on WRP sites within the Lower Grand River basin. Design strategy did not contribute toward explaining variation in the proportion of area occupied by amphibian species on WRP sites within the Lower Grand River basin. Model 1 $\psi(\cdot)$, the null model that indicates no covariate effect, was the most supported model for eight of the 10 species included in the site occupancy analysis, and, of the two species in which the most supported model was different, convergence and bounding issues reduced those models to the null (Table 6). All evidence points toward design strategy as a poor explanatory variable including $AIC_c/QAIC_c$ delta values within each candidate set of models, evidence ratios between the most supported and least supported model within a candidate set, and odds ratios. The delta values between the most supported and least supported occupancy models with design strategy as a covariate were <5 for eight of the 10 set of candidate models and <10 for the remaining two, again indicating that design strategy is not a strong explanatory

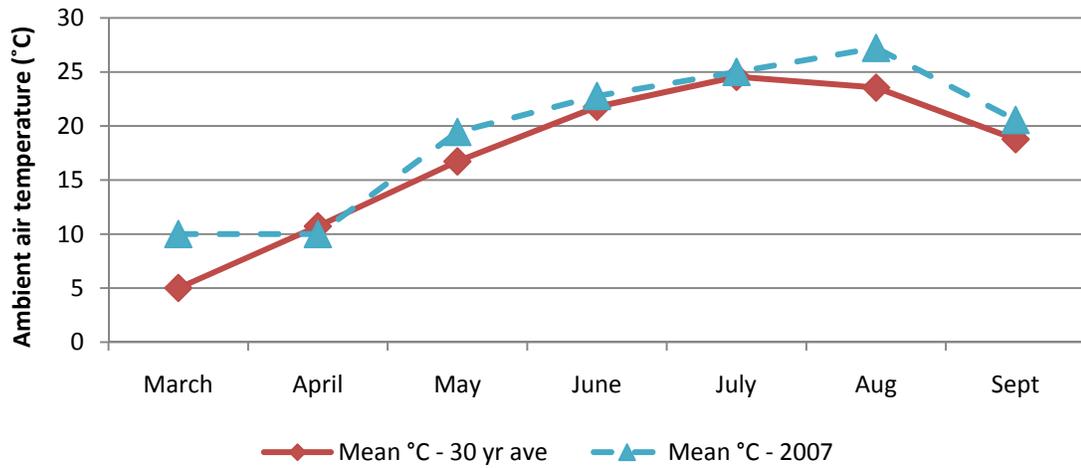


Figure 13. Average monthly ambient air temperatures in degrees Celsius for period March - September 1971-2000 compared to monthly average ambient air temperatures in degrees Celsius that occurred during the 2007 field season (March – September) of the amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

Table 6. Ranking of occupancy models that assessed the effect of design strategy (W=walk-away, M=maximize hydrology, and N=naturalistic) on occupancy probability (ψ), assuming detection probability (p) was constant, for species from the regional species pool detected on Wetlands Reserve Program sites in Lower Grand River basin, north-central Missouri during 2007 field season. AIC_c is Akaike's information criterion adjusted for small sample size, $QAIC_c$ is Akaike's information criterion adjusted for overdispersion ($\hat{c} > 1$) and small sample size, \hat{c} is variance inflation factor, Δ_i is the difference in $AIC_c/QAIC_c$ values from the top ranked model, w_i is the Akaike weight, model likelihood ($\exp(-\frac{1}{2}\Delta_i)$) is a relative measure of the model, given the data, as being the most likely model among the candidate set of models, K is the number of estimated parameters, and $-2LL$ is $-2 * \log$ -likelihood. GOF indicates the model used to run a goodness-of-fit test. Models with inestimable standard errors are designated with an *.

Model	$AIC_c/QAIC_c$	\hat{c}	Δ_i	w_i	Model Likelihood	K	$-2*LL$
Small-mouthed salamander							
$\Psi (.),p(.)$	148.53		0.00	0.44	1.00	3	274.12
$\Psi (W MN),p(.)$	150.27		1.74	0.19	0.42	4	273.13
$\Psi (M WN),p(.)$	150.47		1.94	0.17	0.38	4	273.51
$\Psi (N WM),p(.)$	150.79		2.26	0.14	0.32	4	274.11
$\Psi (W M N),p(.)$ GOF	152.57	1.90	4.04	0.06	0.13	5	273.00
American toad							
$\Psi (.),p(.)$	155.02		0.00	0.37	1.00	2	150.76
$\Psi (N WM),p(.)$	156.00		0.98	0.22	0.61	3	149.48
$\Psi (M WN),p(.)$ GOF	156.06	0.13	1.04	0.22	0.59	3	149.54
$\Psi (W MN),p(.)$	157.26		2.24	0.12	0.33	3	150.74
$\Psi (W M N),p(.)$	158.01		2.99	0.08	0.22	4	149.12
Northern cricket frog							
$\Psi (W MN), p(.)^*$	270.34		0.00	0.51	1.00	4	580.4
$\Psi (W M N), p(.)^*$	272.71		2.37	0.16	0.31	5	580.4
$\Psi (.), p(.)$ GOF	272.99	2.20	2.65	0.14	0.27	3	591.21
$\Psi (N WM), p(.)$	273.65		3.31	0.10	0.19	4	587.69
$\Psi (M WN), p(.)$	273.77		3.43	0.09	0.18	4	587.94
Grey treefrog complex							
$\Psi (.),p(.)$	243.47		0.00	0.38	1.00	3	334.9
$\Psi (W MN),p(.)$	244.34		0.87	0.25	0.65	4	332.95
$\Psi (N WM),p(.)$	245.17		1.70	0.16	0.43	4	334.11
$\Psi (M WN),p(.)$	245.58		2.11	0.13	0.35	4	334.68
$\Psi (W M N),p(.)$ GOF	246.69	1.40	3.22	0.08	0.20	5	332.92
Spring peeper							
$\Psi (.),p(.)$	74.57		0.00	0.38	1.00	3	91.41
$\Psi (N WM),p(.)$	75.64		1.07	0.23	0.59	4	89.85

Table 6. (continued)

Model	AIC _c /QAIC _c	\hat{c}	Δ_i	w_i	Model Likelihood	K	-2*LL
Ψ (M WN),p(.)	75.98		1.41	0.19	0.49	4	90.30
Ψ (W MN),p(.)	76.78		2.21	0.13	0.33	4	91.33
Ψ (W M N),p(.) GOF	77.89	1.30	3.32	0.07	0.19	5	89.70
<i>Pseudacris</i> sp.							
Ψ (.),p(.)	328.35		0.00	0.42	1.00	2	324.09
Ψ (N WM),p(.)	330.14		1.79	0.17	0.41	3	323.62
Ψ (W MN),p(.)	330.28		1.93	0.16	0.38	3	323.76
Ψ (M WN),p(.)	330.55		2.20	0.14	0.33	3	324.03
Ψ (W M N),p(.) GOF	332.42	0.89	4.07	0.05	0.13	4	323.53
Plains leopard frog							
Ψ (W M N),p(.)*	355.73		0.00	0.57	1.00	4	346.84
Ψ (M WN),p(.)*	357.7		1.97	0.21	0.37	3	351.18
Ψ (.),p(.)	358.82		3.09	0.12	0.21	2	354.56
Ψ (W MN),p(.) GOF	359.61	0.85	3.88	0.08	0.14	3	353.09
Ψ (N WM),p(.)	362.57		6.84	0.02	0.03	3	356.05
American bullfrog							
Ψ (.),p(.)	236.45		0.00	0.32	1.00	3	534.04
Ψ (W MN),p(.)	236.83		0.38	0.26	0.83	4	529.7
Ψ M WN),p(.)	237.25		0.8	0.21	0.67	4	530.68
Ψ (N WM),p(.)	238.67		2.22	0.10	0.33	4	533.95
Ψ (W M N),p(.) GOF	238.74	2.30	2.29	0.10	0.32	5	528.65
Southern leopard frog							
Ψ (.),p(.)	341.16		0.00	0.73	1.00	2	440.80
Ψ (W MN),p(.)	345.57		4.41	0.08	0.11	4	440.76
Ψ (N WM),p(.)	345.60		4.44	0.08	0.11	4	440.80
Ψ (M WN),p(.)	345.60		4.44	0.08	0.11	4	440.80
Ψ (W M N),p(.) GOF	347.94	1.30	6.78	0.02	0.03	5	440.76
Leopard frog complex							
Ψ (.),p(.)	150.58		0.00	0.39	1.00	3	614.56
Ψ (W MN),p(.)	151.62		1.04	0.23	0.59	4	609.43
Ψ (N WM),p(.)	152.44		1.86	0.15	0.39	4	612.85
Ψ (M WN),p(.)	152.47		1.89	0.15	0.39	4	612.97
Ψ (W M N),p(.) GOF	153.99	4.20	3.41	0.07	0.18	5	609.43

variable. Evidence ratios also indicated lack of support for design strategy as a strong explanatory covariate as only Plains leopard frog had an evidence ratio in double digits ($w_1/w_5=30.5$) and a minimum of ~ 150 is required to provide strong evidence in favor of a particular model within a candidate set of models. Odds ratios, calculated when appropriate, also indicated no effect of the covariate in all situations except one; the one exception occurred with the bullfrog in which it appears there is a positive response by the species toward pooling maximize hydrology and naturalistic sites with a negative response toward walk-away sites.

Although design strategy was not a strong explanatory variable, the occupancy analysis provides information regarding amphibian species' distribution on WRP sites in the Lower Grand River basin as estimating the site occupancy of each species based on probabilistic sampling enables an estimate of the proportion of sites in which a species occurs (Table 7). In all instances the naïve estimates were less than the real parameter estimates, indicating that not accounting for detectability resulted in negatively biased estimates (Table 7). Naïve estimates were calculated by dividing the number of sites at which a species was detected at least once by the total number of sites surveyed and does not account for detectability. In this analysis, the real parameter is the occupancy estimate that accounts for detectability. However, three species or groups; northern cricket frog, American bullfrog, and leopard frog complex; had nearly identical naïve and real parameter occupancy estimates (Table 7) suggesting that, if they were present, their detection probabilities were large enough that they were detected. Seven species or groups; American toad, northern cricket frog, *Pseudacris* spp., Plains leopard frog, American bullfrog, southern leopard frog, and leopard frog complex; had occupancy

Table 7. Naïve estimates ($\tilde{\psi}$) and real parameter estimates ($\hat{\psi}$) with associated standard errors (se) and 95% confidence intervals of proportion of area, or sites, occupied for species detected from the regional species pool on 50 sampled Wetlands Reserve Program sites during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. Naïve estimates are calculated by dividing the number of sites where a species was detected at least once by the total number of sites surveyed, without accounting for detectability. Real parameter is an estimate of occupancy that accounts for detectability. An * indicates the parameter hit the bounds of the maximum likelihood and the standard error could not be estimated.

Species	$\tilde{\psi}$	$\hat{\psi}$	se	95% CI	
				lower	upper
Small-mouthed salamander	0.58	0.66	0.08	0.48	0.70
American toad	0.36	0.85	0.32	0.04	1.00
Northern cricket frog	0.92	0.93	0.04	0.81	0.98
Grey treefrog complex	0.66	0.78	0.08	0.58	0.90
Spring peeper	0.18	0.39	0.22	0.10	0.80
<i>Pseudacris</i> spp	0.76	0.94	0.09	0.37	1.00
Plains leopard frog	0.70	0.87	0.10	0.55	0.98
American bullfrog	0.82	0.83	0.05	0.69	0.91
Southern leopard frog	0.90	1.00	*	*	*
Leopard frog complex	0.96	0.97	0.03	0.84	0.99

estimates exceeding 0.80 indicating all are widely-distributed on WRP sites irrespective of design strategy (Table 7). Grey treefrogs and small-mouthed salamanders were not as widely-distributed as the previous seven species but had high enough occupancy estimates to indicate they were moderately-distributed on WRP sites. Spring peeper had a more limited distribution among WRP sites.

Goodness-of-fit tests on the most general model in each candidate set of models indicated overdispersion in the data for seven of the 10 species in the analysis including small-mouthed salamander, northern cricket frog, grey treefrog complex, spring peeper, American bullfrog, southern leopard frog, and leopard frog complex (Table 6). Overdispersion is generally related to violation of model assumptions. The two assumptions most likely violated during the 2007 field season were 1) the system was closed to changes in occupancy during the sampling period, and 2) independence assumption. More detailed results regarding the evidentiary information supporting the site occupancy data analysis is available in Appendix E.

Heterogeneity in Detection Probability Estimates

The second objective of my study was to determine if amphibian seasonal movements or varying vulnerability to detection methods contributed to heterogeneity in detection probability. This was accomplished by determining detection probability estimates using day of survey in both a linear and quadratic form and detection method as survey-covariates for each species from the regional species pool detected on WRP sites within the Lower Grand River basin. Varying vulnerability to detection methods contributed to heterogeneity in detection probability estimates for the small-mouthed salamander, American toad, northern cricket frog, Plains leopard frog, and American

bullfrog indicating that the probability of detecting these species differed based on the detection method used. Amphibian seasonal movements contributed to heterogeneity in detection probability estimates for the grey treefrog complex, *Pseudacris* spp., southern leopard frog, and leopard frog complex indicating that these species did exhibit seasonal movement patterns as the probability of detection increased over time to a peak, then declined.

Given that the most supported model for all species included in the occupancy analysis was the null, i. e., $\psi(\cdot)$, this model was paired with each model in the candidate set of models for each species in the detection analysis. Model 4 $\psi(\cdot)$, $p(\text{method})$, the model that evaluated if there was a difference in detection probability based on detection method, was the most supported model for five of the 10 species in this analysis including the small-mouthed salamander, American toad, northern cricket frog, Plains leopard frog, and American bullfrog (Table 8). Detection probabilities varied by species and detection method with visual encounter survey (VES) having the highest detection probability estimates for American toads and northern cricket frogs (Table 9). Northern cricket frogs were twice as likely to be detected by VES than either dip-nets or aquatic funnel traps which had similar detection probability estimates (Table 9). There is substantial model selection uncertainty in the toad model set likely due to few detections resulting in a sparse data set; therefore, although VES appears the most effective method for detecting toads, overlapping confidence intervals indicate all three methods are equally effective (Table 9). Traps and nets had similar effectiveness at detecting small-mouthed salamanders, whereas VES was not an effective detection strategy for this species (Table 9). VES and traps exhibited similar detection probability estimates for

Table 8. Ranking of detection models that assessed the effect of sampling day (day and day sq) and detection method (method) on detection probability (p), assuming occupancy probability (ψ) was constant, for species from the regional species pool detected on Wetlands Reserve Program sites in Lower Grand River basin, north-central Missouri during 2007 field season. AIC_c is Akaike's information criterion adjusted for small sample size, $QAIC_c$ is Akaike's information criterion adjusted for overdispersion (if $\hat{c} > 1$) and small sample size, \hat{c} is variance inflation factor, Δ_i is the difference in $AIC_c/QAIC_c$ values from the top ranked model, w_i is the Akaike weight, model likelihood ($\exp(-\frac{1}{2}\Delta_i)$) is a relative measure of the model, given the data, as being the most likely model among the candidate set of models, K is the number of estimated parameters, and $-2LL$ is $-2*\log$ -likelihood. GOF indicates the model used to run a goodness-of-fit test

Model	$AIC_c/QAIC_c$	\hat{c}	Δ_i	w_i	Model Likelihood	K	$-2LL$
Small-mouthed salamander							
$\Psi(\cdot), p(\text{method})$ GOF	222.27	1.10	0.00	0.9759	1.00	5	234.72
$\Psi(\cdot), p(\text{day sq})$	229.67		7.40	0.0241	0.03	5	242.86
$\Psi(\cdot), p(\cdot)$	253.46		31.19	0.0000	0.00	3	274.12
$\Psi(\cdot), p(\text{day})$	255.72		33.45	0.0000	0.00	4	274.12
American toad							
$\Psi(\cdot), p(\text{method})$ GOF	151.7	0.16	0.00	0.6883	1.00	4	142.81
$\Psi(\cdot), p(\cdot)$	155.02		3.32	0.1309	0.19	2	150.76
$\Psi(\cdot), p(\text{day sq})$	155.27		3.57	0.1155	0.17	4	146.38
$\Psi(\cdot), p(\text{day})$	156.41		4.71	0.0653	0.09	3	149.89
Northern cricket frog							
$\Psi(\cdot), p(\text{method})$ GOF	411.7	1.30	0.00	0.9978	1.00	5	523.66
$\Psi(\cdot), p(\text{day sq})$	423.93		12.23	0.0022	0.00	5	539.55
$\Psi(\cdot), p(\text{day})$	437.23		25.53	0.0000	0.00	4	559.92
$\Psi(\cdot), p(\cdot)$	453.01		41.31	0.0000	0.00	3	583.38
Grey treefrog complex							
$\Psi(\cdot), p(\text{day sq})$	175.35		0.00	1.0000	1.00	5	233.04
$\Psi(\cdot), p(\cdot)$	243.47		68.12	0.0000	0.00	3	334.9
$\Psi(\cdot), p(\text{day})$	245.56		70.21	0.0000	0.00	4	334.66
$\Psi(\cdot), p(\text{method})$ GOF	245.85	1.40	70.50	0.0000	0.00	5	331.75
Spring peeper							
$\Psi(\cdot), p(\cdot)$	69.55		0.00	0.5428	1.00	3	91.41
$\Psi(\cdot), p(\text{day sq})$	71.36		1.81	0.2196	0.40	5	87.46
$\Psi(\cdot), p(\text{day})$	71.76		2.21	0.1798	0.33	4	91.33
$\Psi(\cdot), p(\text{method})$ GOF	74.03	1.40	4.48	0.0578	0.11	5	91.20

Table 8. (continued)

Model	AIC _c /QAIC _c	\hat{c}	Δ_i	w_i	Model Likelihood	K	-2LL
<i>Pseudacris</i> sp.							
$\Psi(\cdot),p(\text{day sq})$	280.94		0.00	1.0000	1.00	4	272.05
$\Psi(\cdot),p(\text{day})$	315.36		34.42	0.0000	0.00	3	308.84
$\Psi(\cdot),p(\text{method})$ GOF	323.23	0.83	42.29	0.0000	0.00	4	314.34
$\Psi(\cdot),p(\cdot)$	328.35		47.41	0.0000	0.00	2	324.09
Plains leopard frog							
$\Psi(\cdot),p(\text{method})$ GOF	339.25	0.55	0.00	0.9987	1.00	4	330.36
$\Psi(\cdot),p(\text{day sq})$	352.87		13.62	0.0011	0.00	4	343.98
$\Psi(\cdot),p(\text{day})$	356.64		17.39	0.0002	0.00	3	350.12
$\Psi(\cdot),p(\cdot)$	358.82		19.57	0.0001	0.00	2	354.56
American bullfrog							
$\Psi(\cdot),p(\text{method})$ GOF	268.74	1.90	0.00	0.6853	1.00	5	493.71
$\Psi(\cdot),p(\text{day})$	270.87		2.13	0.2363	0.34	4	502.26
$\Psi(\cdot),p(\text{day sq})$	273.08		4.34	0.0782	0.11	5	501.97
$\Psi(\cdot),p(\cdot)$	285.33		16.59	0.0002	0.00	3	534.04
Southern leopard frog							
$\Psi(\cdot),p(\text{day sq})$	421.56		0.00	0.9882	1.00	4	415.04
$\Psi(\cdot),p(\text{method})$ GOF	430.44	0.91	8.88	0.0117	0.01	4	423.92
$\Psi(\cdot),p(\text{day})$	440.62		19.06	0.0001	0.00	3	436.37
$\Psi(\cdot),p(\cdot)$	442.88		21.32	0.0000	0.00	2	440.80
Leopard frog complex							
$\Psi(\cdot),p(\text{day sq})$	149.56		0.00	0.9999	1.00	5	520.48
$\Psi(\cdot),p(\text{method})$ GOF	170.21	3.70	20.65	0.0000	0.00	5	596.89
$\Psi(\cdot),p(\cdot)$	170.35		20.79	0.0000	0.00	3	614.56
$\Psi(\cdot),p(\text{day})$	172.62		23.06	0.0000	0.00	4	614.56

Table 9. Detection probability (p), standard error (se), and 95% confidence interval estimates for species in which detection method was the most supported model among the candidate set of models for amphibian occupancy and species richness study conducted in Lower Grand River basin, north-central Missouri in 2007.

Species and detection method	p	se	95% CI	
			lower	upper
Small-mouthed salamander				
Aquatic funnel trap	0.53	0.03	0.39	0.67
Dip-net	0.43	0.07	0.30	0.56
Visual encounter survey	0.05	0.07	0.02	0.15
American toad				
Aquatic funnel trap	0.04	0.09	0.07	0.41
Dip-net	0.09	0.17	0.03	0.23
Visual encounter survey	0.19	0.37	0.01	0.15
Northern cricket frog				
Aquatic funnel trap	0.32	0.05	0.23	0.41
Dip-net	0.26	0.04	0.18	0.35
Visual encounter survey	0.69	0.05	0.60	0.78
Plains leopard frog				
Aquatic funnel trap	0.27	0.05	0.18	0.37
Dip-net	0.05	0.02	0.02	0.11
Visual encounter survey	0.21	0.04	0.14	0.30
American bullfrog				
Aquatic funnel trap	0.58	0.06	0.46	0.71
Dip-net	0.22	0.05	0.12	0.32
Visual encounter survey	0.55	0.06	0.42	0.67

both Plains leopard frogs and bullfrogs; nets were not very effective for either species (Table 9).

Model $p(\text{day sq})$, the model that evaluated day of survey as a quadratic function, was the most supported model for grey treefrogs, *Pseudacris* sp., southern leopard frog, and leopard frog complex (Table 8). This model reflects that detection probability varies temporally and among species likely due to differing seasonal behavioral patterns and responses to changing environmental conditions. Detection probability estimates for the grey treefrog complex peaked in the middle of June (Figure 14), for *Pseudacris* spp. in early May (Figure 15), for southern leopard frog in early July (Figure 16), and for leopard frog complex in mid-June (Figure 17). Although not the most supported model for all species, examination of the $p(\text{day sq})$ detection curves for all species are helpful as they clearly show the timeframe from April 1 –July 1 as the period in which the most species can be detected while still meeting the population closure assumption (Figure 18).

The most supported model for the spring peeper was the constant detection model, i. e., detection probability estimates did not vary temporally or by method (Table 8). There is substantial model selection uncertainty in the peeper candidate model set as evidenced by the delta QAIC values and evidence ratios.

Goodness-of-fit tests on the most general model in each candidate set of models indicated overdispersion in the data for six of the 10 species in the analysis (Table 8) including small-mouthed salamanders, northern cricket frogs, grey treefrog complex, spring peeper, American bullfrog and leopard frog complex. Again, overdispersion is generally related to violation of model assumptions. The most likely assumption violated as regards detection probability was the assumption of independence among individuals.

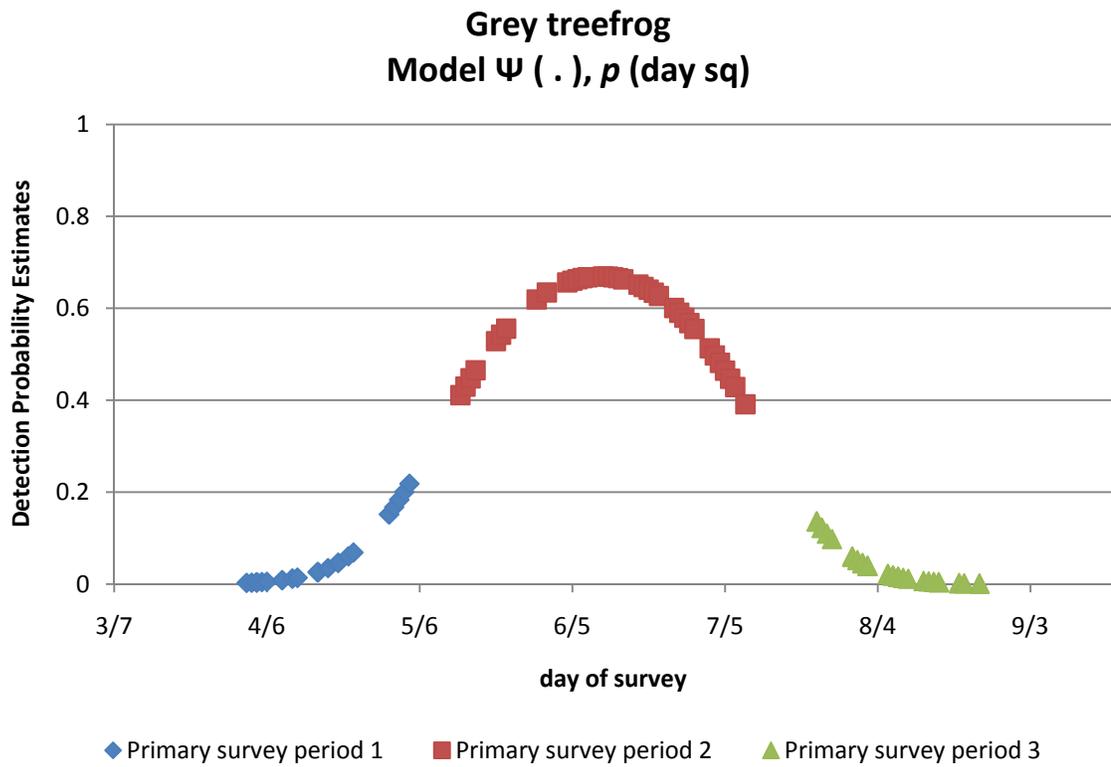


Figure 14. Detection probability estimates for the most supported model among the candidate set of models for grey treefrog complex for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

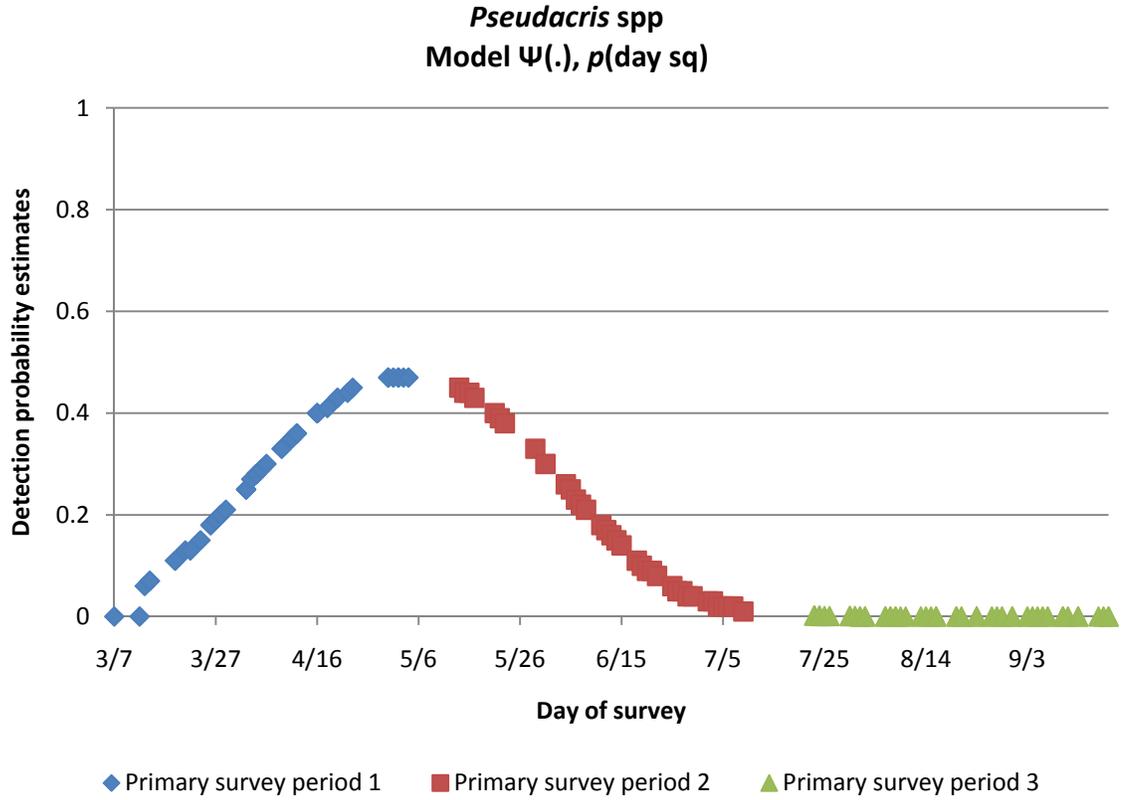


Figure 15. Detection probability estimates for the most supported model among the candidate set of models for *Pseudacris* spp. for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

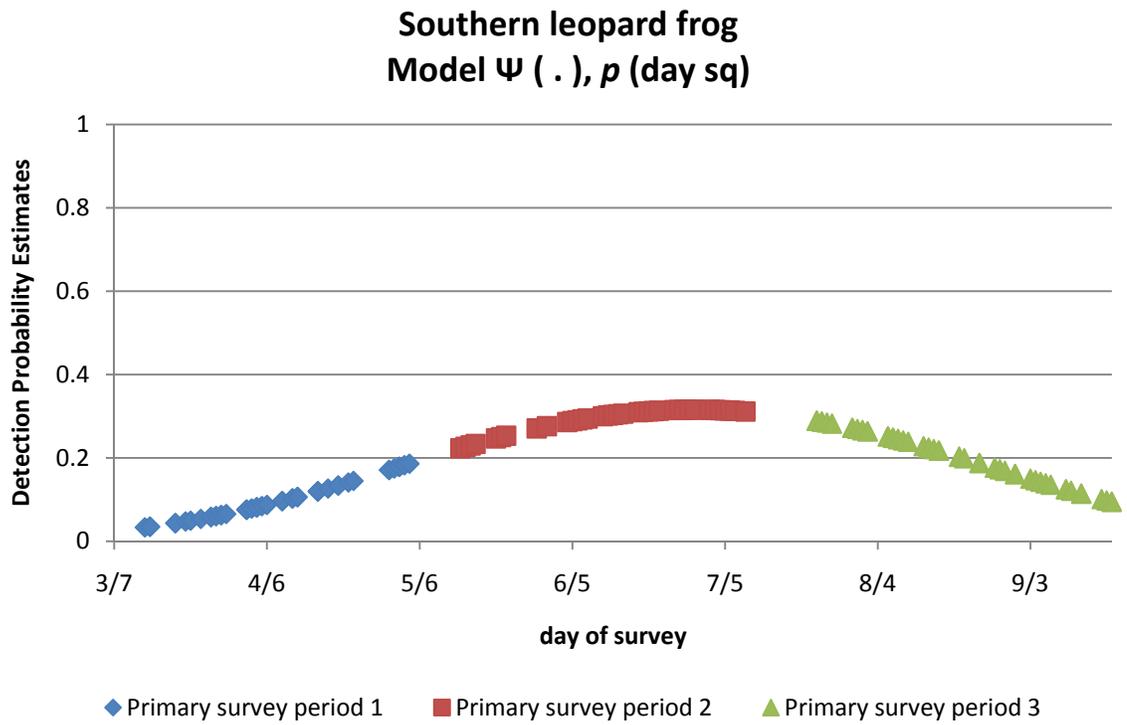


Figure 16. Detection probability estimates for the most supported model among the candidate set of models for southern leopard frog for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

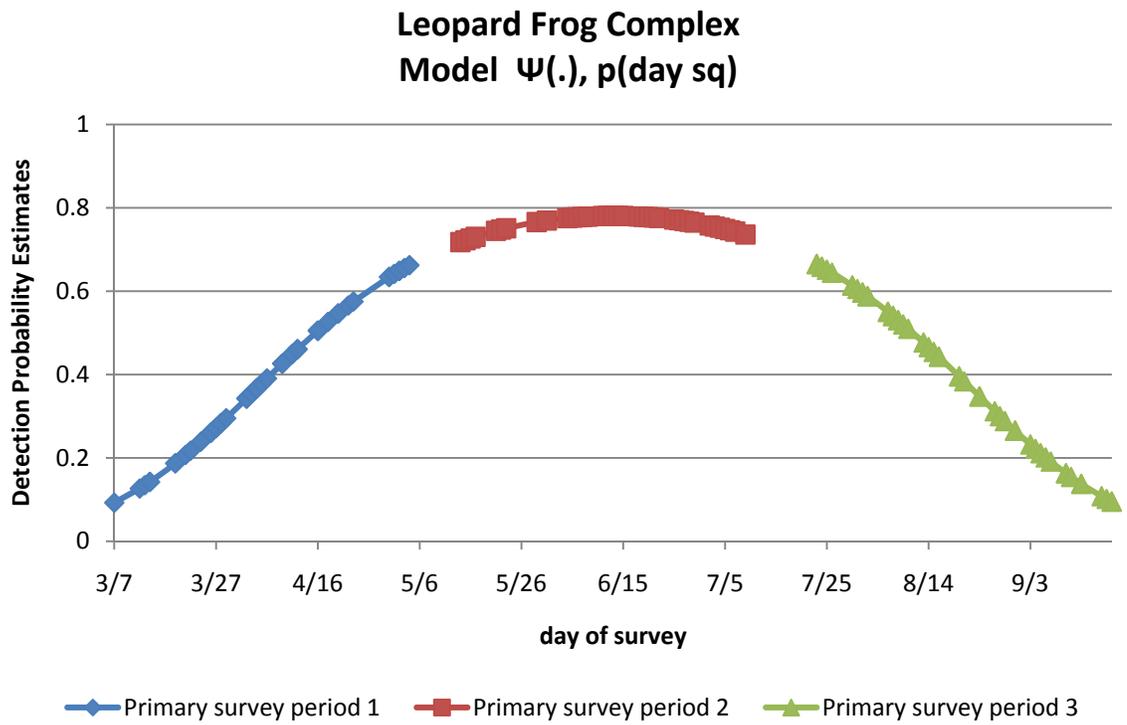


Figure 17. Detection probability estimates for the most supported model among the candidate set of models for leopard frog complex for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri.

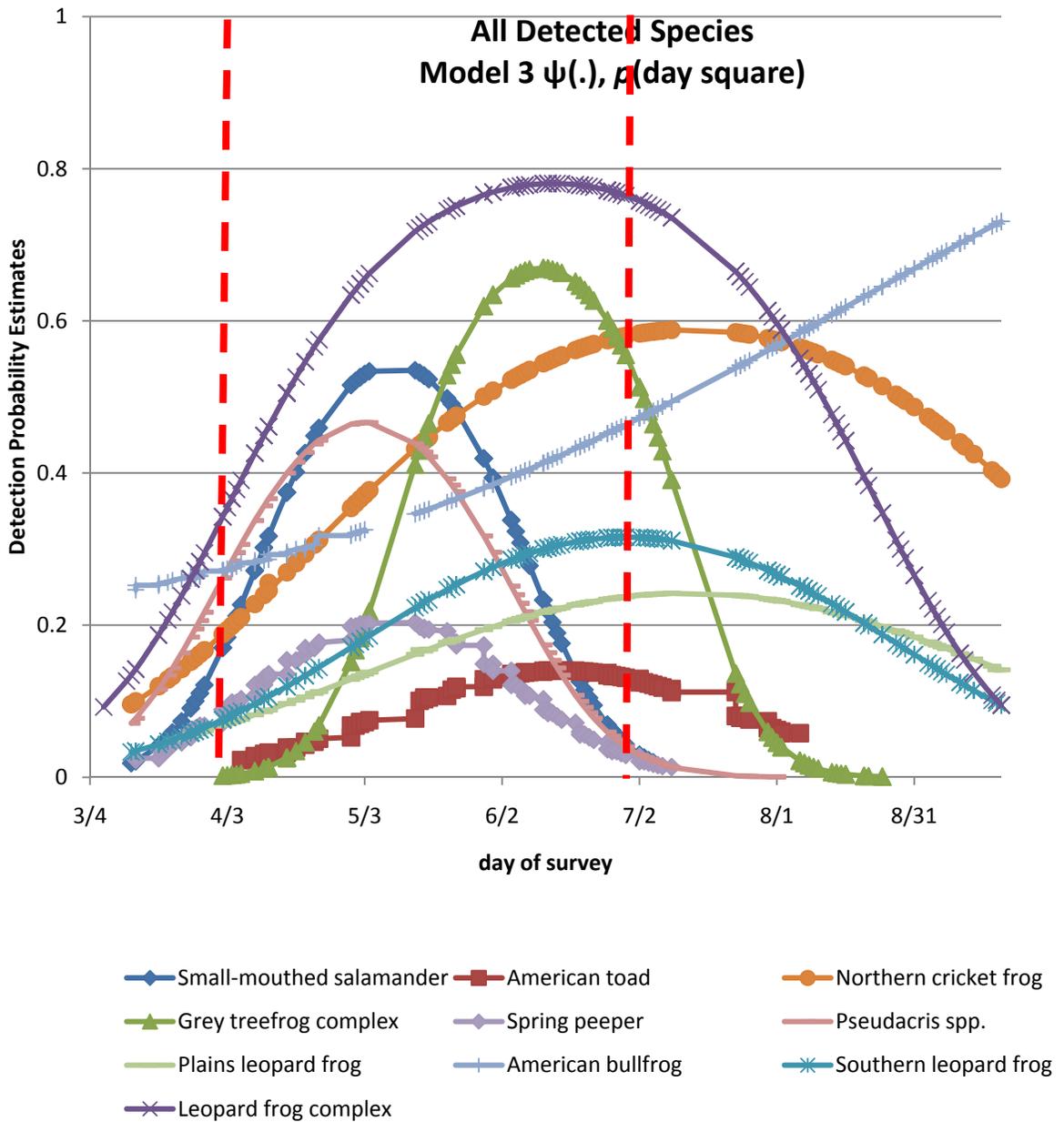


Figure 18. Detection probability estimates for Model 3 $\psi(\cdot), p(\text{method})$ for all species detected during 2007 field season (March – September) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. The dashed red vertical lines indicate the period of April 1 through July 1 as the time period in which the most species can be detected while still meeting the population closure assumption.

More detailed results regarding the evidentiary information supporting the detection probability analysis is presented in Appendix F.

Recruitment Success

My third objective dealt with evaluating if biological wetland characteristics had been restored on WRP properties in the Lower Grand River basin as indicated by amphibian recruitment. This was accomplished by using the multi-state occupancy model to estimate the overall probability that a species successfully recruited young into the adult population. Species included in this analysis exhibited variation in the overall probability of successful recruitment indicating that biological wetland characteristics had been restored for some species but were somewhat lacking for other species. Plains leopard frog, leopard frog complex, and southern leopard frogs ($\hat{\Psi}^1 * \hat{\Psi}^2 = 0.84, 0.94,$ and $0.99,$ respectively) had high estimates for overall probability that a species successfully recruited young into the adult population indicating that biological wetland characteristics had been restored for these species. The small-mouthed salamander had a moderately high estimate ($\hat{\Psi}^1 * \hat{\Psi}^2 = 0.61$), whereas, *Pseudacris* spp, cricket frogs and bullfrogs exhibited moderate estimates ($\hat{\Psi}^1 * \hat{\Psi}^2 = 0.53, 0.48$ and $0.49,$ respectively) indicating that biological wetland characteristics were somewhat lacking for small-mouthed salamanders to moderately lacking for the other three species. Grey treefrog complex had a low estimate ($\hat{\Psi}^1 * \hat{\Psi}^2 = 0.36$) of overall probability of successful recruitment indicating that biological wetland characteristics were lacking for this group. However, potential violation of the closure assumption for grey treefrogs may have confounded the estimate of overall recruitment success. American toad and spring peepers were not included in the analysis due to sparse data sets.

The most supported model for six of the eight species or groups included in this analysis was the model with delta, δ , i. e., the probability of observing evidence of recruitment given detection of occupancy, varying by primary sampling period (Table 10). The two species for which it was not the most supported model, northern cricket frog and American bullfrog, had δ (constant) as the most supported model after converting to QAIC_c based on overdispersion values >10. The model with delta varying by primary sampling period reflects the seasonal nature of amphibian breeding strategies and that detection of recruitment, or metamorphs, varies by species over time. Very few detections of metamorphs occurred during the first primary sampling period and few occurred during the third primary sampling period (northern cricket frog and bullfrogs were the exceptions with $\hat{\delta}_3 = 0.18$ and 0.12, respectively in the δ (period) model and $\hat{\delta} = 0.13$ and 0.18, respectively, in the $\delta(\cdot)$ model); the second primary sampling period had the highest probability of detecting successful recruitment for all species except northern cricket frog (Table 11). This supports results from the Detection section that the optimal time to survey amphibians in the Lower Grand River basin is 1 April – 1 July, particularly if detection of recruitment is a study objective.

All model-averaged estimates for individual species included in this analysis, i. e., proportion of sites occupied ($\hat{\Psi}^1$), probability that successful recruitment occurred on occupied sites ($\hat{\Psi}^2$), and overall probability of successful recruitment ($\hat{\Psi}^1 * \hat{\Psi}^2$), were higher than estimates provided by naïve estimates based on raw data (Table 12). This indicates that not accounting for detectability resulted in underestimation of occupancy parameters. The probability that successful recruitment occurred on an occupied site ($\hat{\Psi}^2$) was virtually 1.00 for both the Plains and southern leopard frogs, indicating either 1)

Table 10. Ranking of multi-state occupancy models that assessed the effect of primary survey period (period) and detection method (method) on the probability of detecting successful recruitment, given detection of occupancy (δ), assuming all other parameters were constant (occupancy probability (ψ^1), probability that successful recruitment occurred, given site is occupied (ψ^2), probability that occupancy is detected given true state of site =1 (p^1), and probability that occupancy is detected given true state of site = 2 (p^2)), for species from the regional species pool detected on Wetlands Reserve Program sites in Lower Grand River basin, north-central Missouri during 2007 field season. QAIC_c is Akaike's information criterion adjusted for overdispersion (if $\hat{c} > 1$) and small sample size, \hat{c} is variance inflation factor, Δ_i is the difference in QAIC_c values from the top ranked model, w_i is the Akaike weight, model likelihood ($\exp(-1/2\Delta_i)$) is a relative measure of the model, given the data, as being the most likely model among the candidate set of models, K is the number of estimated parameters, and $-2LL$ is $-2*\log$ -likelihood. Goodness-of-fit tests were run on the $\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (method) model for all candidate set of models.

Model	QAIC _c	\hat{c}	Δ_i	w_i	Model Likelihood	K	$-2LL$
Small-mouthed salamander							
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (period)	101.77		0.00	0.82	1.00	6	307.39
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (.)	105.55		3.78	0.12	0.15	6	320.25
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (method)	107.34	3.4	5.57	0.05	0.06	7	317.52
Northern cricket frog							
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (.)	78.81		0.00	0.68	1.00	6	674.51
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (period)	80.76		1.94	0.26	0.38	7	668.02
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (method)	83.50	10.5	4.68	0.07	0.10	8	668.31
Grey treefrog complex							
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (period)	135.08		0.00	0.66	1.00	6	408.28
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (.)	136.74		1.66	0.29	0.44	6	413.74
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (method)	139.93	3.3	4.84	0.058	0.09	8	406.76
<i>Pseudacris</i> spp.							
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (period)	85.65		0.00	0.62	1.00	5	371.42
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (method)	87.80	5.0	2.16	0.21	0.34	6	368.25
$\Psi^1(\cdot)$, $\Psi^2(\cdot)$, $p^1(\cdot)$, $p^2(\cdot)$, δ (.)	88.34		2.69	0.16	0.26	5	384.88

Table 10. (continued)

Model	QAICc	\hat{c}	Δ_i	w_i	Model Likelihood	K	-2LL
Plains leopard frog							
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (period)	74.92		0.00	0.83	1.00	5	409.39
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (.)	78.41		3.49	0.15	0.17	5	431.04
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (method)	82.25	6.2	7.33	0.02	0.03	7	423.45
American bullfrog							
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (.)	75.05		0.00	0.81	1.00	6	636.91
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (method)	79.06	10	4.00	0.11	0.13	8	623.96
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (period)	79.58		4.52	0.08	0.10	8	629.12
Southern leopard frog							
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (period)	60.31		0.00	0.98	1.00	4	505.65
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (.)	68.48		8.16	0.02	0.02	5	560.12
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (method)	72.17	9.4	11.85	0.00	0.00	7	547.19
Leopard frog complex							
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (period)	89.11		0	0.80	1.00	5	41.88
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (.)	92.05		2.94	0.18	0.23	5	44.82
$\Psi^1(\cdot), \Psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta$ (method)	96.94	10	7.53	0.02	0.02	7	44.11

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Table 11. Real parameter estimates for probability of detecting successful recruitment (δ) from model $\{\psi^1(\cdot), \psi^2(\cdot), p^1(\cdot), p^2(\cdot), \delta(\text{period})\}$ in which delta varied by primary survey period for species from the regional species pool detected on Wetlands Reserve Program sites in Lower Grand River basin, north-central Missouri during 2007 field season. An * indicates the parameter hit the bounds of the maximum likelihood and the standard error (se) could not be estimated.

Species	$\hat{\delta}_1$	\hat{se}	$\hat{\delta}_2$	\hat{se}	$\hat{\delta}_3$	\hat{se}
Small-mouthed salamander	0.00	*	0.38	0.24	n/a	n/a
Northern cricket frog	0.00	*	0.15	0.21	0.18	0.28
Grey treefrog complex	0.00	*	0.64	0.19	0.00	*
<i>Pseudacris</i> spp.	0.03	0.07	0.45	0.30	0.00	*
Plains leopard frog	0.00	*	0.57	0.23	0.08	0.14
American bullfrog	0.07	0.15	0.31	0.28	0.12	0.14
Southern leopard frog	0.00	*	0.81	0.17	0.08	0.17
Leopard frog complex	0.00	*	0.27	0.13	0.08	0.12

Table 12. Naïve estimates ($\tilde{\Psi}^1$) and model-averaged, real parameter estimates ($\hat{\Psi}^1$) for probability of occupancy, naïve estimates ($\tilde{\Psi}^2$) and model-averaged real parameter estimates ($\hat{\Psi}^2$) for probability of successful recruitment, given a site is occupied, and naïve estimates ($\tilde{\Psi}^1 * \tilde{\Psi}^2$) and model-averaged, derived parameter estimates ($\hat{\Psi}^1 * \hat{\Psi}^2$) for overall probability of successful recruitment for species from the regional species pool detected on Wetlands Reserve Program sites in Lower Grand River basin, north-central Missouri during 2007 field season.

Species	$\tilde{\Psi}^1$	$\hat{\Psi}^1$	$\tilde{\Psi}^2$	$\hat{\Psi}^2$	$\tilde{\Psi}^1 * \tilde{\Psi}^2$	$\hat{\Psi}^1 * \hat{\Psi}^2$
Small-mouthed salamander	0.58	0.70	0.24	0.87	0.14	0.61
Northern cricket frog	0.92	0.96	0.22	0.50	0.20	0.48
Grey treefrog complex	0.66	0.79	0.39	0.46	0.26	0.36
<i>Pseudacris</i> spp.	0.74	0.98	0.24	0.53	0.18	0.53
Plains leopard frog	0.70	0.99	0.46	0.95	0.32	0.84
American bullfrog	0.82	0.86	0.32	0.57	0.26	0.49
Southern leopard frog	0.90	1.00	0.60	1.00	0.54	0.99
Leopard frog complex	0.96	1.00	0.54	0.95	0.52	0.94

these species, where detected, are successfully recruiting young into the adult population and successfully exploit WRP sites and associated habitats or 2) $\hat{\psi}^2$ is overestimated for these species as young-of-year individuals could not be identified to species until they reached the metamorph stage. However, leopard frog complex also had an estimate of successful recruitment on occupied sites of 0.95 lending further support to the idea that leopard frogs are successfully recruiting young on occupied sites. Small-mouthed salamanders also exhibited a relatively high probability of successful recruitment ($\hat{\psi}^2 = 0.87$; $se=0.22$); however, the high variance associated with this estimate indicates caution should be taken in making too strong an inference. *Pseudacris* spp. had a moderate probability of successful recruitment on occupied sites ($\hat{\psi}^2 = 0.53$; $se=0.45$); although, again, the high standard error tempers the ability to make too strong an inference. Northern cricket frogs and grey treefrogs had similar estimates of successful recruitment on occupied sites ($\hat{\psi}^2 = 0.50$ and 0.46 , respectively) whereas bullfrogs had an estimate of successful recruitment on occupied sites of 0.57 . Again, all estimates have associated high standard errors, however, the moderate recruitment estimate for northern cricket frog ($\hat{\psi}^2 = 0.50$) is somewhat intriguing given the high occupancy estimate ($\hat{\psi}^1 = 0.96$) for this species.

Deviance-based goodness-of-fit tests on the most general model in each candidate set of models indicated overdispersion in the data for all species included in this analysis (Table 10). Although the lack of model fit is assumed to be related to overdispersion, it may also indicate inadequate model structure; however, given that no covariates are included in the models, I proceeded with the assumption that the \hat{c} values, or overdispersion, were related to violation of model assumptions. Higher \hat{c} values occurred

for the late-breeding species (northern cricket frog, Plains leopard frog, American bullfrog, southern leopard frog, and leopard frog complex) than for the early breeding species (small-mouthed salamander, grey treefrog complex, and *Pseudacris* spp.). This may reflect that deviance is a poor goodness-of-fit test statistic for occupancy models as the majority of the expected values were < 2 or it may reflect the fact that these species tended to congregate together more so than the early breeding species, and had a higher degree of dependency in their movements, thus, violating the assumption of independence.

More detailed results regarding the evidentiary information supporting the multi-state occupancy analysis is available in Appendix G.

Relative Species Richness Metric Estimates

My fourth objective dealt with assessing wetland restoration efforts through use of a relative species richness metric. Estimates for the relative species richness metric were determined using two methods: 1) occupancy method and 2) capture/recapture method. Results for these two methods are presented below. Once the relative species richness metric was determined, wetland restoration efforts on WRP sites in the Lower Grand River basin were rated as follows: sites with relative species richness estimates that represented ≤ 0.24 of the regional species pool were rated poor, sites with a relative species richness estimate of 0.25 – 0.49 were rated fair, sites with a relative species richness estimate of 0.50 – 0.69 were rated good, sites with a relative species richness estimate of 0.70 - 0.79 were rated very good, and sites with a relative species richness estimate ≥ 0.80 were rated excellent.

Occupancy Method

Results for the species richness analysis using the occupancy method included average relative species richness estimates for maximize hydrology sites of 0.37 (sd=0.09; CI=0.30–0.44), for naturalistic sites of 0.41 (sd=0.12; CI=0.36-0.46), and for walk-away sites of 0.28 (sd=0.15; CI=0.21-0.35). These estimates represent the average probability that a member of the regional species pool was present on a site. Applying the relative species richness metric resulted in all three category of design strategy receiving an overall rating of fair for wetland restoration efforts. Similar to results from the occupancy analysis, the relative species richness estimates calculated using this method had overlapping confidence intervals indicating no effect due to design strategy.

Relative species richness estimates for 18 maximize hydrology sites ranged from a low of 0.24 on sites m32 and m34 to a high of 0.51 on site m49 (Table 13). Relative species richness estimates for 19 naturalistic sites ranged from a low of 0.24 on sites n29 to a high of 0.72 on site n13 (Table 13). However, a high standard error (se=0.40) and the very wide confidence interval (Table 13) associated with site n13 requires that this estimate be viewed with skepticism; this is a case in which the naïve estimate of 0.35 may be a more accurate reflection of site conditions. Sparse data associated with walk-away sites w18 and w21 resulted in no estimate for these sites because of bounding and convergence issues. Relative species richness estimates for the 11 remaining walk-away sites ranged from a low of 0.19 on sites w10 and w13 to a high of 0.42 on sites w3 and w22 (Table 13). Applying the relative species richness metric resulted in 14% of the sites, irrespective of design strategy, ranked as poor, 74% of the sites ranked as fair, 10%

Table 13. Naïve relative species richness estimates (calculated by dividing the number of species detected by 17, i.e., the number of species in the regional species pool) and relative species richness estimates with associated standard errors (se), and 95% confidence intervals computed using the occupancy method on 18 maximize hydrology (m), 19 naturalistic (n), and 13 walk-away (w) sites surveyed for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. These estimates represent the average probability that a member of the regional species pool is present on a site.

Site	Naïve estimate	Relative species richness		95 % confidence interval	
		richness est	se	lower	upper
m6	0.35	0.36	0.12	0.17	0.6
m10	0.35	0.37	0.12	0.18	0.63
m12	0.29	0.41	0.17	0.14	0.74
m18	0.41	0.47	0.14	0.23	0.73
m21	0.18	0.27	0.17	0.06	0.66
m22	0.24	0.29	0.13	0.1	0.58
m26	0.24	0.27	0.12	0.1	0.54
m28	0.35	0.38	0.12	0.18	0.63
m32	0.24	0.24	0.1	0.09	0.49
m34	0.24	0.24	0.1	0.09	0.49
m35	0.41	0.46	0.14	0.23	0.72
m38	0.47	0.5	0.13	0.27	0.73
m43	0.47	0.47	0.12	0.26	0.7
m44	0.35	0.36	0.12	0.17	0.6
m46	0.35	0.35	0.12	0.17	0.6
m47	0.41	0.42	0.12	0.21	0.66
m49	0.47	0.51	0.13	0.27	0.75
m50	0.29	0.29	0.11	0.13	0.54
n2	0.47	0.48	0.12	0.26	0.71
n3	0.53	0.53	0.12	0.3	0.75
n5	0.41	0.42	0.12	0.21	0.66
n6	0.41	0.41	0.11	0.21	0.65
n7	0.29	0.29	0.12	0.13	0.54
n8	0.47	0.47	0.11	0.26	0.7
n10	0.29	0.29	0.11	0.13	0.54
n11	0.35	0.35	0.4	0.17	0.6
n13	0.35	0.72	0.12	0.05	0.99
n16	0.35	0.36	0.13	0.17	0.6
n17	0.35	0.39	0.11	0.18	0.65
n18	0.29	0.31	0.12	0.13	0.56
n20	0.53	0.54	0.12	0.3	0.75
n22	0.29	0.31	0.12	0.13	0.57

Table 13. (continued)

Site	Naïve estimate	Relative species		95 % confidence interval	
		richness est	se	lower	upper
n24	0.35	0.37	0.11	0.17	0.62
n26	0.24	0.24	0.12	0.09	0.5
n29	0.41	0.43	0.12	0.22	0.67
n31	0.41	0.41	0.12	0.21	0.65
n33	0.53	0.55	0.13	0.31	0.76
w1	0.41	0.41	0.12	0.21	0.65
w3	0.41	0.42	0.12	0.21	0.61
w4	0.29	0.31	0.12	0.13	0.57
w9	0.24	0.26	0.12	0.1	0.54
w10	0.18	0.19	0.1	0.06	0.45
w12	0.41	0.41	0.12	0.21	0.65
w13	0.18	0.19	0.1	0.06	0.46
w14	0.29	0.34	0.13	0.14	0.62
w18 ^a	0.12	*	*	*	*
w19	0.18	0.36	0.3	0.04	0.87
w20	0.35	0.36	0.12	0.17	0.61
w21 ^a	0.12	*	*	*	*
w22	0.35	0.42	0.15	0.18	0.7

^a Sparse data resulted in no estimate for these sites because of bounding and convergence issues.

of the sites ranked as good, and 2% of the sites ranked as very good for wetland restoration efforts (Figure 19). However, if n13 is reduced to a fair ranking, then no sites would be ranked as very good, 76% of the sites would be ranked as fair and the percent poor and good would remain the same.

Capture/Recapture Method

Results for the species richness analysis using the capture/recapture method included average species richness estimates for maximize hydrology sites of 4.6 (sd=2.3; CI=3.7-5.5), for naturalistic sites of 5.7 (sd=3.4; CI=4.4-7.1), and for walk-away sites of 3.5 (sd=2.2; CI=2.4-4.6). These estimates represent the average number of species detected on a site, based on the number of detected species with no upper limit on the estimate. Converting these averages to a proportion of the regional species pools resulted in relative species richness estimates of 0.27 for maximize hydrology sites, 0.34 for naturalistic sites, and 0.21 for walk-away sites. Applying the relative species richness metric resulted in both maximize hydrology and naturalistic sites receiving a fair assessment for wetland restoration efforts and walk-away sites receiving a poor assessment.

Species richness estimates for 18 maximize hydrology sites ranged from a low of one on site m21 to a high of 11 on site m38 (Table 14). Species richness estimates for 19 naturalistic sites ranged from a low of three on sites n7, n17, and n22 to a high of 18 on site n31 (Table 14). However, the high standard error and wide confidence interval (se=6.6; CI=18-38) associated with n31 requires that this estimate be treated with caution. Species richness estimates for 13 walk-away sites ranged from a low of 1 on sites w10, w21, and w22 to a high of eight on site w20 (Table 14). Converting these

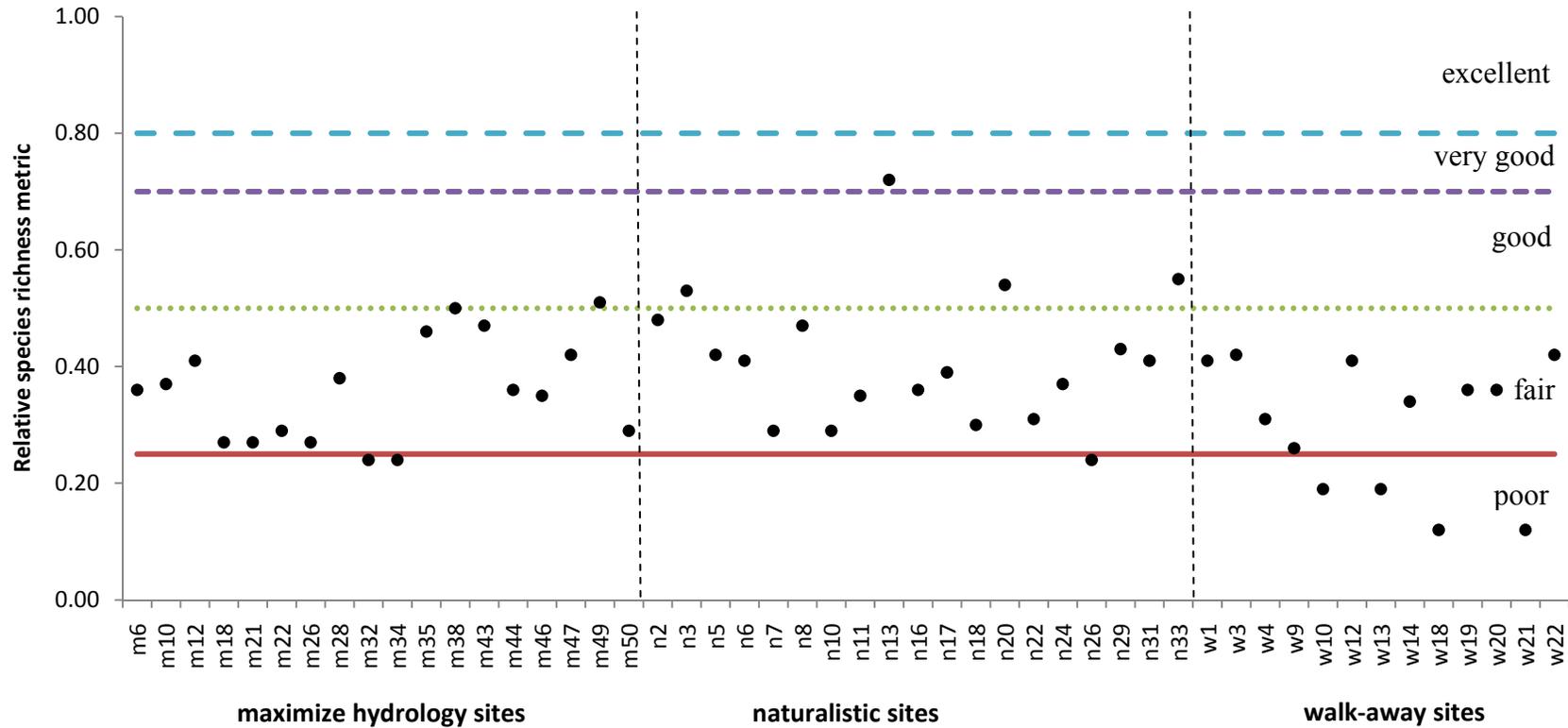


Figure 19. Rankings applied to Wetlands Reserve Program sites using relative species richness metric based on the occupancy method for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. Sites with a relative species richness estimate of ≤ 0.24 (below red, solid line) were ranked as poor, sites with a relative species richness estimate of 0.25 – 0.49, inclusively, were sites ranked as fair (on or above red, solid line and below green, dotted line), sites with a relative species richness estimate of 0.50 – 0.69, inclusively, were ranked as good (on or above green, dotted line and below purple, short dash line), and sites with a relative species richness estimate of 0.70 – 0.79, inclusively, were ranked as very good (on or above purple, short dash line and below aqua, long dash line) for wetland restoration efforts. Sparse data associated with sites w18 and w21 resulted in no estimate for these sites so the naïve estimates (2 species detected divided by 17, i.e., number of species in the regional species pool) were used instead.

Table 14. Species richness estimates (\hat{N}), standard errors (se), and 95% confidence intervals computed using the capture/recapture method on 18 maximize hydrology (m), 19 naturalistic (n), and 13 walk-away (w) sites surveyed for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. Models were selected using model selection algorithm in program CAPTURE. If the original model selected did not provide a reasonable estimate, the next highest ranked model was used that produced reasonable estimates. A description of model definitions can be found in Table 3.

Site	Model	\hat{p}	\hat{N}	se(\hat{N})	Coefficient of variation	95% CI	Number quadrats with detections
m6	M _h	0.14	6	3.37	0.56	6 to 23	5 of 11
m10	M _{th}	varies by quadrat	6	0.68	0.11	6 to 10	5 of 11
m12	M _{tth}	assumed-no recaptures so analysis stopped-proceeded to next model					2 of 10
	M _h	0.1	3	2.46	0.82	3 to 3	
m18	M ₀	0.36	2	0.12	0.06	2 to 2	5 of 11
m21	M _h	because only 1 animal recaptured estimate taken as 1 with variance of 1					1 of 10
m22	M ₀	0.18	2	0.6	0.3	2 to 2	4 of 11
m26	M ₀	0.17	2	0.92	0.46	2 to 2	2 of 11
m28	M ₀	0.22	5	0.78	0.16	5 to 5	7 of 10
m32	M ₀	0.23	4	0.56	0.14	4 to 4	7 of 11
m34	M ₀	0.27	4	0.38	0.09	4 to 4	8 of 11
m35	M ₀	0.08	8	3.46	0.43	8 to 23	5 of 11
m38	M _{th}	varies by quadrat	11	4.44	0.4	11 to 30	6 of 11
m43	M ₀	0.27	5	0.42	0.08	5 to 5	8 of 11
m44	M _{th}	varies by quadrat	5	0	0	5 to 5	6 of 10
	M ₀	0.32	5	0.35	0.07	5 to 5	
m46	M ₀	0.25	4	0.46	0.11	4 to 4	8 of 11
m47	M _h	0.19	8	2.72	0.34	8 to 21	5 of 11
m49	M ₀	0.09	7	3.41	0.49	7 to 23	4 of 10
m50	M _{tth}	no estimator-proceeded to next model					7 of 10
	M _{bh}	no output-proceeded to next model					
	M ₀	0.52	3	0.07	0.02	3 to 3	
n2	M ₀	0.13	8	2.07	0.26	8 to 18	7 of 11
n3	M _h	0.29	9	1.96	0.22	9 to 17	8 of 11
n5	M ₀	0.24	5	0.58	0.12	5 to 5	8 of 11
n6	M ₀	0.3	4	0.31	0.08	4 to 4	8 of 11
n7	M ₀	0.33	3	0.19	0.06	3 to 3	8 of 11
n8	M ₀	0.49	5	0.05	0.01	5 to 5	11 of 11
n10	M _h	0.27	5	0.22	0.04	5 to 5	9 of 15
n11	M ₀	0.39	4	0.14	0.03	4 to 4	10 of 11

Table 14. (continued)

Site	Model	\hat{p}	\hat{N}	se(\hat{N})	Coefficient of variation	95% CI	Number quadrats with detections	
n11	M0	0.39	4	0.14	0.03	4 to 4	10 of 11	
n13	M _h	0.07	5	2.00	0.39	5 to 13	2 of 11	
n16	M _h	0.27	5	0.20	0.04	5 to 5	6 of 11	
n17	M ₀	0.21	3	0.55	0.18	3 to 3	5 of 11	
n18	M ₀	0.16	4	1.38	0.34	4 to 10	4 of 11	
n20	M ₀	0.18	4	0.84	0.21	4 to 4	7 of 11	
n22	M ₀	0.21	3	0.55	0.18	3 to 3	5 of 11	
n24	M ₀	0.12	6	2.00	0.33	6 to 16	6 of 11	
n26	M _{th}	varies by quadrat	5	1.22	0.24	5 to 11	5 of 11	
n29	M ₀	0.17	8	1.30	0.16	8 to 15	8 of 11	
n31	M _h	0.11	18	6.60	0.37	18 to 38	10 of 11	
n33	M ₀	0.21	7	0.89	0.13	7 to 11	8 of 11	
w1	M ₀	0.38	6	0.23	0.04	6 to 6	9 of 10	
w3	M ₀	0.22	5	0.68	0.14	5 to 5	7 of 11	
w4	M _{tbb}	assumed-no recaptures so analysis stopped-proceeded to next model						
	M _h	0.10	4	3.85	0.96	4 to 4		
w9	M ₀	0.14	2	0.94	0.47	2 to 2	2 of 11	
w10	M _h	because only 1 animal was captured estimate taken as 1 with variance of 1						3 of 10
w12	M _{th}	varies by quadrat	6	0.00	0.00	6 to 6	8 of 9	
	M ₀	0.54	6	0.08	0.01	6 to 6		
w13	M ₀	0.17	3	0.98	0.33	3 to 7	5 of 10	
w14	M ₀	0.18	3	0.73	0.24	3 to 3	5 of 11	
w18	M _{tbb}	assumed-no recaptures so analysis stopped						2 of 10
	M _h	0.05	4	2.42	0.60	4 to 15		
w19	M _{bh}	varies by quadrat	3	0.00	0.00	3 to 3	3 of 10	
w20	M ₀	0.07	8	5.23	0.65	8 to 33	4 of 10	
w21	M _{tbb}	assumed-no recaptures so analysis stopped-proceeded to next model						1 of 9
	M _h	because only 1 animal was captured estimate taken as 1 with variance of 1						
w22	M _h	because only 1 animal recaptured estimate taken as 1 with variance of 1						2 of 11

figures to a proportion of the regional species pool and applying the relative species richness metric resulted in 52% of the sites, irrespective of design strategy, receiving a poor ranking, 42% of the sites receiving a fair rating, 4% of the sites receiving a good rating and 2% of the sites receiving an excellent rating for wetland restoration efforts (Figure 20).

The final model selections for the species richness analysis using Program CAPTURE included model M_0 for 33 or 66% of the sites, model M_h for 13 or 26% of the sites, and model M_{th} for 4 or 8% of the sites (Table 14). Overall, 17 or 34% of the sites exhibited some degree of heterogeneity in detection probabilities among species and/or quadrats. However, rather unexpectedly, 66% of the sites did not exhibit heterogeneity in detection probabilities either among species or quadrats.

Design Strategy as Relevant Ecological Criterion

My fifth objective assessed whether design strategy served as useful criterion of wetland restoration efforts by describing a functional ecological attribute of restored sites; i.e., an evaluation of whether design strategy category accurately reflected the hydrological wetland characteristics restored to the sites. K-means cluster analysis provided a method to evaluate if WRP sites classified by design strategy were more similar or different than sites within the same category using means of selected habitat variables. (Average values of habitat variables used to conduct the cluster analysis are presented in Appendix H). If the sites grouped together by design strategy according to my hypothesis, maximize hydrology and walk-away sites would be dominated by the dry cluster representing ephemeral wetlands and naturalistic sites would be dominated by the wet with grassy vegetation cluster representing seasonal wetlands. The sampled WRP

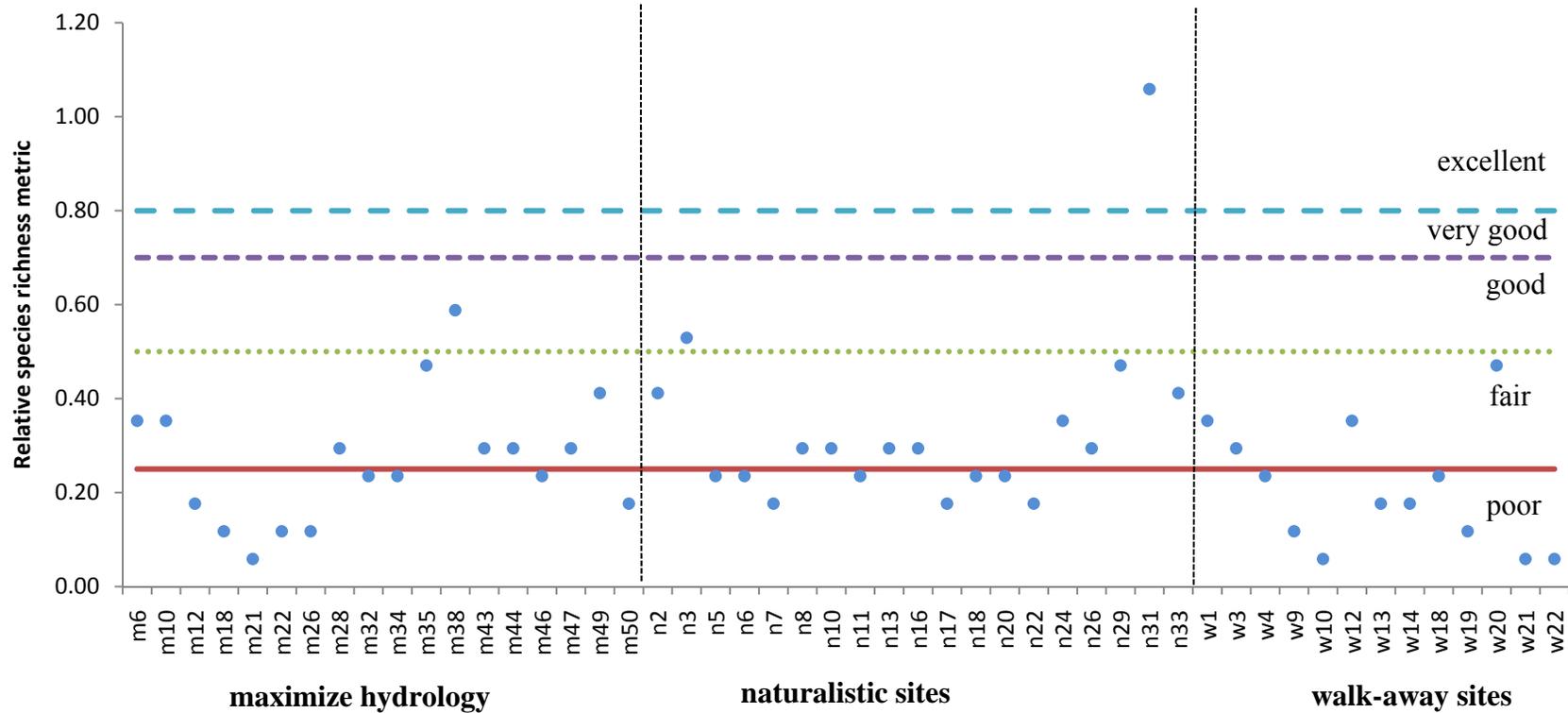


Figure 20. Rankings applied to Wetlands Reserve Program sites after using the capture/recapture method to estimate species richness and converting the estimates with the relative species richness metric (species richness estimate/number of species in regional species pool) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri in 2007. Sites with a relative species richness estimate of ≤ 0.24 (below red, solid line) were ranked as poor, sites with a relative species richness estimate of 0.25 – 0.49, inclusively, were sites ranked as fair (on or above red, solid line and below green, dotted line), sites with a relative species richness estimate of 0.50 – 0.69, inclusively, were ranked as good (on or above green, dotted line and below purple, short dash line), sites with a relative species richness estimate of 0.70 – 0.79, inclusively, were ranked as very good (on or above purple, short dash line and below blue, long dash line), and sites with a relative species richness estimate ≥ 0.80 were ranked as excellent (on or above blue, long dash line) for wetland restoration efforts.

sites, however, did not group together according to design strategy or my dry (ephemeral), wet with grass-like vegetation (seasonal), or open water (permanent) hypothesis. Instead, average proportion of sampled quadrats that were dry was the most dominant variable in the first analysis that used the overall average of each variable per site and average water depth of sampled quadrats was the less influential of the four measured variables based on mean square error and F-test values (Table 15). Cluster 1 included 16 sites with wet, grass-like vegetation as the dominant variable including 3 maximize hydrology sites, 10 naturalistic sites, and 3 walk-away sites; Cluster 2 included five sites with open water as the dominant variable including 3 maximize hydrology and 2 naturalistic sites; and Cluster 3 included 29 sites with dry as the dominant variable including 11 maximize hydrology, 7 naturalistic, and 10 walk-away sites (Figure 21). From this analysis, 31 of the 50 sites matched hypothesized conditions; however, six maximize hydrology sites, two naturalistic sites, and three walk-away sites were wetter than expected, and seven naturalistic sites were drier than expected (Figure 22).

Results of the three analyses by individual primary sampling period provided a method to evaluate if site conditions were dynamic across time. Average proportion of sampled quadrats that were dry was the dominant variable in primary sampling periods one and two whereas average water depth of sampled quadrat was the least influential variable based on mean square error and F-test values (Table 16). Average proportion of sampled quadrats covered with open water was the most dominant variable in primary sampling period three whereas average water depth of sampled quadrats was again the least influential variable based on mean square error and F-test values (Table 16). Dominance of the open water cluster in primary survey period three was unexpected

Table 15. ANOVA table for k-means cluster analysis using overall site averages from sampled WRP sites for amphibian occupancy and species richness project conducted in Lower Grand River basin, northcentral Missouri in 2007. Variables include average proportion of sampled quadrats dry (dry), average proportion of sampled quadrats wet with grass-like vegetation (grass), average proportion of sampled quadrats covered with open water (ow), and the average water depth (depth) on sampled quadrats. The larger the mean square error value and the smaller the F value, the less influential the variable. The F tests should be used only for descriptive purposes because the clusters have been chosen to maximize the differences among cases in different clusters. The observed significance levels are not corrected for this and thus cannot be interpreted as tests of the hypothesis that the cluster means are equal.

Variable	Cluster		Error		F	Sig.
	Mean	df	Mean	df		
	Square		Square			
dry	19.297	2	0.221	47	87.150	0.000
grass	15.956	2	0.364	47	43.883	0.000
ow	17.012	2	0.319	47	53.390	0.000
depth	10.300	2	0.604	47	17.047	0.000

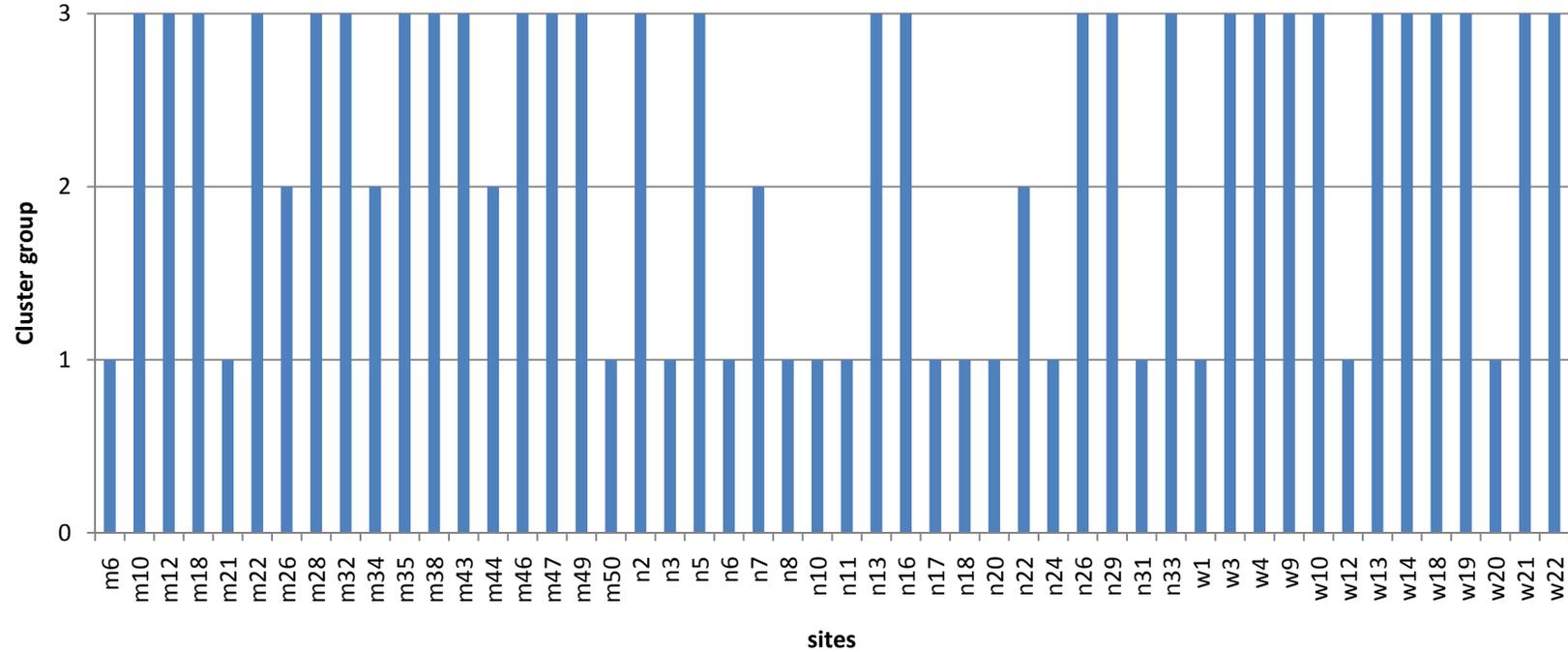


Figure 21. Results of k-means cluster analysis based on overall site averages from 50 sampled Wetlands Reserve Program sites for three primary survey periods conducted during 2007 field season (March – September) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri, illustrating that the sites did not group together by design strategy. If the results had matched hypothesized conditions, all maximize hydrology and walk-away sites would have been dominated by dry conditions whereas naturalistic sites would have been dominated by wet conditions with grass-like vegetation. Cluster group 1 is dominated by wet conditions with grass-like vegetation, Cluster group 2 is dominated by open water, and Cluster group 3 is dominated by dry conditions. (m=maximize hydrology sites, n=naturalistic sites, and w=walk-away sites).

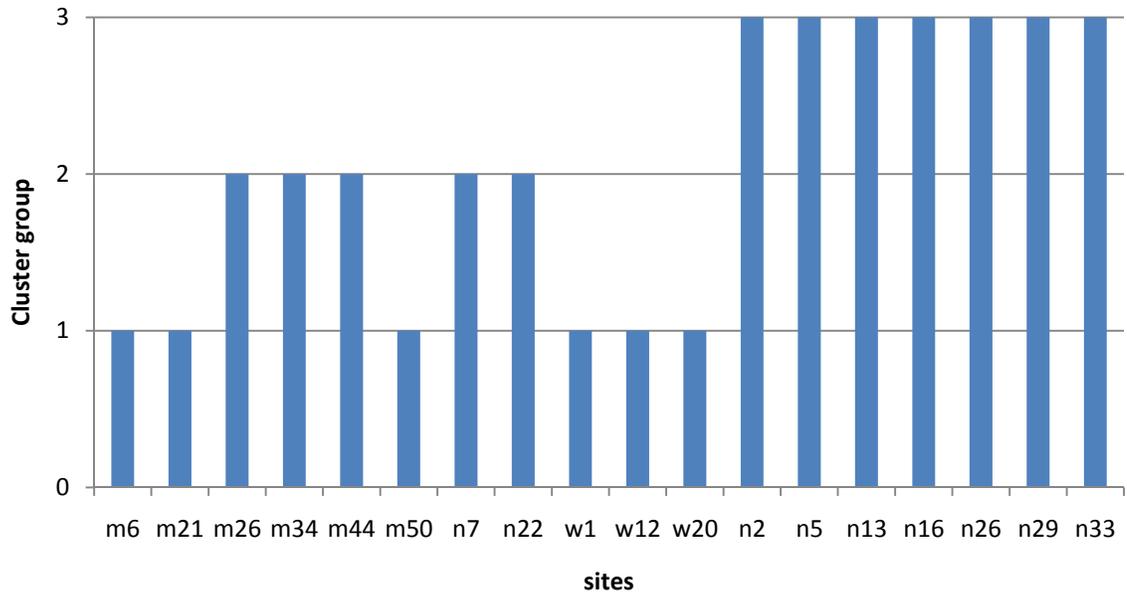


Figure 22. Results of k-means cluster analysis that indicated 18 of 50 sampled Wetlands Reserve Program sites surveyed during 2007 field season (March – September) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri did not match hypothesized conditions. All maximize hydrology and walk-away sites were hypothesized to be dominated by dry conditions (cluster group 3) whereas naturalistic sites were hypothesized to be dominated by wet conditions with grass-like vegetation (cluster group 1). Instead, six of 18 maximize hydrology sites were wetter than expected and dominated by either wet conditions with grass-like vegetation (cluster group 1) or open water (cluster group 2), three of 13 walk-away sites were wetter than expected as demonstrated by dominance of wet conditions with grass-like vegetation (cluster group 1), and two of 19 naturalistic sites were wetter than expected as demonstrated by dominance of open water conditions (cluster group 2). Seven of 19 naturalistic sites were drier than expected as demonstrated by dominance of dry conditions (cluster group 3). (m=maximize hydrology sites, n=naturalistic sites, and w=walk-away sites).

Table 16. ANOVA table for k-means cluster analyses using averages from quadrats sampled during each of three primary survey periods conducted during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. Variables include average proportion of sampled quadrats dry (dry), average proportion of sampled quadrats wet with grass-like vegetation (grass), average proportion of sampled quadrats covered with open water (ow), and average water depth (depth) on sampled quadrats. The number following each variable indicates the primary sampling period with which it is associated, e.g., dry.1 is the average of dry for primary sampling period one. The larger the mean square error value and the smaller the F value, the less influential the variable. The F tests should be used only for descriptive purposes because the clusters have been chosen to maximize the differences among cases in different clusters. The observed significance levels are not corrected for this and thus cannot be interpreted as tests of the hypothesis that the cluster means are equal.

Variable	Cluster		Error		F	Sig.
	Mean Square	df	Mean Square	df		
dry.1	19.053	2	0.211	46	90.467	0.000
grass.1	14.410	2	0.404	46	35.668	0.000
ow.1	18.699	2	0.266	46	70.288	0.000
depth.1	4.883	2	0.809	46	6.033	0.005
dry.2	17.838	2	0.245	46	72.665	0.000
grass.2	14.906	2	0.398	46	37.421	0.000
ow.2	16.260	2	0.339	46	47.999	0.000
depth.2	9.683	2	0.550	46	17.621	0.000
dry.3	16.584	2	0.326	46	50.798	0.000
grass.3	16.454	2	0.343	46	48.037	0.000
ow.3	19.755	2	0.197	46	100.470	0.000
depth.3	2.804	2	0.885	46	3.170	0.051

given that 40 of the 49 sites were dominated by the dry cluster, seven were dominated by the wet with grass-like vegetation cluster, and only two were dominated by the open water cluster (Figure 23); however, the distance between final cluster centers for the open water variable (4.057) varied sufficiently enough from the final grass (2.017) and dry (0.362) cluster centers that it emerged as the dominant variable. Twenty-eight of the 49 sites included in the analysis did not change clusters among the three primary sampling periods including 24 dominated by the dry variable, three sites dominated by the wet, grass-like vegetation variable, and one dominated by the open water variable (Figure 23).

DISCUSSION

Occupancy and Wetland Design Strategy

Design strategy, as a site-level covariate, did not contribute toward explaining heterogeneity in occupancy probability estimates for my study. Covariates provide a means to account for heterogeneity in data; however, because occupancy is assumed constant within a season, covariates associated with the occupancy parameter cannot change throughout the course of the season meaning the covariate must represent a characteristic of the site that does not change (MacKenzie et al. 2006). Design strategy served as a site-level covariate because, once assigned, a site's design strategy category did not change. However, design strategy describes management actions rather than ecological outcomes created by management actions to which species respond. Design strategy was a broad, artificially-created category that attempted to classify sites based on what were, primarily, infrastructure modifications combined with an attempt to predict the ecological consequences of these modifications. Initially, I anticipated that walk-away sites would be homogeneously dry and that maximize hydrology sites would be

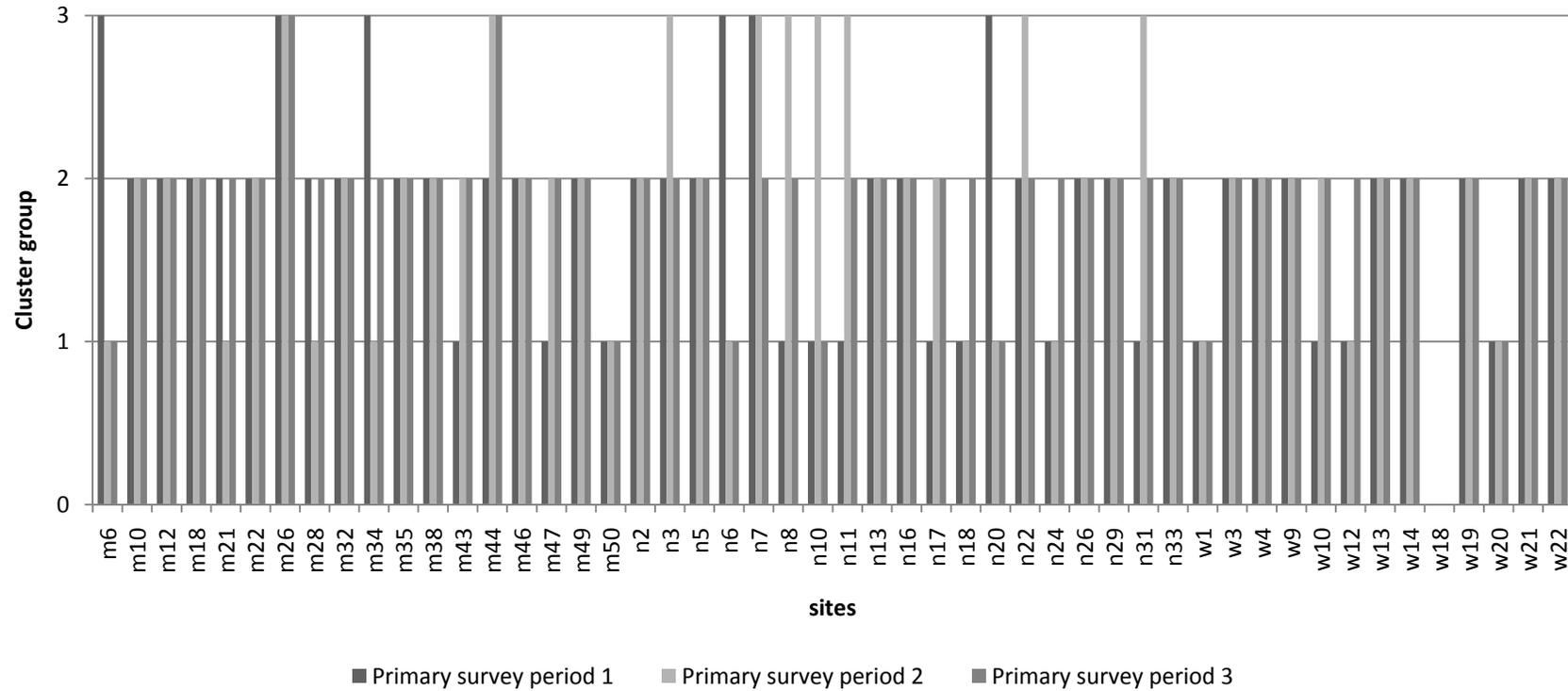


Figure 23. Results of k-means cluster analysis for each of three primary survey periods conducted on 49 Wetlands Reserve Program sites during 2007 field season (March – September) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. Dry (cluster group 2) was the most influential variable during primary survey periods one and two whereas open water (cluster group 3) was the most influential variable during primary survey period three even though only two sites were dominated by open water in primary survey period three (m26 and m44). Twenty-eight of the sites did not change clusters among primary survey periods. Cluster group 1 is dominated by wet conditions with grass-like vegetation. (m=maximize hydrology sites, n=naturalistic sites, and w=walk-away sites). Site w18 was not included in the analysis due to missing data.

homogeneous open water followed by homogeneous dry conditions due to early drawdowns with both strategies representing ephemeral hydroperiods that were not of sufficient duration to allow successful amphibian recruitment (Figure 2). I anticipated that naturalistic sites would have longer hydroperiods that would enable successful amphibian recruitment and that this would be reflected by higher occupancy probability estimates for naturalistic sites than either walk-away or maximize hydrology sites. Instead, amphibian species representing seasonal to permanent hydroperiods occurred across all sites resulting in no difference among design strategies based on occupancy and my study design. Proportion of area occupied estimates of amphibians representing different hydroperiods indicated sites within all three design strategies had varying hydrological wetland conditions resulting in greater habitat heterogeneity than anticipated on maximize hydrology and walk-away sites and less habitat heterogeneity than anticipated on naturalistic sites. There was not a consistent relationship between design strategy and hydroperiod as demonstrated not only by the occupancy results but also by the cluster analysis in which the sites did not group together by design strategy, thus, falsifying my hypothesis (Figure 21). Instead, approximately 40% of the 50 sampled WRP sites did not conform to hypothesized conditions including 33% of maximize hydrology sites, 11% of naturalistic sites, and 23% of walk-away sites that were wetter than expected and 37% of naturalistic sites that were drier than expected (Figure 22). Design strategy did not serve as a surrogate for hydroperiod and was not a useful covariate to explain any heterogeneity in occupancy probability estimates. Design strategy represents a historical perspective that presented a framework around which to evaluate how changes to management approach met program objectives. However, the

degree of heterogeneity in habitat conditions at the within site-scale demonstrated that amphibians are responding to ecological conditions that occur at a finer resolution than the site scale.

Additionally, results of the site occupancy model selection process in which the null model was most supported may represent a Type II error (accepting the null hypothesis of no difference among design strategies when, in fact, it should be rejected) in which there were differences among design strategies but the combination of overdispersed and sparse data obscured my ability to distinguish the differences. Compensation is made for overdispersed data when using the model selection procedure by adjusting AIC_c values to a quasi-likelihood through the use of $QAIC_c$ and inflating the variance associated with parameter estimates. This results in conservative inference because simpler models are selected as the variance inflation factor, \hat{c} , increases. The danger then becomes that the models are too simple and do not adequately account for heterogeneity in the data (Anderson 2008). Therefore, if the best model selected in the procedure is the null with no effect due to design strategy, rather than infer that design strategy has no effect, one should infer that the variation in some parameter imbedded within design strategy is small and such variation could not be detected with the amount of information in the data (Anderson 2008). The differences among design strategies appeared stark enough that the effect size would be sufficient to detect based on amphibian occupancy; however, results from both the occupancy and cluster analyses illustrate that this was not the case and, again, emphasizes the fact that amphibians are responding to ecological conditions that occur at a finer resolution than the site-level scale.

The question as to whether hydrological wetland characteristics as indicated by amphibian distribution varied by design strategy represented an attempt to evaluate if management actions had preceded in an adaptive fashion based on experience gained with wetland restoration efforts in a riverine-floodplain. It is still a valid question as it attempts to determine if wetland restoration efforts are achieving program objectives; however, it highlights the importance of ensuring a criteria (i.e., design strategy) used to assess a biological response (i.e., amphibian distribution) to management actions (i.e., hydrological restoration) matches the scale of that response. Amphibians distribute themselves along a hydrological gradient (Babbitt 2003, 2005) and so provide a means to evaluate whether restoration efforts are restoring a gradient of hydrological conditions. Therefore, a site-level covariate assessing restoration efforts based on amphibian distribution should be a measure of the primary ecological attribute driving amphibian distribution for the restored wetland feature, irrespective of design strategy (Shulse et al. 2010). Answering this question requires redefining the sample site, or sample unit, used for my study from an entire WRP site to a specific wetland feature within a site and defining an ecologically relevant, site-level covariate that accounts for heterogeneity in occupancy among these wetland features. Hydroperiod remains a relevant site covariate. Management actions create different wetland features with differing hydroperiods. Selecting these wetland features; i.e., excavated wetlands, borrows, shallow pools, etc; as the primary sample site and characterizing the duration of their hydroperiod then results in an occupancy covariate that more closely reflects the ecological heterogeneity affecting species occupancy (Shulse et al. 2010). This provides a direct measure of the abiotic process driving amphibian distribution (i.e., hydroperiod) while still measuring

the biological component (i.e., amphibian occupancy). Additionally, time could be used as an alternate covariate to design strategy as a means to evaluate if management actions reflect incremental learning. The year a feature is constructed or completed would remove the issue of whether each property fit the idealized characteristics of design strategy and yet still capture modifications made to management actions that occur as experience is gained over time.

Violations of Model Assumptions

Results from the occupancy analyses, particularly the multi-state occupancy analysis, reflected a high degree of overdispersion. Overdispersion generally occurs when model assumptions are violated. The two assumptions most likely violated during the 2007 field season were the closure assumption (i.e., sites were closed to changes in occupancy throughout the course of the study), and the independence assumption (i.e., individuals were independently and identically distributed [Anderson 2008]). The closure assumption may have been violated due to the seasonal movement patterns exhibited by amphibians resulting in movements into or out of the sampled sites. I used truncated data sets in an attempt to compensate for any potential closure violations; this was likely adequate for species that were present throughout the majority of the field season such as leopard frogs, cricket frogs, and bullfrogs. However, it may have been inadequate for species that exhibited very pronounced seasonal movements such as grey treefrogs. Grey treefrogs metamorphose in an extremely short timeframe with hundreds of metamorphs inhabiting vegetation along the edge of the wetland from which they transformed until a rain event occurs. Once precipitation falls, all treefrog metamorphs exit the immediate vicinity of the wetland. It is unlikely we visited all sites with grey

treefrog metamorphs before a rain event initiated a mass dispersal movement. Thus, the assumption of closure was likely violated resulting in negatively biased estimates, particularly relevant for the estimate of overall recruitment success. This highlights the importance of completing repeat surveys in a short timeframe when conducting occupancy studies. Also, future studies involving amphibians and occupancy might consider an open model described by Kery et al. (2009) that presents a method of species richness and occupancy estimation while still accounting for imperfect detection probability. Their model is a community-level assessment applicable in a seasonal environment in which a species may be absent at the beginning of a season, then enter the system and be present for a period of time before exiting the system. The Kery et al. (2009) method, although computer-intensive, appears, intuitively, to be a good fit for amphibians, a group of species that exhibit strong seasonal activity patterns, and is certainly worth further investigation.

Another potential violation to the closure assumption was the May flood event that likely redistributed amphibians both within and among sites; these movements may represent colonization events if individuals were deposited on unoccupied sites and extinction events if all individuals were removed from a site. Amphibian movements associated with the flood event may be a combination of both random and non-random movements depending on the species, size, and life history stage of the individual as well as the amount of refugia available on a given site (Harvey 1987, Andersen et al. 2000, Tockner et al. 2006). For example, an adult grey treefrog may use an arboreal refuge by climbing up a tree (non-random movement), whereas, a juvenile bullfrog may be displaced by flood waters (random movement). Determining how the flood event

affected occupancy estimates for my study is somewhat difficult given that limited investigations that have been conducted regarding violation of the closure assumption as it applies to the occupancy model. However, due to the close relationship between occupancy and closed capture/recapture models, it is generally assumed Kendall's (1999) evaluation of species movements and how they bias population estimates are also applicable to occupancy models (MacKenzie et al. 2006). Kendall (1999) found that completely random movements of both immigration (colonization) and emigration (extinction) did not introduce bias to population estimators, although it did decrease precision. However, if the movements were non-random and occurred in one direction, e. g., a migratory species that is present at the beginning of a study and begins to exit before the study is complete, then Kendall (1999) determined the closure assumption was violated and population estimators were biased. Rota et al. (2009) expanded on this concept by evaluating how or if violation of the closure assumption biased occupancy parameter estimates. They demonstrated that even small amounts of simultaneous colonization and extinction movements lead to overestimates of occupancy. If the May 2007 flood event represented non-random movements or resulted in redistribution of species to previously unoccupied sites, then occupancy estimates for my study would be positively biased (Rota et al. 2009). Future studies can avoid closure violations by ensuring a sufficient number of repeat surveys are conducted during a short timeframe when species are available for detection. This timeframe is April through July for the local amphibian assemblage in the Lower Grand River basin. However, floods are frequent occurrences during the April through July period so if flood events represent a closure violation that results in biased occupancy estimates, then closed models may not

be an appropriate choice for use in riverine floodplains, particularly for a suite of species such as amphibians that exhibit strong seasonal movements. Again, the open model described by Kery et al. (2009) may be a feasible alternative although flood events may also prove problematic with this model which once more emphasizes the importance of completing repeat surveys in a short timeframe.

The independence assumption may also have been violated during my study based on the tendency of amphibians to group together in common habitats (i.e., juvenile leopard frogs, cricket frogs, and bullfrogs congregating on the wet margins of wetlands or larval amphibians grouping together under a shaded area along a wetland bank). Additionally, the independence assumption may have been violated due to study design issues and the manner in which the detection methods were deployed. The study design did not specify a minimum distance between quadrats which introduced a lack of independence. Also, although each detection method was conducted sequentially, they were all utilized in the same quadrat, introducing the possibility that an individual may have been detected by more than one method. This would result in less information and positively biased estimates. Future studies can avoid violations to the independence assumption by following previous recommendations given to avoid closure violations; i.e., ensuring a sufficient number of repeat surveys are conducted during a short timeframe when species are available for detection, and by specifying a minimum distance between quadrats. This would enable combining information from multiple detection methods, thus removing the independence issue and result in more precise and unbiased estimates.

Restoration of Hydrological and Biological Wetland Characteristics

Species assemblages can provide distinct information about ecosystem structure and function (Tockner et al. 1999). Amphibians served as indicators of water permanency, i.e., hydroperiod, (Morand and Joly 1995, Tockner et al. 1999, Babbitt et al. 2004) for my study to determine if each design strategy was restoring a gradient of hydrological conditions to WRP sites in the Lower Grand River basin based on occupancy by members of the regional species pool. The outcome of this investigation highlights a fundamental concept underpinning knowledge of riverine ecosystems, i.e., riverine floodplains are a dynamic mosaic of patterns and processes occurring across spatial (wetland, site, floodplain, catchment) and temporal (seasonal, geological) scales (Tockner et al. 2000, Ward et al. 2002, Tockner et al. 2010). Attempting to segregate discrete groups of sites within the floodplain based on design strategy failed to consider the dynamic nature of floodplains and the processes acting therein. Restoring wetland features on WRP sites across the landscape subjects those sites to the dominant hydrogeomorphic and ecological processes that originally created and shaped the shifting patterns of successional stages typical of a riverine floodplain (Poole 2010, Tockner et al. 2010). Therefore, for the purposes of my study, it is more appropriate to consider all Lower Grand River basin WRP sites collectively to determine if the program is restoring the “hydrological and biological characteristics of wetlands” rather than consider groups of parcels based on design strategy.

The occupancy analysis indicated that seven of the 11 species or complexes detected; American toad, northern cricket frog, *Pseudacris* spp., Plains leopard frog, American bullfrog, southern leopard frog, and leopard frog complex; were widely-

distributed among WRP sites in the Lower Grand River basin ($\hat{\psi} > 0.80$). This group represents species that are generally associated with both seasonal and permanent wetlands implying that both seasonal and permanent wetlands are also widely-distributed on WRP sites in the Lower Grand River basin. However, this implication is somewhat tempered by a closer examination of the occupancy results.

American toads, *Pseudacris* spp., and leopard frogs represent the seasonal species included in the widely-distributed group. Both American toad and *Pseudacris* spp. are early breeding species (Johnson 2000) whereas the breeding chronology of leopard frogs is slightly later than toads and *Pseudacris* spp. and of a longer duration. Therefore, while leopard frogs, generally, are also associated with seasonal wetlands, late-hatched leopard frog larvae frequently overwinter in breeding pools, indicating an ability to exploit wetlands with longer duration than seasonal (Johnson 2000). The wide-distribution of these three species or complexes implies seasonal wetlands are also widely-distributed on WRP sites in the Lower Grand River basin although the wide-spread distribution of leopard frogs also hints at the wide-spread distribution of wetlands with longer duration than seasonal. Additionally, the occupancy estimate associated with the American toad should be treated with caution due to the combination of a sparse data set, high standard error ($se=0.29$) and a detection probability estimate less than 0.30 (maximum $\hat{p}_{amt}=0.19$) (MacKenzie et al 2002). The detection probability estimates associated with both Plains and southern leopard frogs are also relatively low (maximum $\hat{p}_{plf \text{ and } slf}=0.27$), due to relatively sparse data; however, the sparse data associated with these species were related to an inability to differentiate between species until they reach the metamorph stage. Leopard frogs, in general, as indicated by both the occupancy and detection estimates for

leopard frog complex ($\hat{\psi}_{\text{lf}}=0.97$ and maximum $\hat{p}_{\text{lf}}=0.78$) are very widely-distributed across WRP sites in the Lower Grand River basin and are highly detectable. Therefore, given the uncertainty associated with the toad occupancy estimate and the ability of leopard frogs to exploit both seasonal and more permanent wetlands, the occupancy estimate associated with *Pseudacris* spp. provides the strongest evidence that seasonal wetlands are well represented on WRP sites.

Although the occupancy estimate associated with the American toad must be treated with caution due to the accompanying low detection probability estimate (MacKenzie et al. 2002), it seems reasonable that toads would be somewhat widely distributed across the Lower Grand River basin given the prevalence of dry conditions on sampled WRP sites and the toad's association with drier, more terrestrial conditions (Johnson 2000). The low number of toad detections and resulting detection probability estimate may simply reflect the long intervals between primary survey events that, when combined with the toad's accelerated life cycle as compared to the other detected species associated with seasonal wetlands (approximately 35 days to metamorphose versus average of approximately 56 days for other species; Johnson 2000) may have resulted in a high number of false negatives. Or, alternatively, it may reflect that riverine floodplains are disturbance-dominated landscapes with floods as major regulators of both aquatic and terrestrial communities (Cogalniceanu and Maud 2003) and that toad populations exhibit a negative relationship with a highly disturbed floodplain. However, Kuhn (1993 in Tockner et al 2006) demonstrated that the common toad (*Bufo bufo*) exhibited a pronounced reproductive plasticity including spawning synchronized with hydrology and a fast metamorphosis that allowed it to exploit dynamic and ephemeral

habitats in gravel-bed rivers. Similarly, Tockner et al (2006) found *B. bufo* to be a common species in active and riparian floodplain habitats, serving as an indicator species of island- and large woody debris-associated waters in the active floodplain. Therefore, the low number of toad detections in my study may be an artifact of study design or it may indicate a depressed population level in the Lower Grand River basin due to lack of suitable habitat with sufficient refugia; a richer data set would be required to answer these questions.

Cricket frogs and bullfrogs, the remaining two members of the widely-distributed group, are late breeders typically associated with permanent wetlands, i. e., wetlands that retain water >12 months. This indicates that permanent wetlands are also widely distributed among WRP sites in the Lower Grand River basin. The bullfrog has a more restricted distribution ($\hat{\Psi}_{bul}=0.83$) than the cricket frog ($\hat{\Psi}_{acr}=0.93$) which may indicate either a slightly more restricted distribution of vegetated, permanent wetlands typically associated with bullfrogs on WRP sites in the Lower Grand River basin or an ability by cricket frogs to exploit a wider range of habitat conditions. Cricket frogs metamorphose in <1 year enabling them to successfully exploit seasonal as well as permanent wetlands, whereas bullfrog larvae require one year to complete metamorphosis thereby requiring the longer hydroperiod of more permanent wetlands to successfully complete their life cycle. Additionally, bullfrogs were the only detected species to exhibit a negative relationship with walk-away sites, again indicating a more restricted distribution of suitable habitat. Given that southern leopard frog, leopard frog complex, *Pseudacris* spp., and cricket frogs had the highest occupancy estimates, and that all these species, to varying degrees, are associated with seasonal wetlands and all but *Pseudacris* spp. are

associated with permanent wetlands, it appears both seasonal and permanent wetlands are well-represented on WRP sites in the Lower Grand River basin.

All species in the widely-distributed group are successfully recruiting young into the adult population based on detection of metamorphs; however, due to a sparse data set, American toad was not included in the multi-state occupancy analysis and, therefore, lacks an estimate of reproductive success. Leopard frogs had the highest overall probability of successfully recruiting young ($\hat{\Psi}^1 * \hat{\Psi}^2_{\text{lf}} = 0.94$, $\hat{\Psi}^1 * \hat{\Psi}^2_{\text{plf}} = 0.84$, and $\hat{\Psi}^1 * \hat{\Psi}^2_{\text{sif}} = 0.99$) whereas *Pseudacris* spp. exhibited moderate reproductive success ($\hat{\Psi}^1 * \hat{\Psi}^2_{\text{Pse}} = 0.53$). The lower estimate of reproductive success for *Pseudacris* spp., the species most strongly associated with seasonal wetlands in the widely-distributed group, may indicate that seasonal wetlands, although widely-distributed on WRP sites, do not provide suitable larval development habitat long enough for a higher success rate by this species. Rather unexpectedly, the two species most strongly associated with permanent water, cricket frog and bullfrog, had moderate overall probability of successfully recruiting young ($\hat{\Psi}^1 * \hat{\Psi}^2_{\text{acr}} = 0.48$ and $\hat{\Psi}^1 * \hat{\Psi}^2_{\text{bul}} = 0.49$). Both species were widely distributed on WRP sites in the Lower Grand River basin; however, the probability of detecting successful recruitment, given detection of occupancy (δ), for both species was low ($\hat{\delta}_{\text{acr}} = 0.13$ and $\hat{\delta}_{\text{bul}} = 0.16$). Perhaps our detection methods were not effective at detecting cricket frog and bullfrog metamorphs or, alternatively, perhaps the later timing of their breeding cycle combined with drying conditions prevalent in the third primary sampling period resulted in a lower probability of overall successful recruitment. This is a reasonable explanation for bullfrogs but less so for cricket frogs as they complete their life cycle in less than a year. The highest percent of cricket frog detections were

juveniles; perhaps cricket frog metamorphs, although widely-distributed, have a low detection probability as they rapidly transform into juveniles once all four limbs have emerged resulting in the majority of our detections being juveniles.

The remaining detected species included in the occupancy analysis; small-mouthed salamander, grey treefrog complex, and spring peeper, are early breeders associated with seasonal wetlands (Johnson 2000); the small-mouthed salamander and grey treefrog complex are moderately-distributed whereas the spring peeper is sparsely distributed on WRP sites in the Lower Grand River basin. Similar to the American toad, the spring peeper had a sparse data set and low detection probability estimate ($\hat{p}_{\text{spp}}=0.10$) making the occupancy estimate unreliable (MacKenzie et al. 2002). The sparse data set may be the result of our inability to distinguish chorus frog tadpoles from spring peeper tadpoles early in the field season or it may indicate that habitat generally associated with peepers, i. e., small fishless ponds in a wooded setting located along floodplain terraces, occurs infrequently on WRP sites in the Lower Grand River basin. Unlike the toad; however, which was hypothesized to be common and likely on WRP sites, the peeper was classified as infrequent but likely, so fewer detections of peepers was not as unexpected as was fewer detections of toads. The small-mouthed salamander and grey treefrog were both moderately-distributed among WRP sites providing additional support for the notion that seasonal wetlands are also moderately- to widely- distributed on the sites although perhaps not as widely-distributed as more permanent wetlands. The small-mouthed salamander exhibited a higher overall probability of successfully recruiting young into the adult population ($\hat{\psi}^1 * \hat{\psi}_{\text{sms}}^2=0.61$) than the grey treefrog ($\hat{\psi}^1 * \hat{\psi}_{\text{gtf}}^2=0.36$). This may indicate treefrogs are not as successful recruiting young on WRP sites or it may

be an artifact of study design combined with grey treefrog behavior. Grey treefrogs tend to metamorphose simultaneously with groups of metamorphs congregating in vegetation along the edge of the breeding wetland and disperse with the first rain event. Again, the long interval between primary sampling periods may have resulted in a high number of false negative detections for treefrog metamorphs in the later survey periods which may have resulted in an underestimate of treefrog recruitment success.

A relatively unexpected finding from this study was the number and distribution of fishes on WRP sites with amphibian detections in the Lower Grand River basin. Fish are generally considered regulators of amphibian presence due to their role as predators (Morand and Joly 1995, Wellborn et al. 1996, Hecnar and M'Closkey 1997, Skelly et al. 1999). Centrarchids, in particular, are voracious predators and, yet, Centrarchids were detected on 46 of the 50 WRP sites with amphibian detections. This is consistent with Babbitt et al. (2003) who found neither amphibian species richness nor abundance in permanent wetlands was significantly influenced by presence of predatory fish and Tockner et al. (2006) who found an unexpectedly positive relationship between fish density and amphibian diversity in the active floodplain. Babbitt et al. (2003) noted that variation among regions in the effect of predatory fish on amphibian species richness may depend largely on the predator and prey species pool. Their study was conducted in northern New England, a region that contains a high proportion of amphibian species known to utilize permanent wetlands with predatory fish whereas other regions (e.g., the southeast) contain a higher proportion of species (e.g., treefrogs) whose larvae are vulnerable to predatory fish (Babbitt et al. 2003). Tockner et al. (2006) speculated that, on their study area, large woody debris may facilitate coexistence of amphibians and

predatory fish, two groups that are usually mutually exclusive, as the large woody debris provided refugia for amphibians. They also noted that surveyed ponds were primarily colonized by fish larvae and young fish which most probably do not feed extensively on amphibian larvae. We noted a similar phenomenon although our detection methods were not sized to capture large fish; even so, our captures included a preponderance of age-0 sunfish as indicated by size (≤ 5 cm in length) (Carlander 1997).

Overall, results of the occupancy analysis indicate that hydrological wetland characteristics have been restored to WRP sites in the Lower Grand River basin as indicated by the moderate- to wide- distribution of amphibians associated with both seasonal and permanent wetlands. Leopard frogs, as a group, are the most widely-distributed and reproductively successful group of detected amphibians in this study. The remaining detected species, as results of the multi-state occupancy analysis indicate, exhibited low to moderate to moderately high reproductive success indicating that biological wetland characteristics either have not been restored or are not maintained to the same extent as the hydrological wetland characteristics. Results of the detection probability analysis indicate that the largest proportion of amphibian members from the regional species pool are present on WRP sites in the Lower Grand River basin from early April through mid-July. This implies that suitable wetland habitat, i.e., vegetated seasonal wetlands that gradually dry by mid- to late-summer, must be present throughout this time period if wetland restoration efforts are to restore and maintain biological wetland characteristics and contribute toward preventing, reversing, or stabilizing downward population trends. Additionally, some degree of vegetated, permanent

wetlands must be present if species associated with permanent wetlands are to be supported.

Assessment of Wetland Restoration Efforts

The species richness analysis differed from the occupancy analysis in that it was conducted on a site-by-site basis allowing comparison among sites through the use of summary statistics such as means. Design strategy was not included implicitly in the analysis so comparisons among sites grouped by design strategy were possible and provided another method to evaluate design strategy as a useful descriptor of ecological attributes created by wetland restoration efforts. Defining a regional species pool for this study enabled use of the relative species richness metric described by Cam et al. (2000) and places the richness index within a realistic context of species that may occur on a site. The metric should be greater at sites with favorable ecological conditions (i.e., high quality habitat resulting in recruitment indicating both hydrological and biological wetland characteristics have been restored) than at sites without favorable conditions (i.e., poor quality habitat resulting in no recruitment indicating no or limited hydrological and biological wetland characteristics have been restored) (Cam et al. 2000, Van Horne 2002, Dorazio and Royle 2005, MacKenzie et al. 2006). A poor ranking indicated wetland restoration efforts on a site or group of sites resulted in limited to no restoration of hydrological and biological wetland characteristics and, thus, is making a limited contribution to amphibian conservation by providing habitat for only a few species, generally the more abundant. A fair ranking indicated wetland restoration efforts on a site or group of sites resulted in moderate restoration of hydrological and biological wetland characteristics and is making a contribution to amphibian conservation by

providing habitat for the common and abundant species included in the regional species pool. Species associated with permanent wetlands are likely well represented. A good ranking indicated wetland restoration efforts on a site or group of sites resulted in restoration of hydrological and biological wetland characteristics and is making a contribution to amphibian conservation by providing habitat for not only common and abundant species but also for some of the more infrequently encountered species. Species associated with seasonal wetlands are likely well represented and the site is making a strong contribution toward amphibian conservation. Both the very good and excellent rankings indicated wetland restoration efforts on a site or group of sites resulted in outstanding restoration of hydrological and biological wetland characteristics and is not only providing for the abundant, common, and infrequently encountered species but also the rare to very rare species. A site receiving either a very good or excellent rating would be strongly skewed toward species associated with seasonal wetlands and would make an exceptional contribution to amphibian conservation. It is likely unrealistic to expect that one site could provide suitable habitat for all species in the regional species pool; however, it would not be an unrealistic expectation for all WRP sites within the Lower Grand River basin and represents an achievable objective.

Applying the relative species richness metric to the number of species detected on all WRP sites, collectively, resulted in a naive assessment of good for wetland restoration efforts in the Lower Grand River basin. This ranking implies that, together, all WRP sites are providing suitable habitat conditions for the abundant and common species included in the regional species pool as well as a few of the less common species. Given that this suite of species represents amphibians associated with both seasonal and

permanent wetlands, it also implies that a range of hydrological and biological wetland characteristics are distributed among the sites and restoration efforts are progressing in the desired direction.

Applying the metric to WRP sites by design strategy returned mixed results from the two methods used to determine the relative species richness index. Using the occupancy method, all design strategies received a fair rating, whereas using the capture/recapture method, maximize hydrology and naturalistic sites received a fair rating and walk-away sites received a poor rating. This again, points out that there was greater habitat heterogeneity among the sites than expected and that design strategy was not a useful descriptor of ecological conditions. It also highlights the similarity in wetland conditions between maximize hydrology and naturalistic sites and that restoration efforts applied by both strategies are providing habitat conditions for abundant to common amphibian species. Walk-away sites, however, although more diverse than expected, do not appear to provide the same level of wetland habitat and are likely only providing for the more abundant amphibian species. At this scale, particularly according to results from the capture/recapture method, it appears walk-away sites do not contribute significantly to amphibian conservation and have not resulted in wetland restoration efforts that created the desired hydrological and biological wetland characteristics.

When the relative species richness metric is applied to the site level, however, it becomes a bit clearer that individual sites within each design strategy are contributing, to some degree, toward amphibian conservation, whereas others are not and that the habitat heterogeneity occurring across all sites reduces the relevancy of design strategy as a useful descriptor of wetland restoration efforts (Table 17). Generally, walk-away sites

Table 17. Percent of sites in each design strategy category and all sites, collectively, that were assigned to ratings of wetland restoration success based on relative species richness metric applied to Wetlands Reserve Program sites during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. Estimates generated with the capture/recapture method were based on primary survey period two and estimates generated with the occupancy method were based on all three primary survey periods. Comparison of the percent sites assigned to each rating category by each estimation method provided an indication of how conditions within sites changed over time. Ratings were assigned to each site based on the relative species richness metric as described in text. Sites with a metric value ≤ 0.24 were rated poor, 0.25 – 0.49 were rated fair, 0.50 – 0.69 were rated good, 0.70 - 0.79 were rated very good, and ≥ 0.80 were rated excellent.

Design strategy	Rating by capture/recapture method					Rating by occupancy method				
	poor	fair	good	very good	excellent	poor	fair	good	very good	excellent
maximize hydrology	50	44	6	0	0	11	78	11	0	0
naturalistic	42	47	5	0	5	5	74	16	5	0
walk-away	70	30	0	0	0	31	69	0	0	0
all sites, irrespective of design strategy	52	42	4	0	2	14	74	10	2	0

were dry with limited wetland habitat; 70% received a poor rating using the capture/recapture method; however, three of the walk-away sites were wetter than expected (Figure 22) and received a fair rating by both the occupancy and capture/recapture methods of relative species richness estimation (Figures 19 and 20, respectively). Alternatively, one of the maximize hydrology sites was also wetter than expected but received a poor rating by both methods. The difference in these two examples is the walk-away sites were characterized by wet conditions with grassy vegetation indicating a relatively dynamic hydrological regime that generally provided suitable habitat for a range of amphibians, whereas the maximize hydrology site was characterized by open water indicating a stagnant hydrological regime that generally resulted in poor habitat for amphibians. These examples, again, highlight the heterogeneity present within sites that is not adequately captured by design strategy as a covariate.

The capture/recapture method estimates species richness based on the number of species not detected and places no upper limit on the number of estimated species; the site ranked excellent with this method had a species richness estimate of 18 ($se=6.6$) which exceeds the number of species in the regional species pool and may represent an outlier. The occupancy method does place an upper limit on the richness estimate and represents the proportional occupancy of a site or the probability that a member of the regional species pool is present at a site (MacKenzie et al. 2006). Estimates derived from the capture/recapture method resulted in 52% of the sites, irrespective of design strategy, ranked as poor and 42% ranked as fair (Figure 20), whereas only 14% received a poor ranking under the occupancy method and 74% received a fair rating (Figure 19). The

capture/recapture method resulted in two, or 4% of the sites, ranked good and one, or 2%, of the sites ranked excellent. The occupancy method ranked 10% of the sites good, one or 2% of the sites very good, and none in the excellent range. Collectively, WRP in the Lower Grand River basin is providing wetland habitat conditions that provides for abundant and common species in the regional species pool with limited habitat also provided for a few of the less common species. Although this is not a trivial achievement, there is room for improvement by increasing the number of sites with ratings \geq good. Results from this relative species richness assessment indicate that, whereas 74% of the sites received a fair rating over the course of my field season (7 March – 19 September) (Figure 19), 52% of the sites received a poor rating for the second primary sampling period (14 May – 9 July) (Figure 20). The peak of amphibian breeding and larval development occurs from May through July; this is when a higher proportion of species were present in wetland pools as indicated by the detection analysis (Figure 18). The fact that 52% of the sites were ranked poor when the probability of detecting species was greatest indicates that suitable habitat conditions were not present on these sites at that time. One way to accomplish increasing the number of sites with ratings \geq good is by targeting management actions that result in seasonal wetland conditions present throughout the time needed for species to complete their life history requirements as a higher proportion of the regional species pool is associated with seasonal wetlands (Table 2).

Missing Species and Implications

Members of the regional species pool not detected in this study included tiger salamander (*Ambystoma tigrinum*), Great Plains toad (*Anaxyrus cognatus*),

Woodhouse's toad (*A. woodhousii*), western narrow-mouthed toad (*Gastrophryne olivacea*), crawfish frog (*Lithobates areolatus*), green frog (*L. clamitans*), and wood frog (*L. sylvaticus*). This would seem to infer that a full range of wetland conditions has not been restored on WRP sites in the Lower Grand River basin as not all members of the regional species pool were detected. However, all non-detected species except the tiger salamander are species whose known range includes only a portion of the study area and all except the green frog are associated with geomorphic features either rarely encountered or not represented among the surveyed sites. Great Plains toads and Woodhouse's toads are associated with sandy lowlands more typical of the Missouri River floodplain. The western narrow-mouthed toad is also more typical of the Missouri River floodplain; little is known of its life history in Missouri, however, narrow-mouthed toads are generally associated with grassy conditions at a higher elevation within the floodplain. The potential to detect all three species in my study area was possible but not probable. The remaining non-detected species may well occur in the Lower Grand River basin; however, they may not be present on WRP sites because the sites are located within the floodplain on geomorphic settings with associated habitats not typically inhabited by these species. The tiger salamander and wood frog are more closely associated with wooded sites likely occurring along the base of the floodplain bluff, whereas crawfish frogs are associated with prairies also likely located at a higher elevation in the floodplain. The green frog is perhaps the most surprising non-detected species as it is commonly found in river sloughs, swamps, and marshes; however, the study area represents the northern-most extent of its range (Johnson 2000).

WRP is a voluntary program offering landowners the opportunity to protect, restore, and enhance wetlands on their property (NRCS 2007); as such, although the program can designate priority areas and provide incentives for enrollment in targeted locations, it is still reliant on voluntary participation. Sites surveyed in this study primarily occur in that portion of the floodplain associated with annual floods and characterized by herbaceous and early successional tree species (A and B in Figure 24). This portion of the floodplain presented the greatest challenges to agricultural interests due to frequent floods and extended periods of wet conditions; they were among some of the most difficult sites to convert for agricultural purposes and among the first to be restored. Perhaps the reason that only a portion of the regional species pool was detected is because only a fraction of the floodplain is represented. Species assemblages including amphibians distribute themselves both longitudinally along stream corridors and laterally across riverine floodplains (Vannote et al. 1980, Junk et al. 1989, Tockner et al. 2000, Arscott et al. 2005). Tockner et al. (1998) found that amphibian species richness along a connectivity gradient increased as distance from stream increased such that the highest amphibian species richness occurred in isolated wetlands along the upland edge of the floodplain. They also found that overall species richness estimates of five different species assemblages (aquatic macrophytes, amphibians, mollusks, fish, and odonates) peaked, when all groups were combined, at an intermediate disturbance level along the connectivity gradient. The results of my study represent a truncated gradient, both longitudinally as areas with Missouri River influence were excluded from the study area and laterally as there was no location within the study area in which sites extended contiguously from bluff-to-bluff (Figure 25). The non-detection of seven species in the

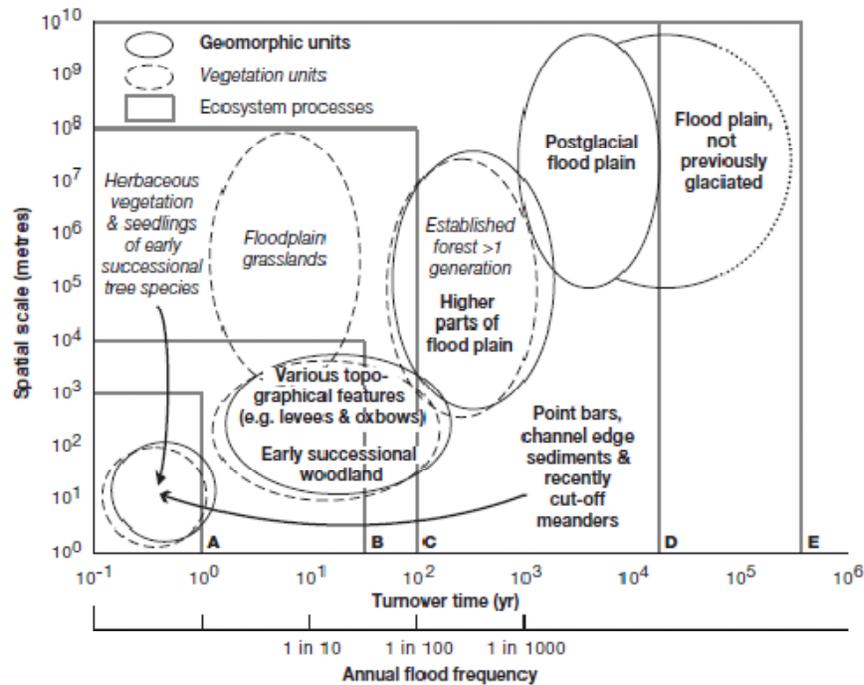


Figure 24. The organization of floodplain components and processes as a spatiotemporal hierarchy (from Tockner and Stanford 2002 after Hughes 1997). A = primary succession of herbaceous vegetation and early successional woody species, associated with annual flood; B=primary and secondary floodplain succession, associated with medium-magnitude/frequency floods; C=long-term floodplain succession, widespread erosion and reworking of sediment, associated with high magnitude/low-frequency floods; D=species migration upstream/downstream, local species postglacial relaxation phenomena on hydrological and sediment inputs to flood plains; and E=species evolution, and changes in biogeographical range, associated with tectonic change, eustatic uplift and climate change.

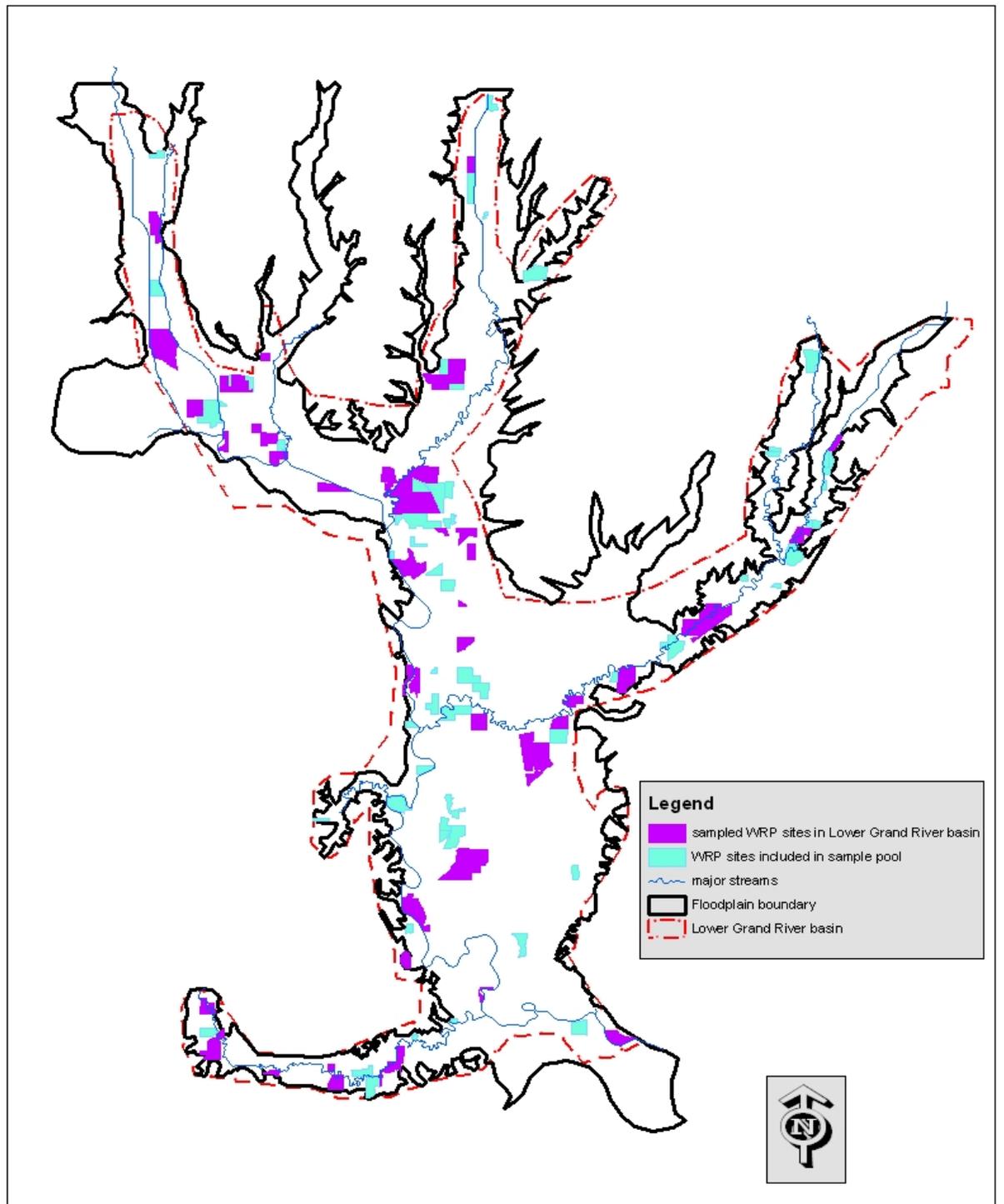


Figure 25. Location of Wetlands Reserve Program properties available for sampling during 2007 field season (March – September) for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri. Sites are shown relative to Lower Grand River floodplain and associated major tributaries.

regional species pool, therefore, indicates either that 1) enrolled WRP sites in the Lower Grand River basin do not represent the entire floodplain gradient, both longitudinally and laterally or 2) that the program is not restoring or providing the full spectrum of wetland conditions within the constraints of the program. Additional investigations are necessary to more fully address these questions.

Lessons learned from this study combined with results from amphibian investigations conducted in riverine-floodplains can aid in designing future studies that identify those wetland features within a site that constitute the ecological drivers determining amphibian distribution (Figure 26). These ecologically relevant attributes become the basis for identifying sampling sites rather than management based criteria. I previously described an approach in the *Occupancy and Wetland Design Strategy* section that would address the specific, within-scale questions of where and why amphibians are distributing themselves on WRP properties in the Lower Grand River basin. An additional, more encompassing approach would be to test the hypothesis that wetland hydroperiod combined with geomorphic setting both longitudinally and laterally across the floodplain are the drivers of amphibian species richness and abundance within a riverine-floodplain system (Pechman et al. 1989, Tockner et al. 2006). This would require selecting reaches within the Lower Grand River basin that vary longitudinally, e. g., lower end of floodplain with Missouri River influence, middle portion of floodplain along Grand River, and upper end of floodplain along a tributary, and sample laterally across the floodplain within each reach. This approach would enable comparing differences that occur spatially both longitudinally and laterally and allow inferences between both WRP sites and non-WRP sites, thus, providing a broader perspective of

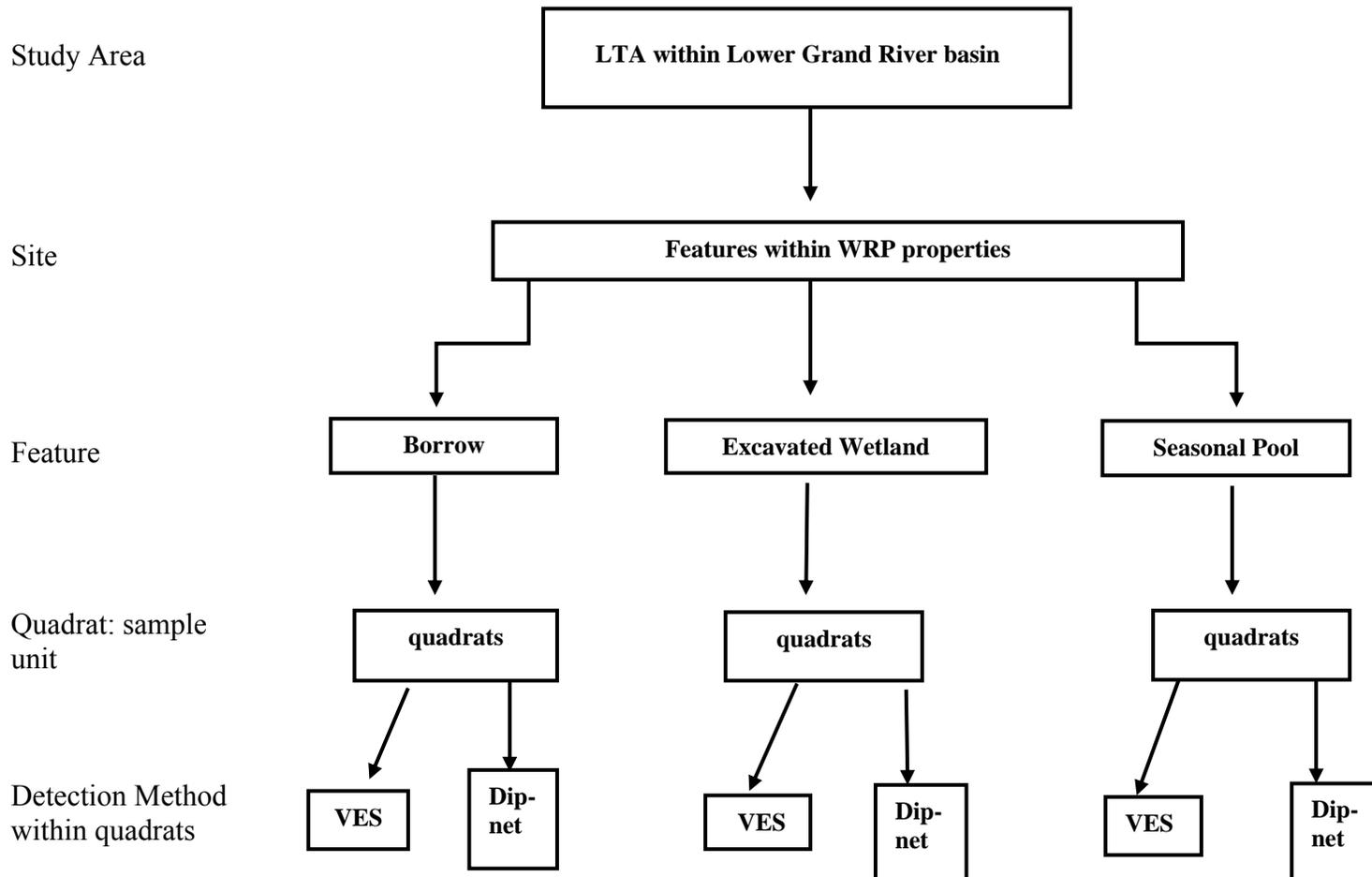


Figure 26. Potential study design for future study aimed at evaluating factors that regulate amphibian distribution at a within-site scale. Each wetland feature within a Wetlands Reserve Program property serves as the primary sample unit, or site. Additional issues to consider include geographic area to survey (this study design suggests limiting the study area to one land type association (LTA); see Nigh and Schroeder 2002), season definition, number of sites, and number of repeat surveys. Lessons learned from amphibian occupancy and species richness study conducted in Lower Grand River basin, north-central Missouri in 2007 indicated a season between 1 April and 1 July, inclusively, will ensure closure for amphibian species included in the regional species pool and at least six repeat, independent surveys would be required during that timeframe to ensure precise estimates. Number of quadrats to survey would be dependent on the size of the feature sampled. The two detection methods include visual encounter surveys (VES) and dip-nets.

both program efforts and how those efforts are progressing in the Lower Grand River basin. Future studies should also explicitly identify geomorphic setting and designed or constructed features and use amphibian (or other appropriate species assemblages) distribution to assess if all parts of a functioning, dynamic riverine floodplain are being restored to the landscape. Such an endeavor would exceed the resources of any one Master's program but, instead, would require a long-term commitment comprised of multiple, related projects that seek to understand the interactions between riverine floodplains, wetland restoration activities within a highly altered landscape, and species responses to both.

CONCLUSION

This study documented that abundant and common amphibian species are using and successfully recruiting young on WRP sites in the Lower Grand River basin. Non-detected members of the regional species pool include four species in which the study area represents the fringe of their known range and three species that are either rare or that occupy habitats rarely encountered within the sampled portion of the floodplain. Design strategy was not a useful covariate in describing heterogeneity in occupancy probability estimates; irrespective of design strategy, occupancy probability estimates of the regional species pool members indicated species representing both seasonal and permanent wetlands are moderately- to widely-distributed on WRP sites thus indicating that hydrological wetland characteristics have been restored to the sites. Although species are successfully recruiting young into the adult population, only leopard frogs had high estimates of reproductive success, whereas the remaining species included in the multi-state occupancy analysis had moderate to low estimates of reproductive success

indicating biological wetland characteristics for these species are lacking. Applying the relative species richness metric indicated that, on average, maximize hydrology and naturalistic sites did a fair job and walk-away sites did a poor job of providing a range of wetland conditions for members of the regional species pool although, at the site scale, there appears to be more habitat heterogeneity than is captured by design strategy. Results from the relative species richness assessment also provide additional support for the idea that biological wetland characteristics may be inadequate during the peak of amphibian breeding and larval development period. Overall, WRP in the Lower Grand River basin has restored hydrological and, to a lesser degree, biological wetland characteristics that provide for abundant and common species in the regional species pool. These results show the value of the WRP at conserving and restoring river-floodplain amphibians, a particularly important contribution given the sudden, inexplicable declines in amphibian populations in other Midwestern states (Lannoo 2005). These efforts also show a significant contribution toward wetland restoration in north-central Missouri; however, opportunities exist to increase both the number of species from the regional species pool using WRP sites and increase the probability of successful recruitment by species currently using WRP sites by targeting management actions that result in seasonal wetland conditions of sufficient duration to ensure successful recruitment.

The NRCS goal for WRP is to achieve the greatest wetland functions and values, along with optimum wildlife habitat, on every acre enrolled in the program (NRCS 2007). Achievement of this goal requires explicitly defining optimum wildlife habitat and determining whether it can realistically be accomplished on every acre enrolled in the

program. Given that the majority of Missouri wetlands are associated with rivers and streams, WRP wetland restoration efforts in the state are, in essence, attempts to restore riverine-floodplains. Riverine-floodplains are, globally, among the most diverse and biologically productive ecosystems (Tockner and Stanford 2002); however, the biotic potential historically associated with riverine systems has been reduced with loss in floodplain structure as occurs under an agricultural regime (Copp 1989, Robinson et al. 2002). The WRP is attempting to restore this floodplain structure with the techniques described throughout this paper to classify design strategy. As this study highlights, however, although habitats may overlap, across the full range of species, habitat requirements vary widely and providing for all species in the regional species pool requires sites that transverse both the longitudinal and lateral floodplain gradient. A site located adjacent to or close to a stream corridor will never provide optimum wildlife habitat for those species adapted to life at the floodplain-upland margin along the hydrological connectivity gradient. If WRP is to fully realize its potential, there must be recognition that optimum wildlife habitat can be defined at multiple spatial and temporal scales that match the landscape setting. For instance, optimum wildlife habitat at a wetland scale is not the same as optimum wildlife habitat at the floodplain scale. Depending on the location of an individual wetland in time and space, providing wetland habitat that accommodates four amphibian species may be optimum for that site but would be woefully inadequate at a floodplain scale which should be providing for a much larger proportion of a regional species pool. The intent of WRP is to convert marginal, flood-prone agricultural lands back into wetlands so enrollment of lands located outside the active floodplain may be impractical or unrealistic. Whereas it may not be possible

to restore riverine-floodplains from bluff-to-bluff on large streams with wide corridors, it is entirely possible on smaller tributaries with more restricted floodplains. Similarly, whereas attaining optimum wildlife habitat on every acre enrolled in the program may not be an achievable objective, providing optimum wildlife habitat for members of a regional species pool within an appropriately defined geography that includes both a longitudinal and lateral gradient represents an objective that is both desirable and attainable.

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APPENDIX A. Definitions used for amphibian species included in regional species pool for occupancy and species richness study conducted in summer 2007 in the Lower Grand River basin, north-central Missouri.

Abbreviations	Amphibian Species
SMS	Small-mouthed salamander
AMT	American toad
ACR	Northern cricket frog
GTF	Grey treefrog complex
SPP	Spring peeper
PSE	<i>Pseudacris</i> spp.
PLF	Plains leopard frog
BUL	American bullfrog
SLF	Southern leopard frog
CEN	Central newt
ETS	Eastern tiger salamander
GPT	Great plains toad
WHT	Woodhouse's toad
NMT	Western narrow-mouthed toad
CRA	Crawfish frog
GRF	Green frog
WFR	Wood frog

APPENDIX B. X-matrix tables with truncated data sets for each member of the regional species pool included in the site occupancy analysis for occupancy and species richness study conducted in summer 2007 in the Lower Grand River basin, north-central Missouri. A table is included for each member of the regional species pool included in the analysis. The species designation and inclusive dates for truncated data sets are indicated above each table. Dots in a row (.) indicate missing data due to the truncated nature of the data sets except for site w18 which was not surveyed in primary survey period one. A one (1) in a column indicates species detection and a zero (0) indicates nondetection. (VES=visual encounter survey; m=maximize hydrology; n=naturalistic; w=walk-away)

Small-mouthed salamander. First detection 12 March 2007 and last detection 6 July 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	1	0	0	0	.	.	.
m10	0	0	0	0	0	0	.	.	.
m12	0	0	0	0	0	0	.	.	.
m18	0	1	0	0	0	0	.	.	.
m21	0	0	0	0	0	0	.	.	.
m22	0	0	0	0	0	0	.	.	.
m26	0	0	0	0	1	0	.	.	.
m28	0	0	0	0	1	1	.	.	.
m32	0	0	0	0	0	1	.	.	.
m34	0	0	0	0	0	0	.	.	.
m35	0	0	0	0	0	1	.	.	.
m38	0	1	0	0	1	1	.	.	.
m43	0	1	1
m44	0	0	0	0	0	0	.	.	.
m46	0	1	1
m47	0	1	0	0	0	0	.	.	.
m49	0	1	0	0	0	1	.	.	.
m50	0	0	1	0	0	0	.	.	.
n2	0	0	1	0	1	0	.	.	.
n3	0	0	0	1	1	1	.	.	.
n5	0	0	0	0	0	0	.	.	.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n6	0	0	0
n7	0	0	0	0	0	0	.	.	.
n8	0	1	1	0	1	1	.	.	.
n10	0	0	0	0	1	1	.	.	.
n11	0	1	0	0	0	0	.	.	.
n13	0	0	0	0	0	1	.	.	.
n16	0	0	0	0	0	0	.	.	.
n17	0	0	1	0	0	1	.	.	.
n18	0	0	0	0	0	0	.	.	.
n20	0	0	1	0	0	0	.	.	.
n22	0	0	0
n24	0	0	0	0	0	0	.	.	.
n26	0	0	0	0	1	1	.	.	.
n29	0	0	0	0	0	0	.	.	.
n31	.	.	.	0	0	1	.	.	.
n33	0	0	1	0	1	1	.	.	.
w1	.	.	.	1	1	1	.	.	.
w3	0	0	0	0	0	0	.	.	.
w4	0	1	1	0	1	1	.	.	.
w9	1	1	1	0	1	1	.	.	.
w10	0	0	0	0	0	0	.	.	.
w12	0	0	1	0	1	1	.	.	.
w13	0	0	0	0	0	0	.	.	.
w14	0	0	0	0	0	0	.	.	.
w18	.	.	.	0	0	0	.	.	.
w19	0	0	0	0	0	0	.	.	.
w20	0	1	1	0	0	0	.	.	.
w21	0	0	0	0	0	0	.	.	.
w22	0	1	1	0	0	0	.	.	.

American toad. First detection 4 April 2007 and last detection 8 August 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	0	0	0	1	0	0
m10	.	.	.	0	0	0	0	0	0
m12	.	.	.	0	0	0	0	0	0
m18	0	0	0	0	0	0	.	.	.
m21	.	.	.	0	0	0	.	.	.
m22	.	.	.	0	0	0	.	.	.
m26	.	.	.	0	0	0	.	.	.
m28	.	.	.	0	0	0	.	.	.
m32	0	0	0	0	0	0	.	.	.
m34	.	.	.	0	0	0	.	.	.
m35	0	0	0	0	0	0	.	.	.
m38	.	.	.	0	0	0	.	.	.
m43	0	1	0	0	0	0	.	.	.
m44	.	.	.	0	0	0	.	.	.
m46	0	0	0	0	0	0	.	.	.
m47	0	0	0	1	1	0	.	.	.
m49	.	.	.	1	0	0	1	0	0
m50	0	0	0	0	0	0	.	.	.
n2	0	0	0	1	0	0	.	.	.
n3	.	.	.	0	0	1	.	.	.
n5	0	1	0	0	0	0	0	0	0
n6	0	0	1	0	0	0	0	0	0
n7	.	.	.	0	0	0	0	0	0
n8	.	.	.	0	0	0	.	.	.
n10	0	0	0	0	0	0	.	.	.
n11	0	0	0	1	0	0	.	.	.
n13	0	0	0	0	0	0	.	.	.
n16	0	0	0	0	0	0	0	0	0

Northern cricket frog. First detection 12 March 2007 and last detection 9 September 2007 (last day of field season); season exclusive of first date and inclusive of last date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	1	0	0	1	0	0	1	1	1
m10	1	0	0	0	1	0	1	0	0
m12	0	0	0	0	0	0	1	1	1
m18	0	0	0	0	0	0	1	0	0
m21	0	0	0	0	0	1	0	0	0
m22	1	0	0	0	0	0	1	0	0
m26	1	0	0	0	0	0	0	1	0
m28	0	0	0	1	0	0	1	1	1
m32	0	0	0	1	1	0	1	0	0
m34	1	0	0	1	1	1	1	1	1
m35	1	0	0	0	0	1	1	0	0
m38	1	0	0	1	0	0	1	0	0
m43	1	0	0	1	1	1	1	0	0
m44	1	0	0	1	1	1	1	0	0
m46	1	0	0	1	1	1	1	0	0
m47	1	0	0	1	1	1	0	0	0
m49	1	0	0	1	0	0	1	0	0
m50	1	0	0	1	1	1	1	1	1
n2	1	0	0	1	1	1	1	0	0
n3	0	0	0	0	1	1	1	0	1
n5	0	0	0	1	1	1	1	0	1
n6	1	0	1	1	0	1	1	1	1
n7	1	0	0	1	1	1	1	1	1
n8	0	0	0	0	0	0	1	0	0
n10	0	0	0	1	0	1	1	1	0
n11	1	0	0	1	1	1	1	1	0
n13	0	0	0	1	0	0	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	1	1	0	1	1	1	1	1	1
n17	0	0	0	0	0	0	1	0	0
n18	1	0	0	0	1	1	1	0	1
n20	1	0	1	1	1	1	1	1	0
n22	1	0	0	1	0	0	1	0	1
n24	0	0	0	1	0	1	1	0	1
n26	1	0	0	1	0	1	1	1	0
n29	1	0	0	1	0	1	1	1	0
n31	.	.	.	0	0	1	1	0	1
n33	0	0	0	1	0	0	1	0	0
w1	.	.	.	1	1	0	1	0	0
w3	0	0	0	1	1	1	1	1	0
w4	1	0	0	0	0	0	1	0	0
w9	1	0	0	0	0	0	1	1	1
w10	0	0	1	0	0	0	1	0	0
w12	0	0	0	0	1	1	1	0	0
w13	0	0	0	0	0	0	0	0	0
w14	0	0	0	0	0	0	1	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	0	0	1	0	0	0
w20	0	0	0	0	0	0	0	0	0
w21	0	0	0	0	0	0	0	0	0
w22	0	0	0	1	0	0	0	0	1

Grey treefrog complex. First detection 2 April 2007 and last detection 27 August 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	1	1	0	0	0	0
m10	.	.	.	1	1	1	0	0	0
m12	.	.	.	0	0	0	0	0	0
m18	0	0	0	0	1	1	.	.	.
m21	.	.	.	0	0	0	.	.	.
m22	.	.	.	0	0	0	.	.	.
m26	.	.	.	0	0	0	0	0	0
m28	.	.	.	0	0	1	0	0	0
m32	0	0	0	1	1	1	0	0	0
m34	.	.	.	0	1	1	0	0	0
m35	0	0	0	0	0	0	.	.	.
m38	0	0	0	1	1	1	0	0	0
m43	0	0	0	1	0	1	.	.	.
m44	0	0	0	1	1	1	.	.	.
m46	0	0	0	0	1	0	.	.	.
m47	0	0	0	1	1	1	.	.	.
m49	.	.	.	0	0	0	0	0	0
m50	0	0	0	0	1	1	0	0	0
n2	0	0	0	0	1	1	0	0	0
n3	.	.	.	1	1	1	.	.	.
n5	0	0	1	1	0	0	0	0	0
n6	0	0	0	0	0	0	1	0	0
n7	.	.	.	0	1	1	0	0	0
n8	.	.	.	1	1	1	.	.	.
n10	0	0	0	0	1	1	0	0	0
n11	0	0	0	0	0	0	0	0	0
n13	0	0	0	0	0	0	.	.	.
n16	0	0	0	1	1	1	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	.	.	.	0	0	1	0	0	0
n18	.	.	.	0	1	1	0	0	0
n20	0	0	1	0	1	1	0	0	0
n22	.	.	.	0	0	0	.	.	.
n24	0	0	0	0	0	0	.	.	.
n26	0	0	0	0	0	0	.	.	.
n29	0	0	0	1	1	1	0	0	0
n31	.	.	.	0	1	1	.	.	.
n33	.	.	.	0	1	1	.	.	.
w1	.	.	.	0	1	1	.	.	.
w3	.	.	.	1	1	1	1	0	0
w4	0	0	0	0	0	0	.	.	.
w9	0	0	0	0	0	0	.	.	.
w10	.	.	.	0	0	0	1	0	0
w12	0	0	0	0	1	1	0	0	0
w13	.	.	.	0	0	0	0	0	0
w14	0	0	0	0	1	0	0	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	1	0	0	0	0	0
w20	0	0	0	0	0	0	0	0	0
w21	0	0	0	0	0	0	0	0	0
w22	0	0	1	0	0	0	0	0	0

Spring peeper. First detection 12 March 2007 and last detection 9 July 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	0	0	0	0	.	.	.
m10	0	0	0	0	0	0	.	.	.
m12	0	0	0	0	0	0	.	.	.
m18	0	0	0	0	0	0	.	.	.
m21	0	0	0	0	0	0	.	.	.
m22	0	0	0	0	0	0	.	.	.
m26	0	0	0	0	0	0	.	.	.
m28	0	0	0	0	0	0	.	.	.
m32	0	0	0	0	0	0	.	.	.
m34	0	0	0	0	0	0	.	.	.
m35	0	0	0	0	0	0	.	.	.
m38	0	0	0	0	1	0	.	.	.
m43	0	0	0	0	0	0	.	.	.
m44	0	0	0	0	0	0	.	.	.
m46	0	0	0	0	0	0	.	.	.
m47	0	0	0	0	0	0	.	.	.
m49	0	0	0	1	0	0	.	.	.
m50	0	0	0	0	0	0	.	.	.
n2	0	0	0	0	0	0	.	.	.
n3	0	0	0	1	1	1	.	.	.
n5	0	0	0	0	0	0	.	.	.
n6	0	1	0
n7	0	0	0	0	0	0	.	.	.
n8	0	0	1	0	0	0	.	.	.
n10	0	0	0	0	0	0	.	.	.
n11	0	0	0	0	0	0	.	.	.
n13	0	0	0	0	0	0	.	.	.
n16	0	0	0	0	0	0	.	.	.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	0	0	0	0	0	0	.	.	.
n18	0	0	0	0	0	0	.	.	.
n20	0	0	1	0	0	0	.	.	.
n22	0	0	0
n24	0	0	0	0	0	0	.	.	.
n26	0	0	0	0	0	0	.	.	.
n29	0	0	0	0	0	0	.	.	.
n31	.	.	.	0	0	0	.	.	.
n33	1	0	0	0	0	0	.	.	.
w1	.	.	.	1	0	0	.	.	.
w3	0	0	0	0	0	0	.	.	.
w4	0	0	0	0	0	0	.	.	.
w9	0	0	0	0	0	0	.	.	.
w10	0	0	0	0	0	0	.	.	.
w12	0	0	0	0	0	0	.	.	.
w13	0	0	0	0	0	0	.	.	.
w14	0	0	0	0	0	0	.	.	.
w18	.	.	.	0	0	0	.	.	.
w19	0	0	0	0	0	0	.	.	.
w20	0	0	0	0	0	0	.	.	.
w21	0	0	0	0	0	0	.	.	.
w22	0	0	1	0	0	0	.	.	.

Pseudacris spp. First detection 12 March 2007 and last detection 2 August 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	0	0	0	0	0	0	0
m10	0	0	0	1	0	0	0	0	0
m12	0	0	1	0	0	0	0	0	0
m18	0	1	1	0	0	0	.	.	.
m21	0	0	0	0	0	0	.	.	.
m22	1	0	1	0	0	0	.	.	.
m26	0	0	0	0	0	0	.	.	.
m28	0	0	0	0	1	1	.	.	.
m32	0	0	0	0	0	0	.	.	.
m34	0	0	0	0	0	0	.	.	.
m35	0	0	0	0	1	0	.	.	.
m38	1	0	1	1	0	0	.	.	.
m43	0	1	1	0	0	0	.	.	.
m44	0	1	1	0	0	0	.	.	.
m46	0	1	1	0	0	0	.	.	.
m47	0	1	1	0	0	0	.	.	.
m49	0	0	1	0	0	0	1	0	0
m50	0	1	1	0	0	0	.	.	.
n2	0	0	0	1	0	0	.	.	.
n3	0	0	0	0	1	1	.	.	.
n5	0	1	1	0	0	0	0	0	0
n6	0	1	1	0	0	0	0	0	0
n7	0	0	0	0	0	0	0	0	0
n8	0	0	1	1	0	0	.	.	.
n10	0	0	0	0	0	0	.	.	.
n11	0	1	0	0	0	0	.	.	.
n13	0	0	0	0	1	1	.	.	.
n16	0	0	1	0	0	0	0	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	0	0	1	0	0	0	0	0	0
n18	0	0	0	0	0	0	0	0	0
n20	0	0	1	0	0	0	.	.	.
n22	0	0	0	1	0	0	.	.	.
n24	0	1	1	0	0	1	.	.	.
n26	0	1	0	1	1	1	.	.	.
n29	0	0	0	1	0	0	.	.	.
n31	.	.	.	1	1	1	.	.	.
n33	0	0	0	1	0	0	.	.	.
w1	.	.	.	1	1	1	.	.	.
w3	0	0	1	0	0	0	0	0	0
w4	0	0	1	0	0	1	.	.	.
w9	0	0	0	0	0	0	.	.	.
w10	0	0	0	0	0	0	.	.	.
w12	0	0	1	0	0	0	.	.	.
w13	0	0	0	0	0	0	.	.	.
w14	0	0	0	0	0	0	.	.	.
w18	.	.	.	0	0	0	.	.	.
w19	0	0	1	0	0	0	.	.	.
w20	0	1	1	0	0	0	.	.	.
w21	0	1	0	0	0	0	.	.	.
w22	0	1	1	0	0	0	.	.	.

Plains leopard frog. First detection 12 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	0	0	1	0	0	0
m10	.	.	.	1	0	1	1	0	0
m12	.	.	.	0	0	1	0	0	0
m18	0	1	0	0	0	0	0	0	0
m21	0	0	0	0	0	0	0	0	0
m22	0	0	0	0	0	0	0	0	0
m26	0	0	0	0	0	0	0	0	0
m28	0	0	0	0	0	0	0	0	0
m32	0	0	0	0	0	0	0	0	0
m34	0	0	0	0	0	0	0	0	0
m35	0	0	1	0	0	0	1	0	0
m38	0	0	0	0	0	0	1	0	0
m43	0	0	0	0	0	0	1	0	0
m44	0	0	0	1	0	1	0	0	0
m46	0	0	0	0	0	0	0	0	0
m47	0	0	0	1	0	0	0	0	0
m49	0	0	1	1	1	1	0	0	0
m50	0	0	0	0	0	0	0	0	0
n2	0	0	0	0	0	0	1	0	0
n3	.	.	.	0	0	0	0	0	1
n5	0	0	0	0	0	1	1	0	1
n6	0	0	1	0	0	0	1	0	0
n7	0	0	1	0	0	0	0	0	0
n8	0	0	0	0	1	0	1	0	0
n10	0	0	0	0	0	0	0	0	0
n11	1	0	0	0	1	0	0	0	0
n13	0	0	0	0	0	0	1	0	0

American bullfrog. First detection 14 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	0	0	0	1	1	1
m10	.	.	.	1	0	1	1	0	0
m12	.	.	.	1	0	1	1	0	0
m18	0	0	0	0	0	0	1	0	0
m21	1	0	1	0	0	0	1	0	1
m22	0	0	0	0	0	1	1	0	0
m26	0	0	1	0	0	0	0	0	0
m28	1	0	1	0	0	0	0	1	1
m32	0	0	0	0	0	1	1	0	0
m34	0	0	0	0	0	1	1	0	1
m35	0	0	1	0	0	1	0	0	0
m38	0	0	0	0	1	0	0	0	0
m43	1	0	1	1	0	1	1	1	1
m44	0	0	1	1	0	1	1	1	1
m46	1	1	1	1	0	1	1	0	0
m47	0	0	0	0	0	0	0	0	0
m49	0	0	0	0	0	0	1	0	1
m50	1	1	1	1	0	1	1	1	1
n2	0	0	1	1	0	0	1	0	0
n3	1	0	1	0	0	0	1	1	1
n5	0	0	0	1	0	1	1	1	1
n6	1	0	1	1	0	1	1	1	1
n7	1	1	1	1	1	1	1	1	1
n8	0	0	0	0	0	0	1	0	0
n10	0	0	1	1	0	1	1	1	1
n11	1	1	1	1	1	1	1	1	1
n13	0	0	0	0	0	0	0	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	0	0	0	1	0	1	1	0	1
n17	0	0	0	0	0	0	0	0	0
n18	0	0	0	0	0	0	1	0	1
n20	1	1	1	1	0	1	1	1	1
n22	0	0	1	0	0	0	1	0	1
n24	0	0	0	0	0	1	0	0	1
n26	0	0	0	0	0	0	0	0	0
n29	0	0	1	0	0	1	1	0	1
n31	.	.	.	0	0	0	1	1	1
n33	0	0	0	0	0	0	1	0	1
w1	.	.	.	1	1	1	1	1	1
w3	0	0	0	1	0	1	1	1	1
w4	1	0	1	1	0	0	1	0	1
w9	0	0	0	1	0	0	1	0	1
w10	0	0	0	0	0	0	1	0	0
w12	0	0	0	0	0	1	1	1	1
w13	0	0	0	0	0	0	0	0	0
w14	0	0	0	0	0	0	0	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	0	0	0	0	0	0
w20	0	0	0	1	0	0	0	0	1
w21	0	0	0	0	0	0	0	0	0
w22	0	0	0	0	0	0	1	1	1

Southern leopard frog. First detection 12 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	0	0	0	1	0	1	0
m10	0	0	0	1	0	1	0	0	0
m12	0	0	0	0	0	1	0	0	0
m18	0	0	0	1	0	1	0	0	0
m21	0	0	0	0	0	0	0	0	1
m22	0	0	0	1	0	0	0	0	0
m26	1	0	0	1	0	1	0	0	0
m28	0	0	0	1	0	0	0	0	0
m32	0	0	0	0	0	0	0	0	0
m34	0	0	1	1	1	0	0	0	0
m35	0	0	0	0	0	0	0	0	1
m38	0	0	0	0	1	0	0	0	0
m43	0	0	0	0	0	1	0	1	0
m44	0	0	0	0	1	1	0	0	0
m46	0	0	0	1	0	0	1	0	0
m47	0	0	0	1	1	1	0	0	0
m49	0	0	1	1	0	0	0	0	0
m50	0	0	0	0	0	0	0	0	0
n2	0	0	0	0	0	1	1	0	0
n3	0	0	0	0	0	1	0	0	0
n5	0	0	0	0	0	1	0	0	1
n6	0	0	1	0	0	1	1	0	0
n7	0	0	0	0	0	0	1	0	0
n8	0	0	1	1	1	1	1	0	0
n10	0	0	0	0	1	1	0	0	0
n11	1	1	0	0	0	0	0	0	0
n13	0	0	0	0	0	0	0	0	1

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	0	0	0	0	1	1	0	0	1
n17	0	0	0	1	0	1	0	0	1
n18	0	0	0	0	0	0	1	0	0
n20	0	0	0	0	0	0	1	0	0
n22	0	0	0	0	0	0	1	0	0
n24	0	0	0	0	0	0	1	0	0
n26	0	0	0	0	0	0	0	0	0
n29	0	0	0	1	0	1	0	0	0
n31	.	.	.	0	0	1	1	0	1
n33	0	0	0	1	0	1	0	0	0
w1	.	.	.	0	0	0	1	0	1
w3	0	0	0	1	0	1	1	0	0
w4	0	0	0	0	0	0	0	0	0
w9	0	0	0	0	0	0	0	0	0
w10	1	0	1	1	1	1	0	0	0
w12	0	0	0	0	1	1	0	0	0
w13	0	0	0	1	0	0	1	0	0
w14	0	0	0	0	1	0	0	0	1
w18	.	.	.	0	1	0	0	0	0
w19	0	0	0	0	0	1	0	0	0
w20	0	0	0	0	0	1	1	0	0
w21	0	0	0	1	0	0	0	0	0
w22	1	0	1	0	0	0	0	0	0

Leopard frog complex. First detection 7 March 2007 and last detection 9 September 2007; season inclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	1	1	1	1	1	0	1
m10	0	0	0	1	1	1	1	0	0
m12	0	0	0	1	0	0	1	0	0
m18	0	1	1	1	1	1	1	0	0
m21	0	0	1	0	1	1	1	0	0
m22	0	0	0	1	0	1	1	0	0
m26	1	0	0	1	1	1	1	0	0
m28	0	0	0	1	1	1	1	0	0
m32	0	0	0	1	1	1	1	0	0
m34	0	0	0	1	1	1	1	0	0
m35	1	0	0	0	1	1	1	0	0
m38	0	1	0	1	1	1	1	0	0
m43	0	1	1	1	0	1	1	0	0
m44	1	1	1	0	1	1	1	0	0
m46	1	1	1	1	1	1	1	0	0
m47	0	0	1	1	1	1	0	0	0
m49	1	0	0	1	0	1	1	0	0
m50	1	1	1	1	1	1	0	0	0
n2	0	1	1	0	1	1	1	0	0
n3	0	0	0	1	1	1	1	0	0
n5	0	1	1	1	1	1	1	1	1
n6	1	1	1	1	0	0	1	0	0
n7	1	0	1	0	1	1	1	0	1
n8	0	0	0	1	1	1	1	0	0
n10	0	0	1	0	1	1	1	0	0
n11	1	1	1	1	1	1	0	0	1
n13	0	0	1	0	1	1	1	0	0
n16	0	1	1	1	1	1	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	0	0	1	1	1	1	1	0	0
n18	0	0	0	1	1	1	1	0	0
n20	0	1	1	1	1	1	1	0	0
n22	1	0	1	1	1	1	1	1	1
n24	0	1	1	0	1	1	1	0	0
n26	0	0	1	1	1	1	0	0	0
n29	0	0	0	1	1	1	0	0	1
n31	0	0	1	0	1	1	1	0	0
n33	0	0	1	1	1	1	1	0	0
w1	0	0	0	1	1	1	1	0	0
w3	0	0	0	1	0	1	1	1	1
w4	0	0	1	0	0	1	0	0	0
w9	1	0	0	0	1	1	1	0	0
w10	1	0	0	1	1	1	1	0	0
w12	0	0	1	0	1	1	1	0	0
w13	0	0	0	1	0	0	0	0	0
w14	0	0	0	1	1	0	1	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	1	0	1	1	0	0
w20	0	1	1	1	0	1	1	0	0
w21	0	0	0	0	0	0	0	0	0
w22	1	1	1	1	0	1	1	0	0

APPENDIX C. X-matrix tables with truncated data sets for each member of the regional species pool included in the multi-state occupancy analysis for occupancy and species richness study conducted in summer 2007 in the Lower Grand River basin, north-central Missouri. A table is included for each member of the regional species pool included in the analysis. The species designation and inclusive dates for truncated data sets are indicated above each table. Dots in a row (.) indicate missing data due to the truncated nature of the data sets except for site w18 which was not surveyed in primary survey period one. A one (1) in a column indicates species detection, a two (2) in a column indicates metamorph detection, and a zero (0) indicates nondetection. (VES=visual encounter survey; m=maximize hydrology; n=naturalistic; w=walk-away)

Small-mouthed salamander. First detection 12 March 2007 and last detection 6 July 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	1	0	0	0	.	.	.
m10	0	0	0	0	0	0	.	.	.
m12	0	0	0	0	0	0	.	.	.
m18	0	1	0	0	0	0	.	.	.
m21	0	0	0	0	0	0	.	.	.
m22	0	0	0	0	0	0	.	.	.
m26	0	0	0	0	1	0	.	.	.
m28	0	0	0	0	1	1	.	.	.
m32	0	0	0	0	0	1	.	.	.
m34	0	0	0	0	0	0	.	.	.
m35	0	0	0	0	0	2	.	.	.
m38	0	1	0	0	1	1	.	.	.
m43	0	1	1
m44	0	0	0	0	0	0	.	.	.
m46	0	1	1
m47	0	1	0	0	0	0	.	.	.
m49	0	1	0	0	0	2	.	.	.
m50	0	0	1	0	0	0	.	.	.
n2	0	0	1	0	1	0	.	.	.
n3	0	0	0	1	1	2	.	.	.
n5	0	0	0	0	0	0	.	.	.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n6	0	0	0
n7	0	0	0	0	0	0	.	.	.
n8	0	1	1	0	2	2	.	.	.
n10	0	0	0	0	2	2	.	.	.
n11	0	1	0	0	0	0	.	.	.
n13	0	0	0	0	0	1	.	.	.
n16	0	0	0	0	0	0	.	.	.
n17	0	0	1	0	0	2	.	.	.
n18	0	0	0	0	0	0	.	.	.
n20	0	0	1	0	0	0	.	.	.
n22	0	0	0
n24	0	0	0	0	0	0	.	.	.
n26	0	0	0	0	1	1	.	.	.
n29	0	0	0	0	0	0	.	.	.
n31	.	.	.	0	0	1	.	.	.
n33	0	0	1	0	1	2	.	.	.
w1	.	.	.	1	1	1	.	.	.
w3	0	0	0	0	0	0	.	.	.
w4	0	1	1	0	1	1	.	.	.
w9	1	1	1	0	1	1	.	.	.
w10	0	0	0	0	0	0	.	.	.
w12	0	0	1	0	1	1	.	.	.
w13	0	0	0	0	0	0	.	.	.
w14	0	0	0	0	0	0	.	.	.
w18	.	.	.	0	0	0	.	.	.
w19	0	0	0	0	0	0	.	.	.
w20	0	1	1	0	0	0	.	.	.
w21	0	0	0	0	0	0	.	.	.
w22	0	1	1	0	0	0	.	.	.

Northern cricket frog. First detection 12 March 2007 and last detection 9 September 2007 (last day of field season); season exclusive of first date and inclusive of last date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	1	0	0	1	0	0	1	1	1
m10	1	0	0	0	1	0	2	0	0
m12	0	0	0	0	0	0	1	1	1
m18	0	0	0	0	0	0	1	0	0
m21	0	0	0	0	0	1	0	0	0
m22	1	0	0	0	0	0	1	0	0
m26	1	0	0	0	0	0	0	1	0
m28	0	0	0	1	0	0	1	1	1
m32	0	0	0	1	1	0	1	0	0
m34	1	0	0	1	2	1	1	2	1
m35	1	0	0	0	0	1	1	0	0
m38	1	0	0	1	0	0	1	0	0
m43	1	0	0	1	2	1	1	0	0
m44	1	0	0	1	1	1	1	0	0
m46	1	0	0	1	1	1	1	0	0
m47	1	0	0	2	2	2	0	0	0
m49	1	0	0	1	0	0	1	0	0
m50	1	0	0	1	1	1	1	2	1
n2	1	0	0	1	1	1	1	0	0
n3	0	0	0	0	1	1	1	0	2
n5	0	0	0	1	1	1	1	0	1
n6	1	0	1	1	0	1	1	1	1
n7	1	0	0	1	1	1	1	1	1
n8	0	0	0	0	0	0	1	0	0
n10	0	0	0	1	0	1	1	1	0
n11	1	0	0	2	2	1	1	2	0
n13	0	0	0	1	0	0	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	1	1	0	1	1	1	2	1	2
n17	0	0	0	0	0	0	1	0	0
n18	1	0	0	0	1	1	1	0	1
n20	1	0	1	1	1	1	1	1	0
n22	1	0	0	1	0	0	1	0	1
n24	0	0	0	1	0	1	2	0	1
n26	1	0	0	1	0	1	1	1	0
n29	1	0	0	1	0	1	1	1	0
n31	.	.	.	0	0	1	1	0	1
n33	0	0	0	1	0	0	1	0	0
w1	.	.	.	1	1	0	1	0	0
w3	0	0	0	1	2	1	1	1	0
w4	1	0	0	0	0	0	1	0	0
w9	1	0	0	0	0	0	1	1	1
w10	0	0	1	0	0	0	1	0	0
w12	0	0	0	0	1	1	1	0	0
w13	0	0	0	0	0	0	0	0	0
w14	0	0	0	0	0	0	1	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	0	0	1	0	0	0
w20	0	0	0	0	0	0	0	0	0
w21	0	0	0	0	0	0	0	0	0
w22	0	0	0	1	0	0	0	0	1

Grey treefrog complex. First detection 2 April 2007 and last detection 27 August 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	1	2	0	0	0	0
m10	.	.	.	1	1	1	0	0	0
m12	.	.	.	0	0	0	0	0	0
m18	0	0	0	0	2	1	.	.	.
m21	.	.	.	0	0	0	.	.	.
m22	.	.	.	0	0	0	.	.	.
m26	.	.	.	0	0	0	0	0	0
m28	.	.	.	0	0	1	0	0	0
m32	0	0	0	1	1	1	0	0	0
m34	.	.	.	0	1	1	0	0	0
m35	0	0	0	0	0	0	.	.	.
m38	0	0	0	1	1	1	0	0	0
m43	0	0	0	1	0	1	.	.	.
m44	0	0	0	2	2	1	.	.	.
m46	0	0	0	0	1	0	.	.	.
m47	0	0	0	2	2	2	.	.	.
m49	.	.	.	0	0	0	0	0	0
m50	0	0	0	0	1	1	0	0	0
n2	0	0	0	0	1	1	0	0	0
n3	.	.	.	1	2	2	.	.	.
n5	0	0	1	1	0	0	0	0	0
n6	0	0	0	0	0	0	1	0	0
n7	.	.	.	0	1	1	0	0	0
n8	.	.	.	1	1	1	.	.	.
n10	0	0	0	0	2	2	0	0	0
n11	0	0	0	0	0	0	0	0	0
n13	0	0	0	0	0	0	.	.	.
n16	0	0	0	2	2	1	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	.	.	.	0	0	1	0	0	0
n18	.	.	.	0	2	1	0	0	0
n20	0	0	1	0	1	1	0	0	0
n22	.	.	.	0	0	0	.	.	.
n24	0	0	0	0	0	0	.	.	.
n26	0	0	0	0	0	0	.	.	.
n29	0	0	0	2	2	1	0	0	0
n31	.	.	.	0	2	2	.	.	.
n33	.	.	.	0	1	1	.	.	.
w1	.	.	.	0	1	1	.	.	.
w3	.	.	.	1	2	2	1	0	0
w4	0	0	0	0	0	0	.	.	.
w9	0	0	0	0	0	0	.	.	.
w10	.	.	.	0	0	0	1	0	0
w12	0	0	0	0	1	2	0	0	0
w13	.	.	.	0	0	0	0	0	0
w14	0	0	0	0	2	0	0	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	1	0	0	0	0	0
w20	0	0	0	0	0	0	0	0	0
w21	0	0	0	0	0	0	0	0	0
w22	0	0	1	0	0	0	0	0	0

Pseudacris spp. First detection 12 March 2007 and last detection 2 August 2007; season exclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	0	0	0	0	0	0	0
m10	0	0	0	1	0	0	0	0	0
m12	0	0	1	0	0	0	0	0	0
m18	0	1	1	0	0	0	.	.	.
m21	0	0	0	0	0	0	.	.	.
m22	1	0	1	0	0	0	.	.	.
m26	0	0	0	0	0	0	.	.	.
m28	0	0	0	0	1	1	.	.	.
m32	0	0	0	0	0	0	.	.	.
m34	0	0	0	0	0	0	.	.	.
m35	0	0	0	0	1	0	.	.	.
m38	1	0	1	1	0	0	.	.	.
m43	0	1	1	0	0	0	.	.	.
m44	0	1	1	0	0	0	.	.	.
m46	0	1	1	0	0	0	.	.	.
m47	0	1	1	0	0	0	.	.	.
m49	0	0	1	0	0	0	1	0	0
m50	0	1	1	0	0	0	.	.	.
n2	0	0	0	2	0	0	.	.	.
n3	0	0	0	0	1	2	.	.	.
n5	0	1	1	0	0	0	0	0	0
n6	0	1	1	0	0	0	0	0	0
n7	0	0	0	0	0	0	0	0	0
n8	0	0	1	2	0	0	.	.	.
n10	0	0	0	0	0	0	.	.	.
n11	0	1	0	0	0	0	.	.	.
n13	0	0	0	0	1	2	.	.	.
n16	0	0	1	0	0	0	0	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	0	0	1	0	0	0	0	0	0
n18	0	0	0	0	0	0	0	0	0
n20	0	0	1	0	0	0	.	.	.
n22	0	0	0	1	0	0	.	.	.
n24	0	1	1	0	0	1	.	.	.
n26	0	1	0	1	1	2	.	.	.
n29	0	0	0	1	0	0	.	.	.
n31	.	.	.	2	1	2	.	.	.
n33	0	1	0	2	0	0	.	.	.
w1	.	.	.	2	1	2	.	.	.
w3	0	0	1	0	0	0	0	0	0
w4	0	0	1	0	0	1	.	.	.
w9	0	0	0	0	0	0	.	.	.
w10	0	0	0	0	0	0	.	.	.
w12	0	0	1	0	0	0	.	.	.
w13	0	0	0	0	0	0	.	.	.
w14	0	0	0	0	0	0	.	.	.
w18	.	.	.	0	0	0	.	.	.
w19	0	1	1	0	0	0	.	.	.
w20	0	1	2	0	0	0	.	.	.
w21	0	1	0	0	0	0	.	.	.
w22	0	1	1	0	0	0	.	.	.

Plains leopard frog. First detection 12 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	0	0	1	0	0	0
m10	.	.	.	1	0	2	1	0	0
m12	.	.	.	0	0	1	0	0	0
m18	0	1	0	0	0	0	0	0	0
m21	0	0	0	0	0	0	0	0	0
m22	0	0	0	0	0	0	0	0	0
m26	0	0	0	0	0	0	0	0	0
m28	0	0	0	0	0	0	0	0	0
m32	0	0	0	0	0	0	0	0	0
m34	0	0	0	0	0	0	0	0	0
m35	0	0	1	0	0	0	1	0	0
m38	0	0	0	0	0	0	1	0	0
m43	0	0	0	0	0	0	1	0	0
m44	0	0	0	1	0	2	0	0	0
m46	0	0	0	0	0	0	0	0	0
m47	0	0	0	2	0	0	0	0	0
m49	0	0	1	1	1	2	0	0	0
m50	0	0	0	0	0	0	0	0	0
n2	0	0	0	0	0	0	2	0	0
n3	.	.	.	0	0	0	0	0	1
n5	0	0	0	0	0	2	1	0	1
n6	0	0	1	0	0	0	1	0	0
n7	0	0	1	0	0	0	0	0	0
n8	0	0	0	0	2	0	1	0	0
n10	0	0	0	0	0	0	0	0	0
n11	1	0	0	0	1	0	0	0	0
n13	0	0	0	0	0	0	1	0	0

American bullfrog. First detection 14 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	.	.	.	0	0	0	1	1	1
m10	.	.	.	1	0	1	1	0	0
m12	.	.	.	1	0	1	1	0	0
m18	0	0	0	0	0	0	1	0	0
m21	1	0	1	0	0	0	1	0	1
m22	0	0	0	0	0	1	1	0	0
m26	0	0	1	0	0	0	0	0	0
m28	1	0	1	0	0	0	0	1	1
m32	0	0	0	0	0	1	1	0	0
m34	0	0	0	0	0	2	1	0	1
m35	0	0	1	0	0	1	0	0	0
m38	0	0	0	0	2	0	0	0	0
m43	1	0	1	1	0	1	1	1	1
m44	0	0	1	1	0	1	1	1	2
m46	1	2	1	1	0	1	1	0	0
m47	0	0	0	0	0	0	0	0	0
m49	0	0	0	0	0	0	1	0	1
m50	1	1	1	1	0	2	1	1	1
n2	0	0	1	2	0	0	1	0	0
n3	1	0	1	0	0	0	1	1	1
n5	0	0	0	1	0	1	1	1	1
n6	1	0	1	1	0	1	1	1	1
n7	1	1	1	1	1	2	1	1	1
n8	0	0	0	0	0	0	1	0	0
n10	0	0	1	2	0	2	1	1	2
n11	1	1	1	1	2	2	1	1	2
n13	0	0	0	0	0	0	0	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	0	0	0	1	0	1	1	0	1
n17	0	0	0	0	0	0	0	0	0
n18	0	0	0	0	0	0	1	0	1
n20	1	1	2	1	0	2	1	1	2
n22	0	0	1	0	0	0	1	0	2
n24	0	0	0	0	0	1	0	0	1
n26	0	0	0	0	0	0	0	0	0
n29	0	0	1	0	0	1	1	0	1
n31	.	.	.	0	0	0	1	1	2
n33	0	0	0	0	0	0	1	0	1
w1	.	.	.	1	1	2	1	1	1
w3	0	0	0	1	0	1	1	1	1
w4	1	0	1	1	0	0	1	0	1
w9	0	0	0	1	0	0	1	0	1
w10	0	0	0	0	0	0	1	0	0
w12	0	0	0	0	0	1	1	1	2
w13	0	0	0	0	0	0	0	0	0
w14	0	0	0	0	0	0	0	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	0	0	0	0	0	0
w20	0	0	0	1	0	0	0	0	1
w21	0	0	0	0	0	0	0	0	0
w22	0	0	0	0	0	0	1	1	1

Southern leopard frog. First detection 12 March 2007 and last detection 9 September 2007; season exclusive of first date and inclusive of second date.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	0	0	0	2	0	2	0
m10	0	0	0	2	0	2	0	0	0
m12	0	0	0	0	0	1	0	0	0
m18	0	0	0	2	0	2	0	0	0
m21	0	0	0	0	0	0	0	0	1
m22	0	0	0	1	0	0	0	0	0
m26	1	0	0	2	0	2	0	0	0
m28	0	0	0	1	0	0	0	0	0
m32	0	0	0	0	0	0	0	0	0
m34	0	0	1	1	2	0	0	0	0
m35	0	0	0	0	0	0	0	0	1
m38	0	0	0	0	2	0	0	0	0
m43	0	0	0	0	0	2	0	1	0
m44	0	0	0	0	2	2	0	0	0
m46	0	0	0	1	0	0	1	0	0
m47	0	0	0	2	2	2	0	0	0
m49	0	0	1	1	0	0	0	0	0
m50	0	0	0	0	0	0	0	0	0
n2	0	0	0	0	0	2	2	0	0
n3	0	0	0	0	0	2	0	0	0
n5	0	0	0	0	0	2	0	0	1
n6	0	0	1	0	0	2	1	0	0
n7	0	0	0	0	0	0	1	0	0
n8	0	0	1	2	2	2	1	0	0
n10	0	0	0	0	2	2	0	0	0
n11	1	1	0	0	0	0	0	0	0
n13	0	0	0	0	0	0	0	0	1

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n16	0	0	0	0	2	1	0	0	1
n17	0	0	0	2	0	2	0	0	1
n18	0	0	0	0	0	0	1	0	0
n20	0	0	0	0	0	0	1	0	0
n22	0	0	0	0	0	0	1	0	0
n24	0	0	0	0	0	0	1	0	0
n26	0	0	0	0	0	0	0	0	0
n29	0	0	0	2	0	2	0	0	0
n31	.	.	.	0	0	2	1	0	1
n33	0	0	0	2	0	2	0	0	0
w1	.	.	.	0	0	0	1	0	1
w3	0	0	0	1	0	2	1	0	0
w4	0	0	0	0	0	0	0	0	0
w9	0	0	0	0	0	0	0	0	0
w10	1	0	1	2	2	2	0	0	0
w12	0	0	0	0	2	2	0	0	0
w13	0	0	0	1	0	0	1	0	0
w14	0	0	0	0	2	0	0	0	1
w18	.	.	.	0	2	0	0	0	0
w19	0	0	0	0	0	2	0	0	0
w20	0	0	0	0	0	2	1	0	0
w21	0	0	0	1	0	0	0	0	0
w22	1	0	1	0	0	0	0	0	0

Leopard frog complex. First detection 7 March 2007 and last detection 9 September 2007; season inclusive of these dates.

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
m6	0	0	1	1	2	1	1	0	1
m10	0	0	0	2	2	2	2	0	0
m12	0	0	0	2	0	0	1	0	0
m18	0	1	1	2	1	2	1	0	0
m21	0	0	1	0	1	1	1	0	0
m22	0	0	0	1	0	1	1	0	0
m26	1	0	0	2	1	1	1	0	0
m28	0	0	0	1	1	2	1	0	0
m32	0	0	0	1	1	2	1	0	0
m34	0	0	0	2	1	1	1	0	0
m35	1	0	0	0	1	1	1	0	0
m38	0	1	0	1	1	1	1	0	0
m43	0	1	1	1	0	1	1	0	0
m44	1	1	1	0	1	2	1	0	0
m46	1	1	1	1	1	1	1	0	0
m47	0	0	1	2	1	1	0	0	0
m49	1	0	0	1	0	1	1	0	0
m50	1	1	1	2	1	1	0	0	0
n2	0	1	1	0	1	1	1	0	0
n3	0	0	0	1	1	2	1	0	0
n5	0	1	1	1	1	1	1	1	1
n6	1	1	1	1	0	0	1	0	0
n7	1	0	1	0	1	1	1	0	1
n8	0	0	0	2	2	2	1	0	0
n10	0	0	1	0	1	2	1	0	0
n11	1	1	1	1	1	1	0	0	1
n13	0	0	1	0	1	1	1	0	0
n16	0	1	1	1	1	1	1	0	0

Site	Primary survey period 1			Primary survey period 2			Primary survey period 3		
	VES	Dip-net	Trap	VES	Dip-net	Trap	VES	Dip-net	Trap
n17	0	0	1	2	1	2	1	0	0
n18	0	0	0	2	1	1	1	0	0
n20	0	1	1	2	1	1	1	0	0
n22	1	0	1	1	1	2	1	1	1
n24	0	1	1	0	1	1	1	0	0
n26	0	0	1	1	1	2	0	0	0
n29	0	0	0	1	1	1	0	0	1
n31	0	0	1	0	1	1	1	0	0
n33	0	0	1	2	1	2	1	0	0
w1	0	0	0	1	1	1	2	0	0
w3	0	0	0	1	0	1	2	1	1
w4	0	0	1	0	0	1	0	0	0
w9	1	0	0	0	1	1	1	0	0
w10	1	0	0	2	1	1	2	0	0
w12	0	0	1	0	1	2	1	0	0
w13	0	0	0	1	0	0	0	0	0
w14	0	0	0	2	1	0	1	0	0
w18	.	.	.	0	0	0	0	0	0
w19	0	0	0	1	0	1	1	0	0
w20	0	1	1	1	0	1	1	0	0
w21	0	0	0	0	0	0	0	0	0
w22	1	1	1	2	0	1	1	0	0

APPENDIX D. X-matrix tables with truncated data sets for each member of the regional species pool for occupancy and species richness study conducted in summer 2007 in the Lower Grand River basin, north-central Missouri. A table is included for each sampled Wetlands Reserve Program site included in the study. The site designation is indicated above the left corner of each table. Dots in a row (.) indicate missing data due to the truncated nature of the data sets by species. A one (1) in a column indicates species detection and a zero (0) indicates nondetection. Dots in a column indicate missing data as the quadrat represented in the appropriate column was not surveyed in the primary sampling period. Data from all three primary sampling periods was used when computing relative species richness estimates using the occupancy method and only data from primary sampling period two was used when computing species richness estimates using the capture-recapture method.

m6

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	1	0	1	0	0	1	1	1
GTF	0	0	0	1	1	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	1	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m10

Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
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m12

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	1	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m18

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3												
	quadrats											quadrats											quadrats												
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11		
SMS	0	0	0	1	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	1	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	1	1	0	0	0	0	1	0	1
SPP	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	1	1	0	0	0	1	0	.	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	1	0	1	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m21

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m22

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	1	0	0	1	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
BUL	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	.
SLF	0	0	0	0	0	0	0	0	0	.	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
CEN	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
ETS	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
GPT	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
WHT	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
NMT	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
CRA	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
GRF	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.
WFR	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	.

m26

Spp.	Primary sampling period 1									Primary sampling period 2									Primary sampling period 3								
	quadrats									quadrats									quadrats								
	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9
SMS	0	0	0	0	0	0	0	0	0	1	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0
ACR	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

200

m28

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	1
AMT	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
GTF	1	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
PLF	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
BUL	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m32

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3											
	quadrats											quadrats											quadrats											
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	
SMS	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1		1	0	0	0	0	0	0	1	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	1	0	1	0	1	0	0	0	0	0	0	1	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

202

m34

Spp.	Primary sampling period 1												Primary sampling period 2											
	quadrats												quadrats											
	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12
SMS	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	.	
AMT	0	0	0	0	0	0	0	0	0	0	0	.
ACR	0	0	1	0	.	1	0	0	0	0	.	0	1	0	1	0	0	1	1	1	0	0	0	.
GTF	0	0	1	0	1	0	0	1	0	0	1	.
SPP	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
PSE	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
PLF	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
BUL	0	0	0	0	.	0	0	0	0	0	.	0	0	1	0	0	0	0	0	0	0	0	0	.
SLF	0	0	0	0	.	0	0	0	0	1	.	0	0	0	1	0	0	0	0	1	0	0	0	.
CEN	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
ETS	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
GPT	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
WHT	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
NMT	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
CRA	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
GRF	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.
WFR	0	0	0	0	.	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	.

m34 continued

Spp.	Primary sampling period 3										
	quadrats										
	1	2	3	4	5	6	7	8	9	10	11
sms
amt
acr		1	0	0	0	0	0	0	1	0	0
gtf	0	0	0	0	0	0	0	0	0	0	0
spp
chf
plf	0	0	0	0	0	0	0	0	0	0	0
bul	1	0	0	0	0	0	0	1	0	0	0
slf	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0

m35

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3											
	quadrats											quadrats											quadrats											
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	1	0	0	
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	
PLF	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
BUL	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
CEN	0	0	0	0	0	0	0	1	1	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	

205

m38

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	.	1	0	1	0	0	0	1	0	0	0	1
AMT	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	1	0	0	0	0	0	0	0	.	0	1	1	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	.	0	1	1	0	0	0	1	0	0	0	1
SPP	0	0	0	0	0	0	0	0	0	0	.	0	0	1	0	0	0	0	0	0	0	0
PSE	0	0	1	0	0	0	1	0	0	0	.	0	0	1	1	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	.	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	.	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m43

Spp.	Primary sampling period 1													
	quadrats													
	1	2	3	4	5	6	7	8	9	10	11	12	13	14
SMS	1	1	1	0	0	0	0	1	.	0	.	1	0	0
AMT	0	0	0	0	0	0	0	0	.	0	.	0	0	0
ACR	0	0	0	0	0	0	1	0	.	0	.	0	0	0
GTF	0	0	0	0	0	0	0	0	.	0	.	0	0	0
SPP	0	0	0	0	0	0	0	0	.	0	.	0	0	0
PSE	0	1	0	0	0	0	0	1	.	0	.	1	0	0
PLF	0	0	0	0	0	0	0	0	.	0	.	0	0	0
BUL	0	1	0	0	0	0	0	1	.	0	.	0	0	0
SLF	0	0	0	0	0	0	0	0	.	0	.	0	0	0
CEN	0	0	0	0	0	0	0	0	.	0	.	0	0	0
ETS	0	0	0	0	0	0	0	0	.	0	.	0	0	0
GPT	0	0	0	0	0	0	0	0	.	0	.	0	0	0
WHT	0	0	0	0	0	0	0	0	.	0	.	0	0	0
NMT	0	0	0	0	0	0	0	0	.	0	.	0	0	0
CRA	0	0	0	0	0	0	0	0	.	0	.	0	0	0
GRF	0	0	0	0	0	0	0	0	.	0	.	0	0	0
WFR	0	0	0	0	0	0	0	0	.	0	.	0	0	0

m43 continued

Spp.	Primary sampling period 2													
	quadrats													
	1	2	3	4	5	6	7	8	9	10	11	12	13	14
SMS
AMT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
ACR	0	1	0	1	0	1	1	1	0	1	1	0	.	.
GTF	0	0	0	0	1	0	0	1	0	0	0	0	.	.
SPP	0	0	0	0	0	0	0	0	0	0	0	0	.	.
PSE	0	0	0	0	0	0	0	0	0	0	0	0	.	.
PLF	0	0	0	0	0	0	0	0	0	0	0	0	.	.
BUL	0	1	0	0	0	1	0	0	0	1	1	0	.	.
SLF	0	0	0	0	0	0	0	1	0	0	0	0	.	.
CEN	0	0	0	0	0	0	0	0	0	0	0	0	.	.
ETS	0	0	0	0	0	0	0	0	0	0	0	0	.	.
GPT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
WHT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
NMT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
CRA	0	0	0	0	0	0	0	0	0	0	0	0	.	.
GRF	0	0	0	0	0	0	0	0	0	0	0	0	.	.
WFR	0	0	0	0	0	0	0	0	0	0	0	0	.	.

m43 continued

Spp.	Primary sampling period 3													
	quadrats													
	1	2	3	4	5	6	7	8	9	10	11	12	13	14
SMS
AMT
ACR	0	0	1	0	0	0	0	1	0	0	1	0	.	.
GTF
SPP
PSE
PLF	0	0	0	0	0	0	0	0	0	0	0	0	.	.
BUL	0	0	1	0	0	0	1	0	0	0	1	0	.	.
SLF	0	0	0	0	0	0	0	0	0	0	0	0	.	.
CEN	0	0	0	0	0	0	0	0	0	0	0	0	.	.
ETS	0	0	0	0	0	0	0	0	0	0	0	0	.	.
GPT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
WHT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
NMT	0	0	0	0	0	0	0	0	0	0	0	0	.	.
CRA	0	0	0	0	0	0	0	0	0	0	0	0	.	.
GRF	0	0	0	0	0	0	0	0	0	0	0	0	.	.
WFR	0	0	0	0	0	0	0	0	0	0	0	0	.	.

m44

Spp.	Primary sampling period 1														
	quadrats														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
SMS	0	0	0	0	0
AMT
ACR	0	1	1	1	0
GTF	0	0	0	0	0
SPP	0	0	0	0	0
PSE	0	1	1	1	0
PLF	0	0	0	0	0
BUL	0	1	1	1	0
SLF	0	0	0	0	0
CEN	0	0	0	0	0
ETS	0	0	0	0	0
GPT	0	0	0	0	0
WHT	0	0	0	0	0
NMT	0	0	0	0	0
CRA	0	0	0	0	0
GRF	0	0	0	0	0
WFR	0	0	0	0	0

m44 continued

Spp.	Primary sampling period 2														
	quadrats														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
SMS	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
AMT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
ACR	.	.	1	1	1	0	0	0	0	0	1	0	.	.	.
GTF	.	.	1	1	1	1	0	0	0	0	1	1	.	.	.
SPP	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
PSE	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
PLF	.	.	0	1	1	0	0	0	0	0	0	0	.	.	.
BUL	.	.	1	1	0	0	0	0	0	0	0	0	.	.	.
SLF	.	.	1	1	1	0	0	0	0	0	0	0	.	.	.
CEN	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
ETS	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
GPT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
WHT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
NMT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
CRA	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
GRF	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
WFR	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.

m44 continued

Spp.	Primary sampling period 3														
	quadrats														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
SMS
AMT
ACR	.	.	1	1	1	0	0	0	0	0	0	0	.	.	.
GTF
SPP
PSE
PLF	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
BUL	.	.	1	1	1	0	1	0	0	0	1	1	.	.	.
SLF	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
CEN	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
ETS	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
GPT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
WHT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
NMT	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
CRA	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
GRF	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.
WFR	.	.	0	0	0	0	0	0	0	0	0	0	.	.	.

m46

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	1	0	0	0	0	0	0	0	1	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	1	1	0	0	1	0	0	0	0	1	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	1	1	1	1	0	0	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	1	0	1	1	1	0	0	0	0	1	1	0	0	1	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	1	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m47

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	1
ACR	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0	0	0	1
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	1	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m49

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3										
	quadrats										quadrats										quadrats										
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
AMT	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
ACR	0	1	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
PLF	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
SLF	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

m50

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	0	0	1	0	1	0	1	0	1	0	0	0	1	1	1	1	0	0	0	0	0	0	1	1	1	1	0	0
GTF	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	1	1	1	1	1	1	1	1	1	0	1	0	0	1	1	1	1	1	0	0	0	0	0	0	1	1	1	1	1	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n2

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	1	1	0	0	1	0	0	1	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0
BUL	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	1	1	0	0	0	0	0	1	0
SLF	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n3

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	0	1	1	0	0
AMT	0	1	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0
GTF	0	1	1	1	1	1	1	1	1	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	0	1	1	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0
BUL	0	0	1	0	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1	1	1	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n5

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	1	0	0	1	1	1	0	0	1	0	1	1	1
GTF	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	1	1	0	0	1	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	1	1	0	0	1	0	0	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n6

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	1	0	1	0	0	0	0	1	1	0	1	0	1	0	0	0	0	1	0	0	1	1	1	0	1	0	0	0	1	1	1
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
BUL	0	0	1	0	0	0	1	0	1	0	0	0	1	1	1	0	0	0	1	1	1	1	0	0	1	1	0	0	0	0	1	1	1
SLF	0	0	0	0	0	0	1	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n7

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	1	0	0	1	1	1	1	1	0	1	1	0	1
GTF	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	1	0	1	0	1	1	1	0	0	1	1	1	1	0	0	1	0	1	0	1	1	0	1	1	0	1	1	0	0	1	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n8

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	1	0	0	1	1	1	0	1	1	0	1	1	1	1	1	0	0	1	1	1	0
AMT	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	1	0	0	0	0
GTF	1	1	0	1	1	1	0	1	1	1	1
SPP	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0
SLF	0	0	0	1	1	0	1	0	0	0	0	1	1	0	1	0	1	1	1	1	1	0	0	0	0	0	0	0	1	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n10 continued

Spp.	Primary survey occasion 3														
	quadrats														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
SMS
AMT
ACR	0	1	0	0	0	1	0	1	1	1	0	1	0	1	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP
PSE
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	1	1	1	0	0	1	0	0	0	0	0	1	1	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n11

Spp.	Primary sampling period 1																	
	quadrats																	
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	1	0	1	0	1	0	1	0	1	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	1	0	0	0	1	0	0	0	0	1	0	1	0
SLF	0	0	1	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n11 continued

Spp.	Primary sampling period 2																	
	quadrats																	
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
SMS	0	0	1	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0
ACR	1	1	1	0	1	0	0	1	0	1	0
GTF	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	1	0	0	0
BUL	1	1	0	0	1	1	1	1	1	1	1
SLF	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0

n11 continued

Spp.	Primary sampling period 3																	
	quadrats																	
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
SMS
AMT
ACR	1	0	1	1	0	0	0	1	1	0	0	
GTF	0	0	0	0	0	0	0	0	0	0	0	
SPP
PSE	0	0	0	0	0	0	0	0	0	0	0	
PLF	0	0	0	0	0	0	0	0	0	0	0	
BUL	0	0	0	1	1	1	0	0	0	0	1	
SLF	0	0	0	0	0	0	0	0	0	0	0	
CEN	0	0	0	0	0	0	0	0	0	0	0	
ETS	0	0	0	0	0	0	0	0	0	0	0	
GPT	0	0	0	0	0	0	0	0	0	0	0	
WHT	0	0	0	0	0	0	0	0	0	0	0	
NMT	0	0	0	0	0	0	0	0	0	0	0	
CRA	0	0	0	0	0	0	0	0	0	0	0	
GRF	0	0	0	0	0	0	0	0	0	0	0	
WFR	0	0	0	0	0	0	0	0	0	0	0	

n13

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3											
	quadrats											quadrats											quadrats											
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	
SMS	0	0	0	0	0	0	0	0	0	0	.	0	0	0	1	0	0	1	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	1	0	0	0	1	0	.	
GTF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	.	0	0	0	1	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	1	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	1	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	1	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0

n16

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	0	0	0	1	0	0	0	0	0	1	0	0	1	0	1	0	1	0	0	0	1	1	0	1	0	1	1	0	0	1	0
GTF	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	1	0
SLF	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n17

Spp.	Primary survey occasion 1												Primary survey occasion 2											
	quadrats												quadrats											
	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12
SMS	0	0	0	0	1	0	0	.	.	0	0	1	1	0	0	0	0	0	0	1	1	0	0	1
AMT	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GTF	0	0	1	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	1	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	.	.	0	0	0	1	0	0	1	0	0	0	0	1	0	0	1
CEN	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n17 continued

Spp.	Primary survey occasion 3											
	quadrats											
	1	2	3	4	5	6	7	8	9	10	11	12
SMS
AMT	0	0	0	0	0	0	0	0	0	0	0	.
ACR	0	0	1	1	0	1	0	0	0	0	0	.
GTF	0	0	0	0	0	0	0	0	0	0	0	.
SPP
PSE	0	0	0	0	0	0	0	0	0	0	0	.
PLF	0	0	0	1	0	0	0	0	0	0	0	.
BUL	0	0	0	0	0	0	0	0	0	0	0	.
SLF	0	0	0	1	0	0	0	0	0	0	0	.
CEN	0	0	0	0	0	0	0	0	0	0	0	.
ETS	0	0	0	0	0	0	0	0	0	0	0	.
GPT	0	0	0	0	0	0	0	0	0	0	0	.
WHT	0	0	0	0	0	0	0	0	0	0	0	.
NMT	0	0	0	0	0	0	0	0	0	0	0	.
CRA	0	0	0	0	0	0	0	0	0	0	0	.
GRF	0	0	0	0	0	0	0	0	0	0	0	.
WFR	0	0	0	0	0	0	0	0	0	0	0	.

n18

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3												
	quadrats											quadrats											quadrats												
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11		
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
ACR	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	1	0	1	1	1	1	0	0	0	0	0	0	1	0
GTF	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n20

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
ACR	1	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0	0	1	0	0	1	0	0	1	0	1	0	1	1	0	0
GTF	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
BUL	1	0	0	0	0	1	0	0	1	0	0	0	0	1	1	0	0	0	1	0	0	0	1	0	1	1	1	0	0	1	1	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n26

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3											
	quadrats											quadrats											quadrats											
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	
SMS	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	0	1	1	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	1	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n29

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
ACR	0	1	0	0	0	0	1	0	0	0	1	0	1	0	0	0	0	1	0	0	0	0	0	1	0	0	1	0	1	0	0	0	1
GTF	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n31-dropped Primary sampling period 1 as it was truncated from all detected species.

Spp.	Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	0	0	0	0	0	0	0	0	1	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	1	0	1
GTF	0	0	1	0	1	1	1	1	1	1	1
SPP	0	0	0	0	0	0	0	0	0	0	0
PSE	1	0	0	1	1	0	1	0	1	1	1
PLF	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	1	1	1	1	0	1
SLF	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0	1	1	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

n33

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3										
	quadrats											quadrats											quadrats										
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11
SMS	1	0	0	0	0	0	0	0	0	0	.	0	1	0	0	1	1	0	0	0	0	0
AMT	0	1	0	0	0	0	0	1	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	.	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
GTF	0	0	0	0	0	0	0	0	0	1	0
SPP	1	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	.	0	1	0	1	0	1	0	0	0	0	0
PLF	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	.	1	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

w1-dropped primary sampling period 1 as it was truncated from all detected species

Spp.	Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	1	1	1	0	0	0	0	1	1	1
AMT	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	1	0	0	0	0	1	1	0	0	1	0	0	0	1	1
GTF	0	0	0	0	0	0	0	1	0	0
SPP	0	0	0	0	0	0	0	1	0	0
PSE	0	1	0	0	1	0	0	1	0	1
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	1	0	1	1	0	1	1	0	0	1	1	0	1	1	0
SLF	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

w4

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	1	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

w12

Spp.	Primary sampling period 1									Primary sampling period 2									Primary sampling period 3								
	quadrats									quadrats									quadrats								
	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9
SMS	0	0	0	0	1	0	1	0	1	1	0	0	0	1	0	0	1	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	1	0	0	0	1	1	1	1	1	1	0
GTF	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	1	0	0	0	0	0	0	1	0	0	0
BUL	0	0	0	0	0	0	0	0	0	1	0	1	0	1	1	0	1	0	0	0	0	1	0	1	0	0	0
SLF	0	0	0	0	0	0	0	0	0	1	1	0	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

w13

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	1	0	0	0	0	0	1	1	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

w14

Spp.	Primary sampling period 1											Primary sampling period 2											Primary sampling period 3											
	quadrats											quadrats											quadrats											
	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	1	2	3	4	5	6	7	8	9	10	11	
SMS	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	
AMT	0	0	0	0	1	0	1	0	0	1	0	1	
ACR	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	1	0
GTF	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
SPP	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	
PSE	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	
PLF	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	1	0	1	0	0	0	
BUL	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
SLF	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	1	0	1	0	0	0	
CEN	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
ETS	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
GPT	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
WHT	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
NMT	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
CRA	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
GRF	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0
WFR	.	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	0	0	0	0	0	0

w18-site w18 was not surveyed in primary survey occasion 1.

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	0	0	0	0	0	0	0	0	0	0	
AMT	0	0	0	0	1	0	0	0	0	0	
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
SPP	0	0	0	0	0	0	0	0	0	0	
PSE	0	0	0	0	0	0	0	0	0	0	
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
SLF	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	

w20

Spp.	Primary sampling period 1										Primary sampling period 2										Primary sampling period 3									
	quadrats										quadrats										quadrats									
	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10	1	2	3	4	5	6	7	8	9	10
SMS	1	0	1	1	1	1	1	.	.	1	0	0	0	0	0	0	0	0	0	0	
AMT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	1	0	0	
ACR	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GTF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
SPP	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	
PSE	0	0	0	1	0	0	0	.	.	1	0	0	0	0	0	0	0	0	0	0	
PLF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
BUL	0	0	0	0	0	0	0	.	.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
SLF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	
CEN	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
ETS	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GPT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WHT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
NMT	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
CRA	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GRF	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
WFR	0	0	0	0	0	0	0	.	.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	

w21

Spp.	Primary sampling period 1									Primary sampling period 2									Primary sampling period 3								
	quadrats									quadrats									quadrats								
	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9	1	2	3	4	5	6	7	8	9
SMS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
AMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ACR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GTF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SPP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PSE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PLF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SLF	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ETS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GPT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WHT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
NMT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GRF	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E. Evidentiary information supporting the Site Occupancy Data Analysis
Small-mouthed salamander

The most general model in the candidate set, Model 2 $\psi(W M N), p(\cdot)$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.005, $\hat{c} = 1.9$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The small-mouthed salamander was detected at 29 of 50 sites for a naïve occupancy estimate of 0.58. The most supported model for the small-mouthed salamander was Model 1 $\psi(\cdot), p(\cdot)$ (Table 6) with an occupancy probability estimate of 0.66 (se=0.08). Model 3 $\psi(W M N), p(\cdot)$ and Model 5 $\psi(M W N), p(\cdot)$ are within 1.74 and 1.94 $QAIC_c$ delta units, respectively, of the most supported model and, thus, also receive strong support. Real parameter estimates for Model 3 included an occupancy probability estimate for walk-away sites of 0.52 (se=0.16) and an occupancy probability estimate for maximize hydrology-naturalistic sites of 0.70 (se=0.09). The odds ratio of occupancy between a walk-away and maximize hydrology-naturalistic site is 2.1 (CI= 0.5, 9.9). Real parameter estimates for Model 5 included an occupancy probability estimate for maximize hydrology sites of 0.74 (se=0.13) and an occupancy probability estimate for walk-away-naturalistic sites of 0.61 (se=0.10). The odds ratio between a maximize hydrology and walk-away-naturalistic site is 0.6 (CI= 0.1, 2.5). The evidence ratio based on $QAIC_c$ weights between the most supported and least supported models (w_1/w_5) in the candidate set is 7.5. The top four models contain 94% of the model weights.

American toad

The most general model in the candidate set, $\psi(W M N), p(\cdot)$, had a large positive untransformed beta estimate and an extremely high standard error associated with the occupancy estimate for N; when using the logit link, this indicates the parameter is near the boundary of 1 and the extremely high standard error associated with the untransformed beta estimate results in an inestimable standard error for the real parameter (Long et al. 2007, MacKenzie 2009). A goodness-of-fit test performed on this model showed no overdispersion (P -value = 0.80, $\hat{c} = 0.13$); however, I also ran a GOF test on Model 5 $\psi(M WN), p(\cdot)$, the next most general model that provided reasonable estimates, with similar results (P -value = 0.78, $\hat{c} = 0.13$) so AIC_c values and standard errors were not adjusted. The low \hat{c} value indicates there was less variation in the data than would be reasonably expected and likely points toward sparse data with a high percent of missing values due to the truncated nature of the data set. American toads were detected at 18 of 50 sites for a naïve occupancy estimate of 0.36. The most supported model for the American toad was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6) with an occupancy probability estimate of 0.85 ($se=0.32$). Model 4 $\psi(N WM), p(\cdot)$ and Model 5 $\Psi(M WN), p(\cdot)$ are within 0.98 and 1.04 AIC_c delta units, respectively, of the most supported model and, thus, also receive strong support. Model 4 contains a high positive untransformed beta for ψ_N and a large negative untransformed beta for ψ_{WM} indicating that the estimate for ψ_N is near the boundary of 1 and the estimate for ψ_{WM} is near the boundary of 0. Real parameter estimates for Model 4 included an occupancy probability estimate for naturalistic sites of 1 ($se=$ inestimable) and an occupancy probability estimate for walk-away-maximize hydrology sites of 0.67 ($se=0.51$). The fact that the estimate for ψ_{WM} is not closer to 0 may indicate the model is over-parameterized for the data set (Donovan and Hines 2007, www.phidot.org). The large beta values associated with

parameters in this model make odds ratio values unusable as it would take a very large difference in the beta to effect the real parameter; odds ratio are a measure based on a 1 unit increase in the covariate. Real parameter estimates for Model 5 $\psi(\text{M WN}), p(\cdot)$ included an occupancy probability estimate for maximize hydrology sites of 0.58 (se=0.32) and an occupancy probability estimate for walk-away-naturalistic sites of 0.94 (se=0.35). The large standard error (se=5.6) associated with the untransformed beta estimate for ψ_{WN} ($\beta_{\text{WN}}=2.43$) rendered the odds ratio between maximize hydrology sites and walk-away-naturalistic sites rather high (OR=11.2) with an unreasonable confidence interval. The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 4.5. The top four models contain 92% of the model weights.

Northern cricket frog

Results of the goodness- of- fit test performed on Model 2 $\psi(\text{W M N}), p(\cdot)$ indicated overdispersion ($P\text{-value} = 0.0001, \hat{c} = 2.2$); therefore, AIC values were converted to QAIC_c , standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). A GOF test was also performed on Model 1 $\psi(\cdot), p(\cdot)$ due to the large positive untransformed betas associated with both ψ_{M} and ψ_{N} in Model 2 and produced similar results ($P\text{-value} = 0.001, \hat{c} = 2.2$). Northern cricket frogs were detected at 46 of 50 sites resulting in a naïve occupancy estimate of 0.92. The most supported model for northern cricket frog was Model 3 $\psi(\text{W MN}), p(\cdot)(\cdot)$ (Table 6) with an estimate of occupancy on walk-away sites of 0.70 (se=0.13) and an estimate of of occupancy on maximize hydrology-naturalistic sites of 1 (se= inestimable). This model has a large positive untransformed beta estimate for ψ_{MN} indicating the real parameter is near the boundary of 1; this is not unreasonable as cricket frogs were

detected on all 18 maximize hydrology sites and all 19 naturalistic sites. One recommended approach to address this situation in program PRESENCE is to delete the beta parameter associated with psi from the design matrix and fix the parameter to 1. This allows the estimation of detection to proceed. Using this method, in essence, reduces the model to the constant $\psi(\cdot)$ model. The next most supported model, $\psi(W M N)$, also had large positive untransformed betas associated with both ψ_M and ψ_N . Again, the recommended solution is to fix the parameters to 1 which reduces the model to $\psi(\cdot)$. The constant model, $\psi(\cdot)$, was the next most supported model in the candidate set and provided a reasonable occupancy probability estimate of 0.93 (se=0.04). The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 5.5. The top four models contain 91% of the model weights.

Grey treefrog complex

A goodness-of-fit test was performed on Model 2 $\psi(W M N)$, $p(\cdot)$ and indicated overdispersion (P -value = 0.16, $\hat{c} = 1.4$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Grey treefrogs were detected on 33 of 50 sites for a naïve occupancy estimate of 0.66. The most supported model for the grey treefrog was Model 1 $\psi(\cdot)$, $p(\cdot)(\cdot)$ (Table 6) with an occupancy probability estimate of 0.78 (se=0.08). Model 3 $\psi(W MN)$, $p(\cdot)$ and Model 4 $\psi(N WM)$, $p(\cdot)$ are within 0.87 and 1.70 delta units, respectively, of the most supported model and, thus, also receive strong support. Real parameter estimates for Model 3 included an occupancy probability estimate for walk-away sites of 0.61 (se=0.16) and an occupancy probability estimate for maximize hydrology-naturalistic sites of 0.85 (se=0.10). The odds ratio between walk-away

sites and maximize hydrology-naturalistic sites is 3.6 (CI = 0.5, 24.5). Real parameter estimates for Model 4 included an occupancy probability estimate for naturalistic sites of 0.86 (se=0.12) and an occupancy probability estimate for walk-away-maximize hydrology sites of 0.72 (se=0.11). The odds ratio between naturalistic sites and walk-away-maximize hydrology sites is 0.43 (CI = 0.05, 3.8). The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 5.0. The top four models contain 92% of the model weights.

Spring peeper

The most general model in the candidate set, Model 2, $\psi(W M N), p(\cdot)$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.21, $\hat{c} = 1.3$); therefore, AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The northern spring peeper was detected on nine out of 50 sites for a naïve occupancy estimate of 0.18. The most supported model for the northern spring peeper was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6) with an occupancy estimate of 0.39 (se=0.22). Model 4 $\psi(N WM), p(\cdot)$ and Model 5 $\psi(M WN), p(\cdot)$ were within 1.07 and 1.41 delta units, respectively, of the most supported model and, thus, also receive strong support. Real parameter estimates for Model 4 included an occupancy probability estimate for naturalistic sites of 0.58 (se=0.36) and an occupancy probability estimate for walk-away-maximize hydrology sites of 0.28 (se=0.18). The odds ratio between naturalistic sites and walk-away-maximize hydrology sites is 0.28 (CI = 0.02, 3.3). Real parameter estimates for Model 5 included an occupancy probability estimate for maximize hydrology sites of 0.23 (se=0.19) and an occupancy probability estimate for walk-away-naturalistic sites of 0.48 (se=0.28). The odds

ratio between maximize hydrology sites and walk-away-naturalistic sites is 3.1 (CI = 0.3, 30). The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 5.3. The top four models contain 93% of the model weights.

Pseudacris spp.

The most general model in the candidate set, Model 2, $\psi(W M N), p(\cdot)$, was used to perform the goodness-of-fit test. No overdispersion was indicated so AIC_c values and standard errors were not adjusted (P -value = 0.31, $\hat{c} = 0.96$). *Pseudacris spp.* were detected on 37 out of 50 sites for a naïve occupancy estimate of 0.74. The most supported model for *Pseudacris spp.* was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6) with an occupancy probability estimate of 0.92 (se=0.09). Model 4 $\psi(N WM), p(\cdot)$ is within 1.79 delta units of the most supported model and, thus, also receives strong support. Real parameter estimates for Model 4 include an occupancy probability estimate for naturalistic sites of 0.97 (se=0.10) and an occupancy probability estimate for walk-away-maximize hydrology sites of 0.87 (se=0.12). The odds ratio between naturalistic sites and walk-away-maximize hydrology sites is 0.22 (CI=0.0003, 164.00). The relatively high standard error (se=3.4) combined with the untransformed beta estimate for psiMN ($\beta_{MN}=1.52$) resulted in the large value on the upper end of the confidence interval. The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 8.4. The top four models contain 89% of the model weights.

Plains leopard frog

The goodness-of-fit test was performed on Model 2, $\psi(W M N), p(\cdot)$, the most general model in the candidate set with no indication of overdispersion (P -value = 0.48, $\hat{c} = 0.76$). This model included a large positive untransformed beta estimate for psiN with an extremely high

standard error so a GOF test was also run on Model 3, $\psi(M WN), p(\cdot)$, the next most general model that produced reasonable estimates. Again, no overdispersion was indicated (P -value = 0.38, $\hat{c} = 0.85$) so AIC_c values and standard errors were not adjusted. Plains leopard frogs were detected on 35 of 50 sites for a naïve occupancy estimate of 0.70. The most supported model for the plains leopard frog was Model 2 $\psi(W M N), p(\cdot)$ (Table 6); the large positive untransformed beta estimate for ψ_iN indicates the real parameter estimate is near the boundary 1 with an inestimable standard error. This is not an unreasonable estimate given that 18 of the 19 naturalistic sites were occupied by plains leopard frogs. The occupancy probability estimates for walk-away sites and maximize hydrology sites were 0.66 (se=0.17) and 0.65 (se=0.14), respectively. Odds cannot be calculated for ψ_iN as it would involve division by zero; the odds ratio between a maximize hydrology site and a walk-away site is 1 (CI = 0.14, 7.4). Little, if any, confidence can be placed on this model give the inestimable standard error associated with ψ_iN . Model 5, $\psi(M WN), p(\cdot)$, within 1.97 delta units, is the next most supported model. Real parameter estimates for Model 5 include an occupancy probability estimate for maximize hydrology sites of 0.68 (se=0.15) and an occupancy probability estimate for walk-away-naturalistic sites of 0.99 (se=0.11). The large standard error associated with the untransformed beta estimate for ψ_iWN (se=13.3) rendered the odds ratio between maximize hydrology sites and walk-away-naturalistic sites rather high (OR=55.6) with an unreasonable confidence interval. The next most supported model in the candidate set, within 3.09 delta units of the most supported model, is Model 1 $\psi(\cdot)$ with an occupancy probability estimate of 0.87 (se=0.10). The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 30.5. The top three models contain 90% of the model weights.

American bullfrog

A goodness-of-fit test was performed on Model 2, $\psi(W M N), p(\cdot)$, and indicated a lack of fit (P -value = 0.0001, $\hat{c} = 2.3$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Bullfrogs were detected on 41 out of 50 sites for a naïve occupancy estimate of 0.82. The most supported model for the bullfrog was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6) with an occupancy probability estimate of 0.83 (se=0.05). Model 3 $\psi(W MN), p(\cdot)$ and Model 5 $\Psi(M WN), p(\cdot)$ are within 0.38 and 0.80 delta units, respectively, of the most supported model and, thus, also receive strong support. Real parameter estimates for Model 3 include an occupancy probability estimate for walk-away sites of 0.62 (se=0.14) and an occupancy probability estimate for maximize hydrology-naturalistic sites of 0.90 (se=0.05). The odds ratio between walk-away sites and maximize hydrology-naturalistic sites is 5.3 (CI = 1.1, 27.1). Real parameter estimates for Model 5 include an occupancy probability estimate for maximize hydrology sites of 0.95 (se=0.05) and an occupancy probability estimate for walk-away-naturalistic sites of 0.76 (se=0.08). The odds ratio between maximize hydrology sites and walk-away-naturalistic sites is 0.17 (CI = 0.02, 1.8). The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 3.1. The top four models contain 90% of the model weights.

Southern leopard frog

A goodness-of-fit test was run on Model 2 $\psi(W M N), p(\cdot)$ model although the untransformed beta estimates for ψ_M and ψ_N were large positives with extremely high associated standard errors. There was not a model in the candidate set that did not exhibit a similar problem. Overdispersion was indicated (P -value = 0.19, $\hat{c} = 1.3$); therefore, AIC_c values

were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Southern leopard frogs were detected at 45 of 50 sites for a naïve occupancy estimate of 0.90. The most supported model for the southern leopard frog was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6); however, the untransformed beta estimate was a large positive indicating the parameter is near the boundary of 1 and the extremely large standard error associated with the beta estimate makes the standard error of the real parameter inestimable (Hines, www.phidot.org). Model 3 $\psi(W MN), p(\cdot)$ also had a large positive untransformed beta estimate and associated standard error while the remaining two models did not provide reasonable estimates of occupancy due to the variance-covariance matrices not computing successfully.

Leopard frog complex

A goodness-of-fit test performed on Model 2 $\psi(W M N), p(\cdot)$ indicated lack of fit (P -value = 0.001, $\hat{c} = 4.2$); however, this model had large positive untransformed betas for ψ_iM and ψ_iN with associated high standard errors so a GOF test was also run on Model 1 with comparable results (P -value = 0.001, $\hat{c} = 4.2$). AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Leopard frogs were detected on 48 out of 50 sites with a naïve occupancy estimate of 0.96. The most supported model for the leopard frog complex was Model 1 $\psi(\cdot), p(\cdot)(\cdot)$ (Table 6) with a occupancy estimate of 0.97 (se=0.03). Model 3 $\psi(W MN), p(\cdot)$ is within 1.04 delta units of the most supported model and, therefore, also receives strong support. Model 3 $\psi(W MN), p(\cdot)$ had a large positive untransformed beta estimate with large standard error for ψ_iMN

indicating the real parameter is close to or equal to 1. Also, as stated earlier, Model 2, $\psi(W M N), p(\cdot)$, within 3.41 delta units of the most supported model, also had large positive untransformed beta estimates with large standard errors for $\psi_i M$ and $\psi_i N$ indicating the real parameters are close to or equal to 1. The remaining 2 models did not provide useable estimates of occupancy probabilities due to failure of the variance-covariance matrix to compute successfully. The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_5) in the candidate set is 5.5. The top four models contain 93% of the model weights.

APPENDIX F. Evidentiary information supporting the Detection Data Analysis
Small-mouthed salamander

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.20, $\hat{c} = 1.1$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for the small-mouthed salamander was Model 4 $\psi(\cdot), p(\text{method})$ (Table 8) with detection probability estimates of 0.05 (se=0.03), 0.43 (se=0.07) and 0.53 (se=0.07) for VES, net and trap, respectively (Table 9). Although not the most supported model in the set, the $p(\text{day sq})$ model still provides useful information with the peak estimate of p occurring from 5/2 – 5/16 at approximately 0.53. The evidence ratio based on $QAIC_c$ weights between the most supported and second most supported models (w_1/w_2) in the candidate set is 40.5. The top model contains 98% of the model weights.

American toad

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated no lack of fit (P -value = 0.70, $\hat{c} = 0.16$) so AIC_c values and standard errors were not adjusted. The most supported model for the American toad was Model 4 $\psi(\cdot), p(\text{method})$ (Table 8) with probability estimates of 0.19 (se=0.08), 0.09 (se=0.05), and 0.04 (se=0.03) for VES, net, and trap, respectively (Table 9). The next most supported model was Model 1 $\psi(\cdot), p(\cdot)$ with a detection probability estimate of 0.10 (se=0.04). Model 3 $\psi(\cdot), p(\text{day sq})$ indicated that detection probability estimates for the American toad peaked from 6/4 – 6/22 at approximately 0.14. The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_4) in the candidate set is 10.5. The top three models contain 93% of the model weights.

Northern cricket frog

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.13, $\hat{c} = 1.3$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for northern cricket frog was Model 4 $\psi(\cdot), p(\text{method})$ (Table 8) with detection probability estimates of 0.69 (se=0.05), 0.26 (se=0.04), and 0.32 (se=0.05) for VES, net and trap, respectively (Table 9). Model 3 $\psi(\cdot), p(\text{day sq})$ also provides useful information as the peak in detection probability estimates for northern cricket frog occurred from 6/26 – 7/31 with p approximately 0.58. The evidence ratio based on $QAIC_c$ weights between the most supported and second most supported model (w_1/w_2) in the candidate set is 450. The top model contains 99% of the model weights.

Grey treefrog complex

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.14, $\hat{c} = 1.4$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for grey treefrogs was Model 3 $\psi(\cdot), p(\text{day square})$ (Table 8) with a peak in detection probability estimates of approximately 0.65 occurring between 6/4 - 6/15 (Figure 14). The top model contains 100% of the model weights.

Spring peeper

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit ($P\text{-value} = 0.16, \hat{c} = 1.4$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for the northern spring peeper was Model 1 $\psi(\cdot), p(\cdot)$ (Table 8) with an estimated detection probability of 0.10 ($se=0.07$). Model 3 $\psi(\cdot), p(\text{day square})$ is within 1.84 delta units of the most supported model and, thus, also receives strong support. The highest probability detection estimates of approximately 0.2 occurred between 4/30 and 5/23. The remaining models, $p(\text{day})$ and $p(\text{method})$, produced detection probability estimates ranging from 0.08 to 0.1 and are virtually identical to the constant p model as evidenced by the similar log-likelihood values. The evidence ratio based on $QAIC_c$ weights between the most supported and least supported models (w_1/w_4) in the candidate set is 9.4. The top three models contain 94% of the model weights.

Pseudacris spp.

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated no lack of fit ($P\text{-value} = 0.33, \hat{c} = 0.83$) so AIC_c values and standard errors were not adjusted. The most supported model for *Pseudacris* sp. was Model 3 $\psi(\cdot), p(\text{day square})$ (Table 8) with probability estimates that peaked at approximately 0.47 during the period 4/30-5/4 (Figure 15). Detection probability estimates by detection method varied from 0.14 ($se=0.04$), 0.21 ($se=0.04$), and 0.31 ($se=0.05$) for VES, net, and trap, respectively. The top model contains 100% of the model weights.

Plains leopard frog

The most general model in the candidate set, Model 4 $\Psi(\cdot)$, $p(\text{method})$, was used to perform the goodness-of-fit test which indicated no lack of fit (P -value = 0.43, $\hat{c} = 0.55$) so AIC_c values and standard errors were not adjusted. The most supported model for plains leopard frog was Model 4 $\psi(\cdot)$, $p(\text{method})$ (Table 8) with detection probability estimates of 0.21 (se=0.04), 0.05 (se=0.02), and 0.27 (se=0.05) for VES, net, and trap, respectively (Table 9). The model $p(\text{day sq})$ also provides useful information as the peak in detection probability estimates occurred from 6/27 - 7/26 at approximately 0.24. The evidence ratio based on AIC_c weights between the most supported and least supported models (w_1/w_4) in the candidate set is 9987. The top model contains 99.9% of the model weights.

American bullfrog

The most general model in the candidate set, Model 4 $\psi(\cdot)$, $p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit (P -value = 0.0019, $\hat{c} = 1.9$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for the bullfrog was Model 4 $\psi(\cdot)$, $p(\text{method})$ (Table 8) with detection probability estimates of 0.55 (se=0.06), 0.22 (se=0.05), and 0.58 (se=0.06) for VES, net and trap, respectively (Table 9). The bullfrog data does not exhibit a strong quadratic effect by day of survey with the detection probability estimates increasing through the 3 primary sampling periods and peaking at 0.70 in model $p(\text{day})$ and at 0.73 in model $p(\text{day sq})$. The evidence ratio based on $QAIC_c$ weights between the most supported and least most supported model (w_1/w_4) in the candidate set is 3427. The top three models contain 99.9% of the model weights.

Southern leopard frog

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated no lack of fit ($P\text{-value} = 0.34, \hat{c} = 0.91$) so AIC_c values and standard errors were not adjusted. All models for the southern leopard frog included a large positive untransformed beta estimate and an extremely high standard error associated with the occupancy estimate. The solution for this situation was to fix the occupancy parameter to 1 in Program PRESENCE which allows the detection estimates to converge. The most supported model for southern leopard frog was Model 3 $\psi(\cdot), p(\text{day square})$ (Table 8) with the peak of detection probability estimates occurring from 6/26-7/2 at approximately 0.32 (Figure 16). The second most supported model was Model 4 $\psi(\cdot), p(\text{method})$ with estimates of 0.24 (se=0.04), 0.10 (se=0.02), and 0.27(se=0.04) for VES, net, and trap, respectively. The evidence ratio based on AIC_c weights between the most supported and third most supported model (w_1/w_3) in the candidate set is 9882. The top model contains 99% of the model weights.

Leopard frog complex

The most general model in the candidate set, Model 4 $\psi(\cdot), p(\text{method})$, was used to perform the goodness-of-fit test which indicated lack of fit ($P\text{-value} = 0.0001, \hat{c} = 3.7$); therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ (MacKenzie and Bailey 2004), and number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The most supported model for leopard frog complex was Model 3 $\psi(\cdot), p(\text{day square})$ (Table 8) with detection probability estimates peaking from 6/4-6/22 at approximately 0.78 (Figure 17). The second most supported model, $p(\text{method})$, included probability detection estimates of 0.62 (se=0.08), 0.38 (se=0.08), and 0.54 (se=0.08) for VES, net, and trap, respectively. The top model contains 99.9% of the model weights.

APPENDIX G. Evidentiary information supporting the Multi-State Occupancy Data Analysis
Small-mouthed salamander

Due to the truncation procedure used to meet the closure assumption, the data set for the small-mouthed salamander only included 2 primary sampling periods with 3 secondary surveys for a total of 6 sampling periods. Deviance for the most parameterized model in the candidate set, $(\psi^1(\cdot), \psi^2(\cdot), p1(\cdot), p2(\cdot), \delta(\text{method}))$, was 57.4 with 17 df resulting in a \hat{c} value of 3.4 indicating overdispersion. Therefore, AIC_c values were converted to $QAIC_c$, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$, and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). The small-mouthed salamander was detected at 29 of 50 sites for a naïve occupancy estimate of 0.58 and metamorphs were detected at least once on 7 of the 29 sites known to be occupied for a naïve recruitment estimate of 0.24. The naïve estimate of overall probability of successful recruitment was 0.14. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.70 (se = 0.18), a probability estimate of 0.87 (se=0.22) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.67 (se=0.42) given the true state of a site was 1, and a detection probability estimate of 0.26 (se=0.10) given the true state of a site was 2. The probability of detecting successful recruitment, given detection of occupancy, was 0 in primary sampling period one and 0.38 (se=0.24) in primary sampling period two. The model-averaged, unconditional probability that a site successfully produced recruits was 0.61 (se=0.21). The evidence ratio based on $QAIC_c$ weights between the most supported and least supported models (w_1/w_3) in the candidate set is 16.2. The top two models contain 94% of the model weights.

Northern cricket frog

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 315.2 with 30 degrees of freedom (df) resulting in a \hat{c} value of 10.5 indicating overdispersion and/or lack of model fit. The high \hat{c} value likely results from the low expected values in this data set; of the 38 observed unique detection histories, 36 had an expected value <1 . The high value may also indicate inappropriate model structure due to heterogeneity in the data; however, as no covariates were included in the model, I proceeded with the analysis and converted AIC_c values to $QAIC_c$, inflated standard error estimates by a factor of $\sqrt{\hat{c}}$ and augmented the number of parameters by one to account for estimation of c (Burnham and Anderson 2002). Northern cricket frogs were detected at 46 of 50 sites for a naïve occupancy estimate of 0.92 and metamorphs were detected at least once on 10 of the 46 sites known to be occupied for a naïve recruitment estimate of 0.22. The naïve estimate of overall probability of successful recruitment was 0.20. The most supported model included delta as constant (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.96 (se = 0.16), a probability estimate of 0.51 (se=0.57) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.25 (se=0.30) given the true state of a site was 1, and a detection probability estimate of 0.56 (se=0.17) given the true state of a site was 2. The probability of successful recruitment, given detection of occupancy, was 0.13 (se=0.15). The model in which delta varied by primary sampling period was within 1.94 delta units of the most supported model so also receives support. Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.96 (se = 0.15), a probability estimate of 0.47 (se=0.59) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a

detection probability estimate of 0.27 (se=0.28) given the true state of a site was 1, and a detection probability estimate of 0.57 (se=0.17) given the true state of a site was 2. The probability of detecting successful recruitment was zero in primary sampling period one, 0.15 (se=0.21) in primary sampling period two, and 0.18 (se=0.28) in primary sampling period three. The model-averaged unconditional probability estimate that a site successfully produced recruits is 0.48 (se=0.58). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 10.4. The top two models contain 94% of the model weights.

Grey treefrog complex

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 80.2 with 24 degrees of freedom (df) resulting in a \hat{c} value of 3.3 indicating overdispersion. Therefore, AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$, and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Grey treefrogs were detected at 33 of 50 sites for a naïve occupancy estimate of 0.66 and metamorphs were detected at least once on 13 of the 33 sites known to be occupied for a naïve recruitment estimate of 0.39. The naïve estimate of overall probability of successful recruitment was 0.26. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.78 (se = 0.16), a probability estimate of 0.46 (se=0.23) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.24 (se=0.09) given the true state of a site was 1, and a detection probability estimate of 0.32 (se=0.11) given the true state of a site was 2. The probability of successful recruitment was zero in primary sampling period one, 0.64

(se=0.19) in primary sampling period two, and zero in primary sampling period three. The model that included delta as a constant was within 1.66 delta units of the most supported model so also receives support. Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.79 (se = 0.16), a probability estimate of 0.44 (se=0.21) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.22 (se=0.09) given the true state of a site was 1, and a detection probability estimate of 0.34 (se=0.12) given the true state of a site was 2. The probability of detecting successful recruitment, given detection of occupancy, was 0.58 (se=0.19). The model-averaged unconditional probability estimate that a site successfully produced recruits is 0.36 (se=0.18). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 12.7. The top two models contain 95% of the model weights.

***Pseudacris* spp.**

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 85.2 with 17 degrees of freedom (df) resulting in a \hat{c} value of 5.0 indicating overdispersion. Therefore, AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$ and the number of parameters was augmented by one to account for estimation of c (Burnham and Anderson 2002). *Pseudacris* spp. were detected at 37 of 50 sites for a naïve occupancy estimate of 0.74 and metamorphs were detected at least once on nine of the 37 sites known to be occupied for a naïve reproduction estimate of 0.24. The naïve estimate of overall probability of successful reproduction was 0.18. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 1.0 (se

inestimable), a probability estimate of 0.68 (se=0.62) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.09 (se=0.16) given the true state of a site was 1, and a detection probability estimate of 0.27 (se=0.14) given the true state of a site was 2. The probability of detecting successful recruitment was 0.03 (se=0.07) in primary sampling period one, 0.45 (se=0.30) in primary sampling period two, and zero in primary sampling period three. The model-averaged unconditional probability estimate that a site successfully produced recruits is 0.53 (se=0.55). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 3.8. The top two models contain 83% of the model weights.

Plains leopard frog

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 130.5 with 21 degrees of freedom (df) resulting in a \hat{c} value of 6.2 indicating overdispersion. Therefore, AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$, and the number of parameters was augmented by one to account for estimation of c (Burnham and Anderson 2002). Plains leopard frogs were detected at 35 of 50 sites for a naïve occupancy estimate of 0.70 and metamorphs were detected at least once on 16 of the 35 sites known to be occupied for a naïve recruitment estimate of 0.46. The naïve estimate of overall probability of successful recruitment was 0.32. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.87 (se=0.25), a probability estimate of 0.99 (se=0.003) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.02 (se=66.6) given the true state of a site was 1, and a detection probability estimate of 0.07 (se=0.17) given the true state of

a site was 2. The probability of detecting successful recruitment was zero in primary sampling period one, 0.57 (se=0.23) in primary sampling period two, and 0.08 (se=0.14) in primary sampling period three. The model-averaged unconditional probability estimate that a site successfully produced recruits is 0.84 (se=0.31). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 39.1. The top two models contain 98% of the model weights.

American bullfrog

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 290.3 with 27 degrees of freedom (df) resulting in a \hat{c} value of >10 indicating overdispersion and/or lack of model fit. The high \hat{c} value likely results from the low expected values in this data set; of the 35 observed unique detection histories, 33 had an expected value <1. The high value may also indicate inappropriate model structure due to heterogeneity in the data; however, given that there were no covariates in the model, I proceeded with the analysis and converted AIC_c values to QAIC_c, inflated standard error estimates by a factor of $\sqrt{\hat{c}}$, and augmented the number of parameters by one to account for estimation of c (Burnham and Anderson 2002). Bullfrogs were detected at 41 of 50 sites for a naïve occupancy estimate of 0.82 and metamorphs were detected at least once on 13 of the 41 sites known to be occupied for a naïve recruitment estimate of 0.32. The naïve estimate of overall probability of successful recruitment was 0.26. The most supported model included delta as a constant (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 0.86 (se = 0.20), a probability estimate of 0.58 (se=0.38) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.23 (se=0.23) given the true state of a site was 1, and a detection probability estimate of 0.58

(se=0.14) given the true state of a site was 2. The probability of detecting successful recruitment, given detection of occupancy, was 0.16 (se=0.12). The model-averaged unconditional probability estimate that a site successfully produced recruits was 0.49 (se=0.35). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 9.6. The top two models contain 92% of the model weights.

Southern leopard frog

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 224.6 with 24 degrees of freedom (df) resulting in a \hat{c} value of 9.4 indicating overdispersion. Therefore, AIC_c values were converted to QAIC_c, standard error estimates were inflated by a factor of $\sqrt{\hat{c}}$, and the number of parameters were augmented by one to account for estimation of c (Burnham and Anderson 2002). Southern leopard frogs were detected at 45 of 50 sites for a naïve occupancy estimate of 0.90 and metamorphs were detected at least once on 27 of the 45 sites known to be occupied for a naïve recruitment estimate of 0.60. The naïve estimate of overall probability of successful recruitment was 0.54. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 1.0 (se inestimable), a probability estimate of 1.0 (se inestimable) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 1.0 (se inestimable) given the true state of a site was 1, and a detection probability estimate of 0.20 (se=0.58) given the true state of a site was 2. The probability of detecting successful recruitment was zero in primary sampling period one, 0.81 (se=0.17) in primary sampling period two, and 0.08 (se=0.17) in primary sampling period three. The model-averaged unconditional probability

estimate that a site successfully produced recruits is 0.99 (se=0.06). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 374.4. The top model contains 98% of the model weights.

Leopard frog complex

Deviance for the most parameterized model in the candidate set, ($\psi^1(\cdot)$, $\psi^2(\cdot)$, $p1(\cdot)$, $p2(\cdot)$, δ (method)), was 441.1 with 31 degrees of freedom (df) resulting in a \hat{c} value of 14.2 indicating very high overdispersion and/or lack of model fit. The high \hat{c} value likely results from the low expected values in this data set and may also indicate inappropriate model structure due to heterogeneity in the data; however, given that there were no covariates in the model, I proceeded with the analysis. I used a value of 10 for \hat{c} as 10 is the highest value for a variance inflation factor accepted by Program MARK. I converted AIC_c values to QAIC_c, inflated standard error estimates by a factor of $\sqrt{\hat{c}}$, and augmented the number of parameters by one to account for estimation of c (Burnham and Anderson 2002). Leopard frogs were detected at 48 of 50 sites for a naïve occupancy estimate of 0.96 and metamorphs were detected at least once on 26 of the 48 sites known to be occupied for a naïve recruitment estimate of 0.54. The naïve estimate of overall probability of successful recruitment was 0.52. The most supported model was the one in which delta varied by primary sampling period (Table 10). Real parameter estimates included an occupancy probability estimate, irrespective of life history state, of 1.0 (se inestimable), a probability estimate of 0.95 (se=0.11) that successful recruitment (detection of metamorphs) occurred, given that a site was occupied, a detection probability estimate of 0.03 (se=0.19) given the true state of a site was 1, and a detection probability estimate of 0.52 (se=0.08) given the true state of a site was 2. The probability of detecting successful recruitment was zero in primary sampling period one, 0.27 (se=0.13) in primary sampling period two, and 0.08 (se=0.12) in

primary sampling period three. The model-averaged unconditional probability estimate that a site successfully produced recruits is 0.94 (se=0.20). The evidence ratio based on QAIC_c weights between the most supported and least supported models (w_1/w_3) in the candidate set is 43.2. The top two models contains 98% of the model weight.

APPENDIX H. Site averages of variables used in k-means cluster analysis to determine differences and similarities between Wetlands Reserve Program sites classified according to design strategy. Variables include area of site (hectares), average proportion of sampled quadrats dry (dry), average proportion of sampled quadrats wet with grass-like vegetation (Grass), average proportion of sampled quadrats covered with open water (Ow), and average water depth (Depth) on sampled quadrats. Data collected during 2007 field season for amphibian occupancy and species richness project conducted in Lower Grand River basin, north-central Missouri

Site	Area (ha)	Dry	Grass	Ow	Depth
m6	64.9	22.9	35.5	25.6	37.7
m10	66.8	94.0	2.6	3.4	29.7
m12	20.1	86.8	2.0	7.4	26.2
m18	17.5	86.0	5.4	0.0	11.5
m21	32.5	63.3	17.2	8.1	25.1
m22	13.1	89.0	2.3	7.6	13.8
m26	8.2	22.6	1.7	61.3	32.1
m28	21.4	72.1	11.9	8.3	18.2
m32	43.3	81.2	2.4	5.6	18.8
m34	29.7	43.3	9.8	32.5	40.3
m35	44.6	94.5	2.7	1.0	14.2
m38	56.8	93.2	2.0	2.0	12.9
m43	28.0	71.6	8.1	7.9	24.7
m44	4.5	37.0	2.8	44.0	42.0
m46	6.3	80.4	6.9	6.1	11.5
m47	31.2	83.5	10.6	4.8	14.3
m49	11.6	93.9	0.4	0.6	24.1
m50	29.3	35.1	27.8	18.4	31.9
n2	242.9	76.9	4.5	14.2	23.3
n3	43.4	56.9	10.6	15.2	34.6
n5	213.1	66.6	8.0	10.3	27.8
n6	101.4	29.5	26.4	32.8	32.9
n7	60.8	44.7	7.6	33.1	36.2
n8	14.9	42.9	23.3	18.7	26.5
n10	89.0	39.0	12.1	16.8	30.3
n11	71.9	47.9	13.9	13.1	41.0
n13	104.3	81.4	9.9	1.1	19.1
n16	45.7	73.0	5.4	7.5	22.7
n17	102.4	54.2	7.6	8.8	30.6
n18	57.1	62.7	15.6	7.3	28.1
n20	67.9	32.8	18.1	26.1	22.3
n22	109.3	65.0	5.0	25.0	31.6
n24	84.6	45.1	38.9	7.3	19.6
n26	55.4	80.0	11.9	4.1	17.2
n29	21.8	82.5	1.1	6.3	22.9
n31	7.7	49.6	16.6	28.8	26.5

Site	Area (ha)	Dry	Grass	Ow	Depth
n33	140.1	76.4	6.1	14.3	20.3
w1	30.3	52.5	21.2	15.1	30.4
w3	16.9	84.8	0.0	10.4	26.7
w4	15.2	90.7	1.7	2.1	28.5
w9	13.0	97.8	0.0	1.3	24.5
w10	10.6	68.3	5.6	13.8	23.3
w12	3.3	45.4	20.3	9.2	20.8
w13	14.6	99.9	0.0	0.1	3.7
w14	23.4	93.9	1.7	0.8	11.1
w18	12.9	98.7	0.0	1.0	0.0
w19	18.0	92.4	1.1	0.0	11.8
w20	7.0	36.4	22.3	14.1	24.4
w21	6.4	97.4	0.4	0.0	4.1
w22	29.3	88.8	0.0	9.5	37.9