



# Historical and Contemporary Characteristics and Waterfowl Use of Illinois River Valley Wetlands

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Received: 1 June 2009 / Accepted: 9 February 2010 / Published online: 7 May 2010  
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**Abstract** Understanding changes in characteristics of floodplain wetlands over time could provide valuable information to guide management and restoration efforts. We compared characteristics of 29 Illinois River valley (IRV) wetlands mapped during two time periods between 1938–1959 and 15 wetlands re-mapped in 2005–2006. Average proportions of wetlands classified as bottomland forest, scrub-shrub, nonpersistent emergent, and mud flat were generally greater during 2005–2006 than 1938–1942 or 1943–1959, but proportions of aquatic-bed and floating-leaved vegetation declined significantly by 2005–2006. We also modeled wetland use by mallards (*Anas platyrhynchos*) and diving ducks (Tribe Aythyini) during falls 1950–1959 in relation to wetland characteristics. Proportion of wetlands classified as nonpersistent emergent and an interspersed-juxtaposition index (IJI) positively associated with mallard use, whereas proportions of scrub-shrub and persistent emergent vegetation influenced diving duck use negatively. Use by both groups associated positively with wetland area and refuge. The loss of submersed and floating-leaved aquatic vegetation emphasizes the need to restore conditions that promote diverse plant communities in IRV wetlands. Composition and arrangement of wetland habitats (indicated by IJI) may be an important attractant to migrating mallards and perhaps a consideration when planning and evaluating wetland conservation efforts in mid-migration regions.

**Keywords** *Anas platyrhynchos* · Dabbling ducks · Degradation · Diving ducks · Restoration

## Introduction

The watersheds of the Illinois, Mississippi, Missouri, and Ohio rivers of the upper Midwest have undergone significant anthropogenically induced changes during the 20th century. Although many of these changes affected the river channel itself (e.g., dredging and channelization), floodplain wetlands typically realized the greatest disturbances (Bellrose et al. 1983; Sparks 1995). When the natural hydrologic ebb and flow of large floodplain rivers are altered, their lakes, backwaters, and wetlands may suffer considerable degradation and become vulnerable to human development (e.g., conversion to croplands; Havera 1999).

Despite these watershed modifications, most large river systems and their floodplain wetlands in the midcontinent region remain critical habitats for migrating waterbirds. Of these systems, the Illinois River valley (IRV) is of primary importance to waterfowl and a focus area of the Upper Mississippi River and Great Lakes Region Joint Venture of the North American Waterfowl Management Plan (UMRGLRJV Management Board 1998). Emphasizing its historical importance to waterfowl, 1.6 million mallards (*Anas platyrhynchos*) were counted during one aerial inventory in the IRV in 1948 and peak abundance of lesser scaup (*Aythya affinis*) exceeded 500,000 prior to the mid-1950s (Havera 1999). Havera (1999) estimated that on average 21% of the Mississippi Flyway wintering mallard population spent at least one day in the IRV during 1955–1996.

Extensive leveeing and drainage has eliminated 53% of the natural wetlands in the IRV and existing wetlands have

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been further degraded by sedimentation, exotic species, and eutrophication (Havera 1999). Fortunately, restoration and reclamation efforts are ongoing in attempts to return structure and function to IRV floodplain wetlands. A commonly cited objective of ecological restoration is to return ecosystems to conditions that approximate previous conditions (National Research Council 1992; Zedler 2000). However, an unbiased evaluation of restoration success is difficult without historical data, which rarely exists or is subjective. Use of wetlands by waterbirds, particularly waterfowl (Anatidae), may serve as an indicator of landscape condition or a measure of restoration success (Austin et al. 2001; Gawlik 2006). Thus, wetland restoration efforts in the IRV and other large river systems would benefit from quantitative information on historical wetland conditions and identification of wetland characteristics associated with waterfowl abundance.

To provide information to aid planning, implementation, and evaluation of restorations of floodplain wetlands, we compiled and analyzed geospatial data from 140 maps of vegetation for 29 unique IRV wetlands produced during 1938–1959. We remapped 15 of these wetlands during 2005–2006 and used these data to compare wetland characteristics among time periods. Finally, we used historical and contemporary waterfowl abundance data collected during fall aerial inventories to identify wetland characteristics associated with waterfowl use. Our specific objectives were to: 1) test for changes in characteristics (e.g., vegetation composition) of selected IRV bottomland lakes among historical and contemporary mapping periods; and 2) model use of IRV wetlands by mallards and diving ducks (Tribe Aythyini) in relation to historical and contemporary wetland characteristics.

## Methods

### Study Area

We conducted research on bottomland lakes located in the Peoria and La Grange reaches of the Illinois River (river miles 80.2–231.1). These floodplain wetlands and their importance to migratory waterfowl have been described in detail (Bellrose et al. 1983; Havera and Bellrose 1984; Havera 1999). Study wetlands were distinct floodplain lakes with associated bottomland forests and marshes that flanked the Illinois River and were generally bounded by the mainstem of the river and the upland bluff (Stafford et al. 2007). Wetland ownership was both private and public. Privately-owned wetlands were managed by waterfowl hunting clubs (Havera 1999), sometimes with multiple entities per wetland. Public properties were managed by the U.S. Fish and Wildlife Service and Illinois Department of

Natural Resources (IDNR) as stopover habitats for migratory waterfowl and/or waterfowl hunting areas.

### Development of Historical Database

Historical maps of wetland vegetation for the period of 18 July to 16 October were produced by F. C. Bellrose (1938–1953 and 1959) and F. Loomis (1955–1957) of the Illinois Natural History Survey (INHS). Bellrose and Loomis used field triangulation to plot vegetation on 1933 U.S. Army Corps of Engineers (USACOE) maps of 1:12000 scale (Bellrose 1941; Bellrose et al. 1979). Maps of wetland vegetation for August 1959 were produced from aerial photographs interpreted by Bellrose et al. (1979).

We digitally scanned historical vegetation maps and georectified images using ERDAS Imagine Orthobase 8.6 and ArcGIS 9.2 software projected in the UTM coordinate system using NAD 1983, Zones 15 and 16 (Environmental Systems Research Institute 1996). We digitized areas of wetland vegetation using on-screen digitizing features in ArcGIS 9.2 and corrected georectification errors based on the 1933 USACOE maps when necessary. We calculated the area of vegetation polygons using the XTools Pro 4.1 extension for ArcGIS (DATA East, LLC 2006).

### Mapping Contemporary Wetland Characteristics

We ranked wetlands by the number of years they were mapped during 1938–1959 and selected the top 15 wetlands for contemporary vegetation mapping. We mapped vegetation of eight wetlands during 2005 and seven in 2006.

To map contemporary vegetation patterns, we identified wetland boundaries from historical maps, present-day bluff lines, and the waterline of the Illinois River or its side channels. We mapped wetland vegetation using line transects (north–south or east–west UTM lines) spaced every 300 m along wetland boundaries. We traversed transects on foot, all-terrain vehicle, or by boat. Wetland mapping began after most wetland plants produced seed heads to aid identification (~Aug. 1), and we delineated changes in vegetation composition (e.g., scrub-shrub, bottomland forest) along transect lines using a handheld global positioning system (GPS). We digitized wetland vegetation in ArcGIS 9.2 using the GPS waypoints (supplemented with field notes and photographs) overlaid on 2005 and 2006 aerial photos obtained from the United States Department of Agriculture's Geospatial Data Gateway (<http://datagateway.nrcs.usda.gov/>).

### Compilation of Wetland Characteristics

We categorized wetland vegetation documented on historical and contemporary maps into nine zones by grouping

species of similar life forms or by the absence of vegetation (i.e., open water; OPENH2O) based on Cowardin et al. (1979). We chose categories that represented broad-scale wetland habitats important to migrating waterfowl (Suloway and Hubbell 1994; U.S. Fish and Wildlife Service 2007). Specifically, we categorized woody vegetation as bottomland forest (FOREST) if trees were >6 m in height or scrub-shrub (SCSH) if ≤6 m tall (Cowardin et al. 1979). Other wetland classifications were nonpersistent emergent vegetation (NPE; e.g., moist-soil plants; Fredrickson and Taylor 1982), persistent emergent vegetation (PE; e.g., cattails [*Typha* spp.] and bulrushes [*Scirpus* spp.]), mud flats (MUD), floating-leaved aquatic vegetation (FLOAT; e.g., American white waterlily [*Nymphaea odorata*]), and aquatic bed (AB; e.g., coontail [*Ceratophyllum demersum*]). Finally, we classified areas with agriculture (e.g., corn), typically wildlife food plots, as croplands (CROP).

We examined maps for continuity of site boundaries prior to compiling wetland characteristics for analyses. Each map included slightly different areas about the periphery of wetlands; thus, we clipped wetland maps for each site and year combination to include only the primary basin in ArcGIS 9.2. We summed areas (ha) of each of the aforementioned wetland habitat categories by site and year and converted them to proportions by dividing each category total by wetland area. Further, we computed the relative richness (RR) of habitat types by dividing the number of wetland habitats present in each map by the total possible habitat types, excluding CROP ( $n=8$ ). Wetlands with diverse habitat types distributed throughout their basins may be more attractive to waterfowl than those with contagious distributions (Weller and Spatcher 1965). Therefore, we computed the Interspersion-Juxtaposition Index (IJI) for individual wetlands (McGarigal and Marks 1995). Values of IJI increase as habitat patches tend to be more evenly interspersed and adjacent to other patches and constitute an index of habitat heterogeneity. We converted wetland maps from polygons to grids (10 m cells) in ArcGIS 9.2, imported grids into ArcView 3.3, and calculated IJI values using the Patch Analyst v3.0 extension (Rempel and Carr 2003) via the FRAGSTATS interface (McGarigal et al. 2002).

#### Aerial Inventories of Waterfowl

Aerial inventories of waterfowl in the IRV were conducted weekly (weather permitting) during falls (1 September–15 December) 1948–2000 and 2002–2006 from a fixed-wing, single-engine aircraft at altitudes of 60–140 m and speeds of 160–240 km/hr (Havera 1999:186). During each count, a single observer recorded abundances of all waterfowl species present except Wood Ducks (*Aix sponsa*). Invento-

ried locations in the IRV were distinct floodplain lakes and associated wetlands that flanked the Illinois River (see Bellrose et al. 1979, 1983; Havera 1999). We did not collect habitat-specific data on wetland use by waterfowl. Rather, we estimated waterfowl abundance for the entire area of each wetland, such that each site was surveyed as a discrete unit (Stafford et al. 2007).

#### Statistical Analyses

##### *Change in Wetland Characteristics*

We intended to analyze changes in wetland composition over time, but habitat proportions were not independent due to the unit-sum constraint. Compositional analysis transforms proportional dependent variables to log-ratios and accounts for this lack of independence (Aebischer et al. 1993), but our data set contained many zeros, and this approach may have lead to severely inflated Type I error rates (Bingham and Brennan 2004; Badzinski and Petrie 2006). Examination of residual plots indicated our errors were not multivariate-normal distributed and arcsine square-root transforming the data did not significantly improve error distributions and complicated interpretability. Therefore, we selected an analytical approach similar to that of wildlife food-habits studies (Afton et al. 1991; Ross et al. 2005; Badzinski and Petrie 2006) and analyzed change in wetland habitat composition using multivariate analysis of variance (MANOVA) with simple proportions as the dependent variable. We acknowledge deviations from statistical assumptions, but consider our tests appropriate because parametric multivariate analyses are considered robust to many violations of assumptions of linear models (Johnson 1995).

The dependent variables in the MANOVA model included RR, IJI, and the proportion of each of the nine previously described wetland habitat types present in each map. We categorized mapping periods (independent variable) as early (1939–1942) and late (1943–1959) historical and contemporary (2005–2006). We chose to separate historical maps into pre- and post-1942 categories because the largest flood on record in the IRV occurred in the spring of 1943 and Bellrose et al. (1979) noted considerable changes in wetland characteristics following the flood. The number of wetlands mapped each year varied and we only included data from wetlands mapped in ≥2 of the time periods. Therefore, we used data from 103 maps of 16 unique sites in the MANOVA ( $n=43$  [early historical], 45 [late historical], and 15 [contemporary]). We conducted our analysis using the MANOVA statement in PROC GLM, SAS v9.1.3, and included wetland location as a random effect to account for dependence among characteristics within individual wet-

lands (SAS Institute 2004). We used Wilk's Lambda to evaluate statistical significance of the MANOVA because it is considered robust to violations of the assumption of multivariate normality (Badzinski and Petrie 2006). If results indicated a significant ( $\alpha < 0.10$ ) difference in wetland characteristics among time periods, we conducted Tukey-Kramer *post-hoc* means comparison tests using the PDIFF option of the LSMEANS statement.

#### *Waterfowl Use in Relation to Wetland Characteristics*

We used the proportion of total wetland area categorized as each of the aforementioned wetland habitat categories (excluding CROP) and IJI as covariates to explain variation in waterfowl use during historical and contemporary mapping periods (Cowardin et al. 1979). In all models, we included covariates to control for the categorical proportion of a site where disturbance was prohibited (i.e., 0–25%, 26–50%, 51–75%, and  $\geq 76\%$ ; REFUGE; Stafford et al. 2007) and wetland size (ha; AREA;  $\bar{x} = 503.6 \pm 69.4$  ha). Information on REFUGE was obtained from interviews with IDNR biologists, site managers, and unpublished historical records held by the INHS. We included each habitat covariate individually (with REFUGE and AREA) and in models representing abundances of: 1) wetland plants that provide waterfowl food (NPE + AB + FLOAT); 2) woody vegetation (FOREST + SCSH); 3) low, dense thermal cover (SCSH + PE); 4) submersed and floating-leaved aquatic vegetation (AB + FLOAT); and 5) interspersed emergent hydrophytes (e.g., IJI + NPE + PE). We did not fully parameterize any model to account for the unit-sum constraint.

We used aerial inventory data on abundance of mallards and diving ducks ( $n = 13$  wetlands, 36 wetland-year combinations) to compute the dependent variable of use-days (UD) for the period 1 October to 15 December 1950–1959 following Stafford et al. (2007). Use-days are an estimate of total use extrapolated over a period of interest. For example, 100 birds using a wetland for 10 days equates to 1,000 UD. During 1950–1959, mallard UD per wetland averaged 1,807,358 (range: 9,750–10,294,900 UD), whereas diving duck UD per wetland averaged 31,227 (range: 0–319,548 UD). Diving duck abundance was limited during 2005–2006 aerial inventories; therefore, we only modeled mallard UD during the contemporary period. Abundance data were not available for all mapped locations in 2005–2006, so we used data from nine wetlands (four mapped in 2005, five mapped in 2006) in contemporary models of mallard UD ( $\bar{x} = 559,188$  UD; range: 82,300–1,112,600 UD).

Regardless of mapping period, we modeled fall UD using the maximum likelihood estimation method in the MIXED procedure, SAS v9.1.3 (SAS Institute 2004). We

used variance inflation factor (VIF) diagnostics to evaluate collinearity among covariates in candidate models and found no evidence of substantial intercorrelation (i.e.,  $VIF \leq 1.73$ ; PROC REG; SAS Institute 2004). For 1950–1959 models only, we accounted for correlation in waterfowl use among sites over time by including wetland location nested in YEAR in the REPEATED statement. We determined best approximating and competing models from our candidate set using second-order Akaike's Information Criterion ( $AIC_c$ ; Burnham and Anderson 2002). We considered models competitive within candidate sets if they were within  $\sim 2.0$   $AIC_c$  units of the best model. When model separation was poor, we model-averaged parameter estimates of variables that appeared in multiple competing models (weighted by model weight,  $w_i$ ; Burnham and Anderson 2002). We interpreted precision of covariates by calculating 95% confidence intervals about parameter estimates. To evaluate model fit, we regressed observed and predicted values for each candidate model to estimate the coefficient of determination ( $R^2$ ; Piñeiro et al. 2008).

## Results

### Change in Wetland Characteristics

Results of our MANOVA revealed significant change in wetland characteristics over the three time periods (MANOVA; Wilks'  $\lambda = 0.38$ ;  $F_{20, 182} = 5.65$ ,  $P < 0.001$ ). Contrasts of least-squares means indicated significantly greater ( $P < 0.10$ ) proportions of wetland classified as FOREST, SCSH, and MUD during the contemporary period compared to the early and late historical periods; however, proportions of these categories did not differ between historical periods (Table 1). Conversely, proportion of wetland area classified as AB was significantly less during 2005–2006 than either historical period. Contemporary wetlands contained significantly less proportions of FLOAT than during the early, but not late, historical period (Table 1). Proportion of NPE wetland increased significantly between the early and late historical periods but did not differ between the late historical and contemporary periods. In general, the trend in NPE increased with each time-period classification (Table 1). In contrast to NPE, proportion of wetland area classified as PE declined between early and late historical periods, but was similar between late historical and contemporary mapping (Table 1). Proportion of OPENH2O wetland and RR did not vary significantly over time. Finally, IJI was significantly greater in the early historical mapping period than the late historical, but average contemporary IJI was similar to both historical periods (Table 1).

**Table 1** Results of MANOVA intended to explain variation in wetland composition over time. Least squares means and standard errors are proportions of wetland area. Mapping periods refer to early (1938–1942) and late (1943–1959) historical and contemporary (2005–2006)

Wetland category (Abbreviation)	Time period					
	1938–1942		1943–1959		2005–2006	
	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Bottomland forest (FOREST)	8.8A <sup>a</sup>	1.3	8.2A	1.3	15.3B	2.2
Nonpersistent emergent (NPE)	12.4A	2.8	21.3B	2.8	32.5B	4.8
Open water (OPENH2O)	38.7A	3.9	41.7A	3.8	37.6A	6.6
Aquatic bed (AB)	11.2A	2.6	14.1A	2.5	<0.1B	4.4
Floating-leaved aquatic (FLOAT)	14.9A	2.2	7.2B	2.1	<0.1B	3.7
Mud flat (MUD)	0.4A	0.2	0.1A	0.2	1.7B	0.3
Persistent emergent (PE)	12.3A	2.0	5.3B	2.0	3.9B	3.4
Scrub-Shrub (SCSH)	1.3A	0.5	2.2A	0.5	5.2B	0.8
Cropland (CROP)	0.0A	0.4	0.0A	0.4	3.7B	0.7
Relative richness (RR)	69.8A	2.1	64.2A	2.0	70.8A	3.5
Interspersion-Juxtaposition index (IJI)	69.6A	1.8	63.7B	1.7	65.8AB	3.0

<sup>a</sup> Values with different letters within wetland or quality index category (rows) indicate significant differences of least-squares means (Tukey-Kramer test:  $P \leq 0.10$ ). Contrasts were based on the full multivariate model (Wilks'  $\lambda = 0.38$ ;  $F_{20, 182} = 5.65$ ,  $P < 0.001$ )

### Mallard Use 1950–1959

Two of 14 models of mallard UD during falls 1950–1959 were considered competitive ( $\Delta AIC_c < 2.0$ ) and accounted for 80.4% of model weight ( $w_i$ ; Table 2). Averaged across competing models, total mallard UD were positively associated with AREA ( $\hat{\beta}_{AREA} = 4,235$ ; 95% CI = 2,997 to 5,473), IJI ( $\hat{\beta}_{IJI} = 49,421$ ; 95% CI = 11,437 to 87,404), and REFUGE ( $\hat{\beta}_{REFUGE} = 312,127$ ; 95% CI = -226,586 to 850,841), although the confidence interval about REFUGE overlapped zero and indicated the effect was equivocal. NPE occurred in the best model and was positively associated with mallard UD ( $\hat{\beta}_{NPE} = 30,728$ ; 95% CI = 2,818 to 58,638). Percent PE was negatively associated with the dependent variable ( $\hat{\beta}_{PE} = -31,039$ ; 95% CI = -77,072 to 14,994), but the 95% CI included zero.

### Mallard Use 2005–2006

The best approximating model of mallard UD during 2005–2006 included only the covariates of REFUGE and AREA and accounted for 97.3% of model weight (Table 3). Intuitively, AREA ( $\hat{\beta}_{AREA} = 195$ ; 95% CI = -250 to 641) and REFUGE ( $\hat{\beta}_{REFUGE} = 222,897$ ; 95% CI = 61,628 to 384,166) were positively associated with mallard UD, although the lower confidence limit of the former parameter estimate was less than zero. No other model in the candidate set was competitive.

### Diving Duck Use 1950–1959

Three of 14 models of diving duck UD during falls 1950–1959 were competitive and accounted for 59.5% of model weight ( $w_i$ ; Table 4). Averaged across competing models, AREA indicated a positive association with fall diving duck

UDs ( $\hat{\beta}_{AREA} = 80$ ; 95% CI = 23 to 125). The model-averaged parameter estimate for REFUGE ( $\hat{\beta}_{REFUGE} = 10,782$ ; 95% CI = -8,233 to 29,796) was also positive, but the 95% confidence interval included zero. The second-best model contained the main effect of NPE ( $\hat{\beta}_{NPE} = 1,036$ ; 95% CI = -2 to 2,075), and the parameter estimate indicated a positive but tenuous association with UD. Model-averaged PE ( $\hat{\beta}_{PE} = -1,964$ ; 95% CI = -3,712 to -215) was negatively associated with diving duck UD, as was SCSH ( $\hat{\beta}_{SCSH} = -7,068$ ; 95% CI = -13,576 to -560), which occurred in the best model.

## Discussion

### Change in Wetland Composition

Our results indicated subtle changes in wetland composition between the early and late historical periods and more pronounced changes by the contemporary period, nearly five decades later. Although not all contrasts were significant, proportion of wetlands classified as NPE generally increased across mapping periods, averaging 8.9% greater (i.e., a 72% increase) during the late than early historical and 20.1% greater (i.e., a 162% increase) during the contemporary than early historical period. We reason that the trend in NPE may have been due to: 1) increased management for moist-soil vegetation to attract waterfowl; and 2) stochastic conditions (e.g., sedimentation, drought) prior to and during the contemporary period that provided favorable hydrology for NPE vegetation.

In support of the management option, NPE wetland was comprised of annual vegetation that typically produces seeds consumed by waterfowl (e.g., moist-soil plants; Low and Bellrose 1944; Fredrickson and Taylor 1982). Bellrose

**Table 2** Candidate models to explain variation in use-days by mallards during fall (1 October–15 December) at locations mapped and inventoried aerially for waterfowl during 1950–1959, ranked by second order Akaike's information criterion ( $AIC_c$ ). Also included are the number of estimable parameters ( $K$ ),  $-2$  log likelihood score ( $-2$  Log), model weight ( $w_i$ ), and coefficient of determination ( $R^2$ )

Model	$K$	$-2$ Log	$AIC_c$	$\Delta AIC_c$	$w_i$	$R^2$
REFUGE + IJI + NPE + PE + AREA	7	1111.5	1129.5	0.0	0.481	0.725
REFUGE + IJI + AREA	5	1118.3	1130.3	0.8	0.323	0.668
REFUGE + NPE + AREA	5	1120.1	1132.1	2.6	0.131	0.651
REFUGE + PE + AREA	5	1124.5	1136.5	7.0	0.015	0.605
REFUGE + AREA	4	1127.3	1136.6	7.1	0.014	0.574
REFUGE + AB + AREA	5	1125.7	1137.7	8.2	0.008	0.592
REFUGE + NPE + AB + FLOAT + AREA	7	1119.9	1137.9	8.4	0.007	0.653
REFUGE + FLOAT + AREA	5	1127.1	1139.1	9.6	0.004	0.576
REFUGE + SCSH + AREA	5	1127.2	1139.2	9.7	0.004	0.574
REFUGE + FOREST + AREA	5	1127.2	1139.2	9.7	0.004	0.574
REFUGE + OPENH2O + AREA	5	1127.2	1139.2	9.7	0.004	0.574
REFUGE + SCSH + PE + AREA	6	1124.4	1139.3	9.8	0.004	0.606
REFUGE + AB + FLOAT + AREA	6	1125.7	1140.6	11.1	0.002	0.593
REFUGE + FOREST + SCSH + AREA	6	1127.2	1142.1	12.6	0.001	0.574

et al. (1979) suggested that wetland area in the IRV with the potential to control hydrology and grow moist-soil vegetation had increased due to ongoing development of wetlands (e.g., construction of levees, water-control structures, etc.) by private hunting clubs, USACOE, USFWS, and IDNR. This trend continued into the contemporary mapping period (e.g., U.S. Army Corps of Engineers 2004).

Bellrose et al. (1979) reported frequent water-level fluctuations in the IRV resulted in only 3–20% of the basin area developing moist-soil vegetation, but under favorable hydrologic conditions  $\geq 44\%$  of the wetland area could produce moist-soil plants. The IRV experienced a significant drought during summer 2005 and a less-severe drought in 2006 (MRCC 2007). These stochastic weather events allowed many wetlands to dry, exposing and maintaining mud flats that allowed for moist-soil plant growth, particularly in 2005. Thus, we reason that increased area

of NPE during the contemporary period was at least partially a function of drought conditions.

Proportion of wetland area classified as AB did not differ statistically between early and late historical mapping periods, whereas proportional area of FLOAT declined 51.7% during the interval. Our results indicated that AB and FLOAT were once relatively abundant in IRV wetlands, accounting for 26.1% and 21.3% of wetland area during the early and late historical mapping periods, respectively. In contrast, wetlands were nearly devoid of AB and FLOAT during 2005–2006 ( $<0.1\%$ ; both categories). The sparse abundance of AB and FLOAT observed during contemporary mapping was not likely due to droughts of 2005 and 2006. Considerable evidence exists that these wetland habitat types largely disappeared from IRV wetlands about 1960 (e.g., Mills et al. 1966; Havera 1999:87–90). Further, sampling by the INHS under the Long Term Resource

**Table 3** Candidate models to explain variation in use-days by mallards during fall (1 October–15 December) at locations mapped and inventoried aerially for waterfowl during 2005–2006, ranked by second order Akaike's information criterion ( $AIC_c$ ). Also included are the number of estimable parameters ( $K$ ),  $-2$  log likelihood score ( $-2$  Log), model weight ( $w_i$ ), and coefficient of determination ( $R^2$ )

Model	$K$	$-2$ Log	$AIC_c$	$\Delta AIC_c$	$w_i$	$R^2$
REFUGE + AREA	4	251.3	269.3	0.0	0.973	0.480
REFUGE + PE + AREA	5	249.5	279.5	10.2	0.006	0.572
REFUGE + IJI + AREA	5	250.1	280.1	10.8	0.004	0.543
REFUGE + AB + AREA	5	250.6	280.6	11.3	0.003	0.515
REFUGE + FOREST + AREA	5	250.9	280.9	11.6	0.003	0.499
REFUGE + FLOAT + AREA	5	250.9	280.9	11.6	0.003	0.503
REFUGE + NPE + AREA	5	251.1	281.1	11.8	0.003	0.491
REFUGE + SCSH + AREA	5	251.3	281.3	12.0	0.002	0.480
REFUGE + OPENH2O + AREA	5	251.3	281.3	12.0	0.002	0.481
REFUGE + SCSH + PE + AREA	6	248.4	302.4	33.1	0.000	0.624
REFUGE + AB + FLOAT + AREA	6	249.5	303.5	34.2	0.000	0.572
REFUGE + FOREST + SCSH + AREA	6	250.6	304.6	35.3	0.000	0.499
REFUGE + NPE + AB + FLOAT + AREA	7	248.9	374.9	105.6	0.000	0.601
REFUGE + IJI + NPE + PE + AREA	7	249.1	375.1	105.8	0.000	0.594

**Table 4** Candidate models to explain variation in use-days by diving ducks during fall (1 October–15 December) at locations mapped and inventoried aerially for waterfowl during 1950–1959, ranked by second order Akaike's information criterion ( $AIC_c$ ). Also included are the number of estimable parameters ( $K$ ),  $-2$  log likelihood score ( $-2$  Log), model weight ( $w_i$ ), and coefficient of determination ( $R^2$ )

Model	$K$	$-2$ Log	$AIC_c$	$\Delta AIC_c$	$w_i$	$R^2$
REFUGE + SCSH + PE + AREA	6	875.3	890.2	0.0	0.311	0.436
REFUGE + NPE + AREA	5	879.4	891.4	1.2	0.170	0.368
REFUGE + PE + AREA	5	880.2	892.2	2.0	0.114	0.354
REFUGE + AREA	4	883.5	892.8	2.6	0.085	0.292
REFUGE + SCSH + AREA	5	881.2	893.2	3.0	0.069	0.335
REFUGE + IJI + NPE + PE + AREA	7	875.6	893.6	3.4	0.057	0.431
REFUGE + IJI + AREA	5	881.7	893.7	3.5	0.054	0.326
REFUGE + FLOAT + AREA	5	882.9	894.9	4.7	0.030	0.303
REFUGE + FOREST + AREA	5	883.4	895.4	5.2	0.023	0.293
REFUGE + AB + AREA	5	883.5	895.5	5.3	0.022	0.292
REFUGE + OPENH2O + AREA	5	883.5	895.5	5.3	0.022	0.292
REFUGE + FOREST + SCSH + AREA	6	880.7	895.6	5.4	0.021	0.345
REFUGE + NPE + AB + FLOAT + AREA	7	878.1	896.1	5.9	0.016	0.390
REFUGE + AB + FLOAT + AREA	6	882.9	897.8	7.6	0.007	0.303

Monitoring Program during 1998–2002 documented no aquatic vegetation in the mainstem or connected backwaters of the LaGrange Pool of the Illinois River (Yin and Langrehr 2005).

Efforts to restore aquatic bed and floating-leaved aquatic vegetation in IRV wetlands have been largely unsuccessful (Yin et al. 2001) because their establishment is hindered in bottomland lakes connected to the Illinois River by extensive sedimentation (Starrett and Fritz 1965), fluctuating hydrology, and invasive species (e.g., exotic carps; Havera 1999). Nevertheless, many bottomland lakes in the IRV were leveed from the Illinois River in the 1920s and drained for agriculture (Bellrose et al. 1983). Restorations of these isolated former wetlands generally have been successful at establishing aquatic vegetation. For example, Hennepin-Hopper, a 1,050 ha former IRV drainage and levee district, was restored in 2001 and submersed aquatic plants quickly recolonized. Thus, restoration of drainage and levee districts isolated from the Illinois River may be a successful strategy to restore aquatic plants in the region until the detrimental effects of sedimentation, invasive species, and variable hydrology can be mitigated.

Wetlands generally contained greater proportions of bottomland forest and scrub-shrub during the contemporary than historical periods. The Chicago Sanitary and Ship Canal was completed in 1900, which diverted large amounts of water from Lake Michigan to the Illinois River (Cruikshank 1998; Havera 1999). Increased flow effectively doubled the area of bottomland lakes (Bellrose et al. 1979:4) and killed most of the mast-producing bottomland hardwood forest in the upper and middle IRV. Water levels remained high until 1938, when diversion was reduced based on a U.S. Supreme Court decision (Havera et al. 1980:1–5; Havera 1999:87). The increased proportion of woody vegetation likely reflected reestablishment of mesic-

tolerant trees (e.g., cottonwood [*Populus deltoides*], silver maple [*Acer saccharinum*]) after diversion was reduced, but extensive sedimentation also likely contributed to increased relative area of forested wetland in some cases (Bellrose et al. 1983).

Our results further indicated that habitat types within wetlands were more heterogeneously distributed (i.e., greater average IJI) in the early historical mapping period, perhaps due to the losses of wetland components, such as AB and FLOAT, which tended to grow in deeper areas. Ground counts of waterfowl during falls 1939–1942 were also some of the highest recorded in the IRV, but we did not have adequate information on waterfowl abundance during the early historical period to evaluate this potential relationship. Although speculative, wetlands with diverse and heterogeneously distributed habitats, perhaps acting as habitat complexes (*sensu* Fredrickson and Reid 1988), may have been particularly attractive to waterfowl in the early historical period (Kaminski and Prince 1981; Murkin et al. 1982; Smith et al. 2004).

#### Waterfowl Use and Wetland Characteristics

Fit of historical and contemporary mallard UD models was reasonable, accounting for 57–73% and 53–63% of variation in the dependent variables, whereas only 29–44% of variation was explained in diving duck UD models. We recognize that most variation in all models was explained by REFUGE and AREA and inclusion of habitat covariates only improved model fit modestly (e.g., <15% additional variance explained). The best model of mallard UD during 2005–2006 included only the control variables of REFUGE and AREA. Thus, we constrain our discussion to models of mallard and diving duck UD during 1950–1959.

We included REFUGE in candidate models to control for the effect of rest area on waterfowl use, and the positive association with UDs was intuitive and consistent with previous research findings (Cox and Afton 1997; Fox and Madsen 1997; Madsen 1998a, b; Evans and Day 2002; Stafford et al. 2007). Waterfowl refuges are often managed to produce vegetation that provides food and cover for migrating ducks and geese; thus the importance of REFUGE in models may have been due to habitat quality rather than lack of disturbance. However, area of PE and NPE, which may correlate with abundance of waterfowl food and cover, were also good predictors of mallard UDs during 1950–1959. Because a *post hoc* investigation revealed weak relationships between these variables and REFUGE ( $R^2=0.04$  [NPE and PE]), we suggest lack of disturbance best explains the relationship of UDs to REFUGE. The importance of REFUGE in most candidate models emphasizes the attractiveness of fall rest areas to waterfowl in Illinois, which may be particularly important given the losses in wetland habitat diversity and heterogeneity we documented.

We hypothesized positive associations would exist between duck use and areas of AB and FLOAT, but models including these variables were not competitive. This was surprising, because considerable evidence existed that AB and FLOAT largely disappeared from IRV wetlands in the 1960s (e.g., Mills et al. 1966; Havera 1999), coinciding with declines in abundance of migrating waterfowl in the region (Havera 1999). For example, mallard use of the IRV during fall declined from an average of 39.5 million UDs/year during 1950–1959 to 22.9 million UDs/year during 1960–1969 and averaged only 9.0 million UDs/year during 1998–2008 (M. M. Horath, INHS, unpublished data). Average diving duck use declined 95.4% (i.e., from 3.5 million UDs/year to 160,000 UDs/year) between the 1950s and 1960s. Further, efforts to restore AB and FLOAT in IRV wetlands have coincided with increased waterfowl use. During 2002, use of the aforementioned Hennepin-Hopper site by five dabbling duck species (Tribe Anatini) accounted for 44–66% of use by these species in the entire Peoria reach of the Illinois River ( $n=22,772$  ha of wetlands; Havera 1999; Horath and Havera 2007).

We are unable to account for the lack of relationship between these lost wetland habitats and duck use. Perhaps changes in these habitat types and concurrent duck use during our historical period were not dramatic enough to identify associations, if they existed. Other researchers have suggested that declines in fingernail clam abundance (e.g., *Musculium* spp.) were responsible for reduced diving duck use in the IRV (Paloumpis and Starrett 1960; Anderson et al. 1978; Havera and Bellrose 1984). If so, it appears that wetland characteristics used in our analysis were not indicative of fingernail clam abundances, despite requiring

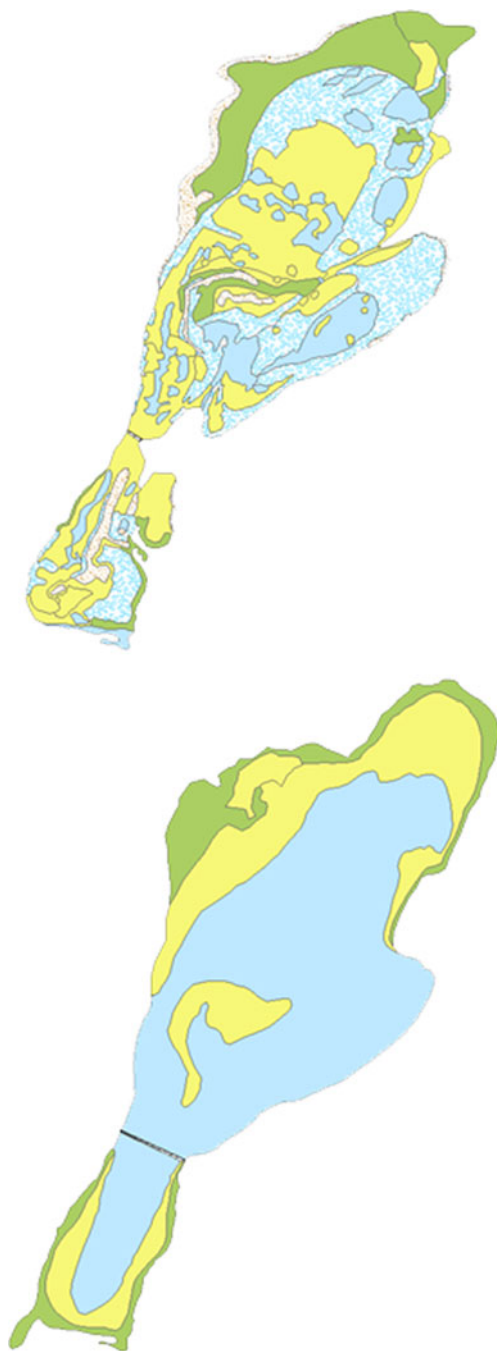
similar wetland conditions as AB (e.g., clear water with consolidated substrates; Havera and Bellrose 1984; Gray et al. 2005).

It is also possible that actual areas of AB and FLOAT did not influence duck use *per se*, but were important in the context of increasing habitat diversity and complexity. We considered wetland complexity in our UD models by including a variable that accounted for the interspersion and juxtaposition of all habitat types. Indeed, IJI was included in the only two competing models of historical mallard UDs. Averaging across models indicated a one-unit increase in IJI would increase UDs by 2.7% above the study-period average. However, interpreting IJI can be difficult, and considerable variation existed during 1950–1959 (Range: 36.6–85.2). For example, the IJI computed for Anderson Lake State Fish and Wildlife Area was relatively high in 1955 (68.1), but near the minimum in 1959 (38.9), demonstrating the variable spatial arrangement of habitat types in this dynamic system (Fig. 1).

Other authors have found positive relationships between complexity of wetland habitats and waterbird use. For example, Kaminski and Prince (1984) also reported that many individual wetland habitat types (e.g., cattail-bulrush) were poor predictors of dabbling duck use of Manitoba wetlands, whereas complexity-related variables (e.g., shoreline development) were relatively more important. Riffell et al. (2003) reported avian abundance and species richness in wet meadows of Michigan increased as landscape context became more complex (i.e., wet meadows adjacent to several patch types). Pearse (2007) reported that mallard abundance during winter in the Mississippi Delta increased with landscape complexity, and also inferred that presence of mallards was associated with increased wetland edge. Indeed, some species of waterfowl are attracted to and forage along wetland edges, such as interfaces between open water and mud flat or aquatic plants (e.g., Kaminski and Prince 1984; Johnson and Rohwer 2000). Although our investigation was unable to separate ecological pattern from process, it is plausible that increasing wetland complexity lead to more suitable foraging sites and, thus, greater “usable space” (*sensu* Guthery et al. 2005).

Finally, a number of studies have suggested that wetlands with interspersed emergent vegetation and open water may maximize use by waterfowl because they allow for spatial segregation that reduces competition and agonistic interactions (Weller and Spatcher 1965; Weller and Fredrickson 1974; Kaminski and Prince 1981; Murkin et al. 1982; Kaminski and Prince 1984; Smith et al. 2004; Brennan 2006). Use of such wetlands may decline when area of emergent cover greatly exceeds open water area or vice-versa (Weller 1978; Smith et al. 2004). The IJI was computed using all wetland habitat categories and does not allow for inference regarding interspersion of open water





**Fig. 1** Maps of Anderson Lake State Fish and Wildlife Area, Illinois, depicting different values of the interspersion and juxtaposition index (IJI). The IJI was greatest in 1955 (top; IJI=68.1), when submersed aquatic vegetation (yellow or medium gray) was spread throughout the wetland and least in 1959 (bottom; IJI=38.9), when most vegetation was contained in the wetland periphery. Maps were produced from ArcGIS shapefiles that were converted to grids (10 m pixels)

and emergent vegetation, but we note that most of the wetland area (67.4%) was comprised of NPE, PE, and OPENH2O during 1950–1959. Regardless, the importance of IJI in historical mallard models supports the concept that arrangement of habitat patches within wetlands may be an

important aspect of attractiveness to mallards, and this relationship warrants further investigation in key migration regions.

Our best model of mallard UD during 1950–1959 also included proportions of area in emergent cover types (NPE and PE). The positive association with NPE was similar to results of Stafford et al. (2007), who identified combined area of PE and NPE as a positive predictor of mallard UD in the IRV during 1977–1987. The best model of mallard UD during 1950–1959 predicted an 8.7% increase in UD if NPE increased 5% above the study-period average ( $\bar{x}$ =22.6%). However, this relationship may not hold in the contemporary landscape of the IRV. Our analysis of change in wetland characteristics over time suggested a significant increase in wetland area classified as NPE, yet abundances of many duck species in the IRV have been stable or declining since the late 1950s (Havera 1999).

Despite increased area of NPE it is possible that quantity or quality of plant foods produced in these areas has declined. For example, Bellrose et al. (1979) did not report the invasive plant, cocklebur (*Xanthium strumarium*), at Chautauqua National Wildlife Refuge, Illinois, during 1939–1959. However, Bowyer et al. (2005) recently documented large stands of cocklebur at the same site that, if untreated (e.g., herbicide, mowing), significantly decreased production of moist-soil plant seeds that provide food for waterfowl. Regardless, NPE appears to have been an important component attracting mallards to IRV wetlands during the 1950s, and we suggest this result supports management practices to promote NPE vegetation as foraging habitat for migratory mallards.

In contrast to NPE, mallard UD declined with increased proportion of PE, although the parameter estimate was variable. Whereas NPE vegetation typically consists of seed-producing annual plants, PE in the IRV often includes dense stands of robust emergents (e.g., river bulrush [*Scirpus fluviatilis*]) that provide benefits to migrating waterfowl such as thermal or escape cover, often at night (*sensu* Magee 1996). Because we conducted aerial surveys during daylight hours, it is possible we simply did not document nocturnal use of wetlands with greater proportions of PE. Alternatively, migrating mallards may have avoided wetlands with more PE because they lacked food resources or only used these wetlands when more desirable habitats were nearby.

No competing model indicated a positive association of wetland habitat characteristics to diving duck UD during falls 1950–1959. The parameter estimate for REFUGE indicated a positive association with UD, which was intuitive given that diving ducks may be particularly susceptible to disturbance (e.g., Thornburg 1973; Korschgen et al. 1985; Havera et al. 1992). Diving duck UD generally declined as proportions of PE and SCSH

increased. Many diving duck species (e.g., lesser scaup, canvasback [*Aythya valisineria*]) are attracted to wetlands with expanses of open water and shallow marsh (Korschgen 1989:159; Paracuellos 2006), and increased PE and SCSH would have reduced open-water foraging sites and perhaps deterred use.

## Implications

We suggest the loss of AB and FLOAT between the late historical and contemporary mapping periods was of greatest detriment to wetland habitat and ecosystem function in the IRV. Because our analyses did not suggest these habitat types were specifically associated with historical or contemporary duck use, support for restoration of these plant communities cannot be based solely on their historical value to waterfowl. Nonetheless, their loss decreased plant diversity and influenced habitat heterogeneity and may have impacted waterfowl use indirectly. We believe the overall weight of evidence supports restoring conditions (e.g., improved water quality) that promote growth of aquatic bed and floating-leaved aquatic plants in IRV wetlands. Such efforts would provide significant benefits to myriad aquatic organisms and should remain a focus of research and conservation activities.

Few variables in our models explained variation in mallard and diving duck UD during fall. Certain wetland habitat types, particularly NPE, appeared to positively associate with historical mallard use, but even these variables explained only modest additional variation in UD. However, we suggest that the importance of IJI as a positive predictor of mallard UD during 1950–1959 indicated that composition and arrangement of habitats within wetlands was an important attractant to waterfowl. Most wetland restoration activities targeting migrating waterfowl in the midcontinent intend to only increase energetic carrying capacity (Soulliere et al. 2007). Once energetic goals are met (i.e., adequate forage per-unit-area), we suggest research and restoration efforts consider improving composition or arrangement of wetland habitats to increase waterfowl use.

Our study was observational and attempted to explain duck use over entire seasons. We recommend future research be conducted at finer spatial and temporal scales to better explain the relationships between duck use and wetland habitat characteristics. For example, predictive ability of models may improve if information on climate, duck abundance, vegetation structure, food availability, and disturbance were recorded daily or weekly. Such an approach may aid understanding of the numerical response of waterfowl to wetland habitats, but future research should also endeavor to understand functional responses (Holling

1959), perhaps by collecting data on behavior or physiological condition of waterfowl or through experimentation.

**Acknowledgments** Funding for this program was provided by a grant/cooperative agreement from the U.S. Department of the Interior, Fish and Wildlife Service (USFWS) through the Upper Mississippi River and Great Lakes Region Joint Venture (UMRGLRJV). M. Bowyer was instrumental in field work and conducted initial GIS work on historical maps during 2005 and P. White provided valuable field assistance during 2005–2006. Conversations with B. Pardo and G. Soulliere (UMRGLRJV), J. Coluccy (Ducks Unlimited, Inc.), and A. Pearce (U.S. Geological Survey) provided useful information that improved our research. G. Sass and J. Vest provided insightful comments on earlier drafts of this manuscript. Aerial inventories of waterfowl used in analyses were funded by project W-43-R of the Federal Aid in Wildlife Restoration Act (Pittman-Robertson), administered by the USFWS and IDNR. The Illinois Natural History Survey's (INHS) Geographic Information Systems (GIS) group provided considerable advice and technical assistance. Finally, we thank the IDNR site personnel and private landowners who provided information and allowed us access to their lands.

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