Ecological Applications, 20(7), 2010, pp. 2024–2035

Fire helps restore natural disturbance regime to benefit rare and endangered marsh birds endemic to the Colorado River

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Abstract. Large flood events were part of the historical disturbance regime within the lower basin of most large river systems around the world. Large flood events are now rare in the lower basins of most large river systems due to flood control structures. Endemic organisms that are adapted to this historical disturbance regime have become less abundant due to these dramatic changes in the hydrology and the resultant changes in vegetation structure. The Yuma Clapper Rail is a federally endangered bird that breeds in emergent marshes within the lower Colorado River basin in the southwestern United States and northwestern Mexico. We evaluated whether prescribed fire could be used as a surrogate disturbance event to help restore historical conditions for the benefit of Yuma Clapper Rails and four sympatric marsh-dependent birds. We conducted call-broadcast surveys for marsh birds within burned and unburned (control) plots both pre- and post-burn. Fire increased the numbers of Yuma Clapper Rails and Virginia Rails, and did not affect the numbers of Black Rails, Soras, and Least Bitterns. We found no evidence that detection probability of any of the five species differed between burn and control plots. Our results suggest that prescribed fire can be used to set back succession of emergent marshlands and help mimic the natural disturbance regime in the lower Colorado River basin. Hence, prescribed fire can be used to help increase Yuma Clapper Rail populations without adversely affecting sympatric species. Implementing a coordinated long-term fire management plan within marshes of the lower Colorado River may allow regulatory agencies to remove the Yuma Clapper Rail from the endangered species list.

Key words: Colorado River; disturbance; emergent marsh; endangered species; habitat quality; habitat restoration; marsh birds; marsh succession; marshlands; prescribed fire; rails; wetlands.

INTRODUCTION

Freshwater emergent wetlands are extremely productive ecosystems that support diverse and unique communities of plants and animals (Mitsch and Gosselink 2000). Wetlands also provide a wealth of ecosystem services to the surrounding landscape (Costanza et al. 1997, Wilson and Carpenter 1999, Zedler 2003, Hansson et al. 2005). Despite their value, emergent wetlands continue to be eradicated in the conterminous United States; emergent wetlands declined 21% between 1950 and 2004 (Dahl 2006). Loss of emergent wetlands and human-caused changes in the natural processes within wetlands are likely having adverse effects on the animals that depend on these unique ecosystems (Greenberg et al. 2006). Wetlands associated with the lower basins of major river systems have been particularly affected by anthropogenic changes. Annual pulse floods were once a significant disturbance event that occurred in many of the large river systems around the world, but most large rivers are now heavily regulated with numerous water control structures (Nilsson and Berggren 2000). Suppression of major disturbance events such as annual floods has been an important component of habitat loss in some ecosystems (Brawn et al. 2001).

The lower Colorado River basin of the southwestern United States and northwestern Mexico is a good example of a major wetland ecosystem that provides abundant ecosystem services but has experienced dramatic human-caused changes during the past century. The Colorado River drains parts of seven western U.S. states and northern Mexico, and provides water for >25 million people, >12 000 km^2 of irrigated farmland, and 11.5 billion kilowatt-hours of hydroelectric power (Good Neighbor Environmental Board 2003). Historically, flows were extremely variable with immense spring floods that distributed water across a broad floodplain supporting a variety of riparian, freshwater, brackish, and saltwater wetlands (Sykes 1937, U.S. Geological Survey 1954, Glenn et al. 1996, Nelson and Anderson 1999, Tiegs and Pohl 2005). These annual

Manuscript received 4 September 2009; revised 8 January 2010; accepted 11 January 2010. Corresponding Editor: R. Greenberg.

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spring floods were a major disturbance event that altered the vegetation and surface water within the floodplain of the lower Colorado River. For example, spring floods likely created and replenished marshes in backwaters and abandoned stream channels, many of which were far from the main river channel (Brown 1923, Nelson and Anderson 1999). These springtime pulse floods also set back the succession of emergent marsh vegetation in the lower river basin, prevented salt accumulation in the soil, scoured silt and decadent vegetation, and delivered water and seeds of emergent plants to the outer areas of the floodplain (Ohmart et al. 1988, Glenn et al. 2001, Stromberg 2001, Pataki et al. 2005). However, the hydrology of the lower Colorado River has been dramatically altered by a series of dams and flood control structures. Indeed, the Colorado River has more of its annual flow allocated to human use than all other large rivers and is among the most heavily regulated river in the world (U.S. Bureau of Reclamation 2000, Haddeland et al. 2006). The construction of the various water control structures on the lower Colorado River has dramatically changed the historical disturbance regime of the lower river basin. The seasonal hydrograph of the lower Colorado River is now flat, with greatly decreased water flow and almost complete elimination of the large spring floods that revitalized the ecosystem annually and set back plant succession within marshlands (Nelson and Anderson 1999, Good Neighbor Environmental Board 2003, Tiegs and Pohl 2005). Without these periodic pulse floods, the river channel has remained constant and its banks have been stabilized. The emergent marshes that occur on the river today are either within the main river channel or are in backwaters that are artificially supplied with a constant supply of river water by management agencies for mitigation purposes. These remaining marshes do not experience the disturbance events that historically occurred in the ecosystem and are hence succeeding toward later seral stages (Glenn et al. 2008).

The Yuma Clapper Rail (*Rallus longirostris yumanensis*) is a marsh-dependent bird endemic to the lower Colorado River. This subspecies may have initially benefitted when dam construction allowed the establishment of large areas of emergent marsh within the river corridor (Ohmart et al. 1975, Anderson and Ohmart 1985). However, the absence of floods or other mechanisms to reverse the senescence of these marshes is likely having adverse affects on the Yuma Clapper Rail. The Yuma Clapper Rail is federally endangered in the United States and federally threatened in Mexico (U.S. Fish and Wildlife Service 1983, Diario Oficial de la Federacion 2002). It is restricted to freshwater marshes in the lower Colorado River basin, and along the west coast of California and Baja California (Conway and Eddleman 2000, Hinojosa-Huerta et al. 2008). Populations of Yuma Clapper Rails are thought to be negatively affected by the accumulation of dead emergent vegetation in marshes that have not experienced a recent disturbance event (Conway et al. 1993, Conway and Eddleman 2000). Dense decadent vegetation may reduce productivity in the marsh by shading the substrate, by making nutrients unavailable, or by impeding rail movement.

To ensure long-term persistence of this species, we need to have effective management and mitigation activities that improve wetland quality to sustain viable populations. Prescribed fire has been repeatedly suggested as a surrogate disturbance event that may mimic the annual spring floods and help restore Yuma Clapper Rail populations (Conway et al. 1993, Eddleman and Conway 1998, Conway and Eddleman 2000) by removing decadent vegetation and encouraging growth of early-successional emergent vegetation. However, this potential management application has yet to be critically evaluated. Currently, some land management agencies are prevented from conducting prescribed fires within wetlands in the region because of concern that prescribed burns may kill endangered rails or destroy rail habitat. Hence, we need to evaluate the effects of fire on Yuma Clapper Rails to determine if prescribed fire can be used as a surrogate for the large springtime flood events that historically occurred.

The goal of this study was to examine the effects of prescribed fire on Yuma Clapper Rails. However, we were also concerned that prescribed fire, even if proven beneficial for Clapper Rails, may adversely affect sympatric avian species. For example, the California Black Rail (*Laterallus jamaicensis coturniculus*) breeds in many of the same marshes where Yuma Clapper Rails occur. California Black Rails are listed as federally endangered in Mexico (Diario Oficial de la Federacion 2002), threatened in California (California Department of Fish and Game 2006), and a species of special concern in Arizona (Arizona Game and Fish Department 1996). Populations have decreased in the southwestern United States over the past 30 years (Repking and Ohmart 1977, Evens et al. 1991, Conway and Sulzman 2007), and Black Rails are thought to be extremely sensitive to habitat disturbance (Eddleman et al. 1994). Consequently, California Black Rails are one of the highest priorities for conservation action in the region (Latta et al. 1999) and are considered a species of national conservation concern in the United States (U.S. Fish and Wildlife Service 2002). Some authors have suggested that fire be investigated as a possible management tool for restoring habitat for California Black Rails (Repking 1975), but others have cautioned that fire may completely eliminate Black Rail habitat from a marsh (Todd 1980). In addition to Black Rails, we were also concerned that prescribed fires might affect habitat quality for three other species of management concern in the region: Virginia Rails (*Rallus limicola*), Soras (*Porzana carolina*), and Least Bitterns (*Ixobrychus exilis*) (Tacha and Braun 1994, Gibbs et al. 1992, Conway 1995, 2008, Melvin and Gibbs 1996). Hence, we utilized a BACI

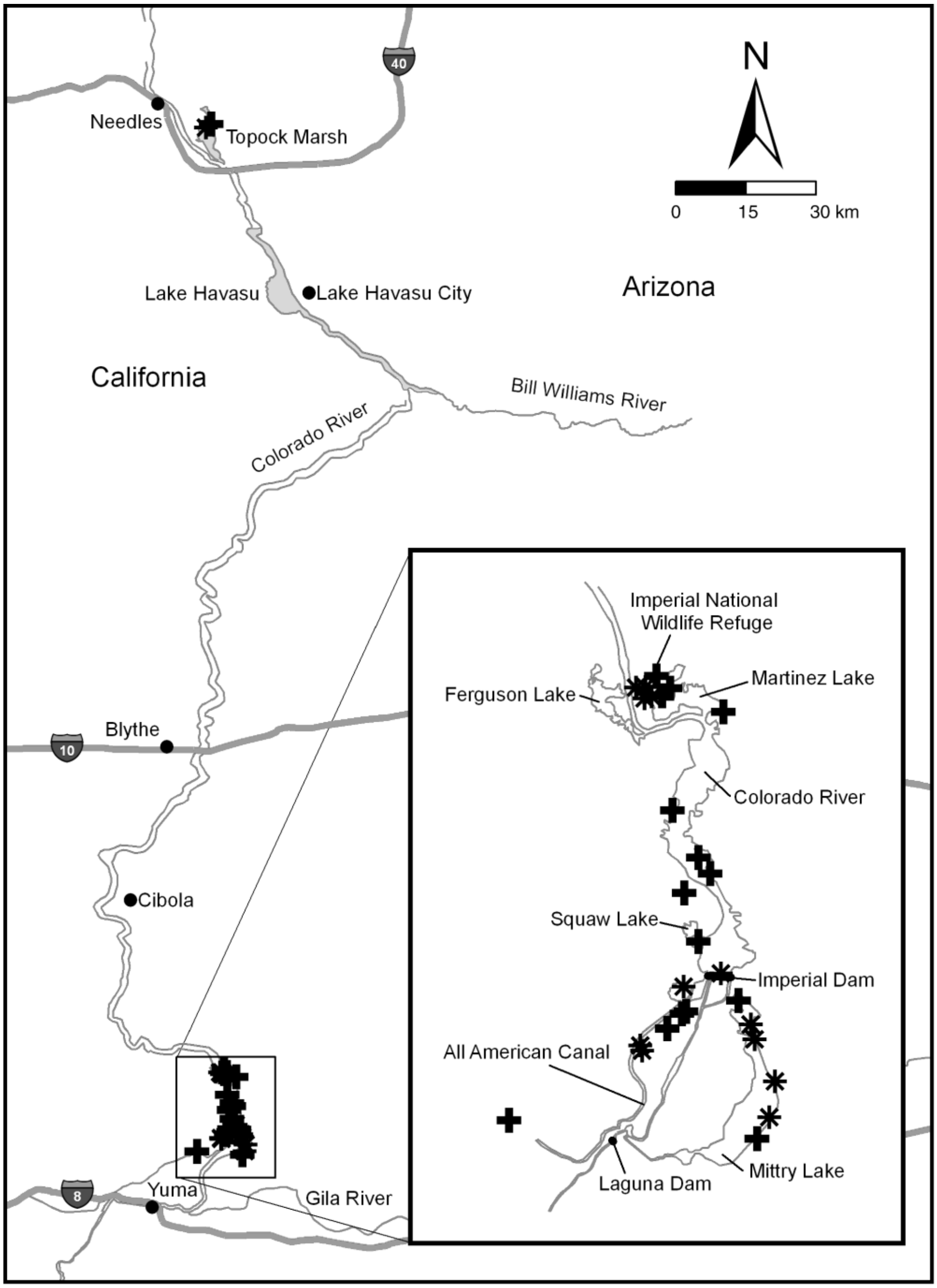

FIG. 1. Location of burn (stars) and control (crosses) plots in Arizona and California, USA.

(before-after-control-impact) experimental design to evaluate the effects of fire on these five sympatric marsh-dependent birds in the lower Colorado River basin of the southwestern United States.

METHODS

Study area

Study plots were in southwestern Arizona and southeastern California, USA, in marshes within the historical floodplain of the lower Colorado River (Fig. 1), primarily on lands managed by U.S. Fish and Wildlife Service (Imperial National Wildlife Refuge, Havasu National Wildlife Refuge), Arizona Game and Fish Department (Mittry Lake Wildlife Management Area), and U.S. Bureau of Reclamation (marshes above and below Imperial Dam). The study plots were distributed across much of the U.S. breeding range of the Yuma Clapper Rail. Mean annual precipitation varies from 8.1 cm to 15.7 cm throughout the study area.

Table 1. Years surveyed (indicated by an x) and year of fire (indicated by a vertical line) for each of 14 burns used to evaluate the effects of fire on marsh-dependent birds on the lower Colorado River in Arizona and California, USA.

			Years surveyed							
Plot	No. points†	Date burned	2000	2001	2002	2003	2004	2005	2006	2007
MWB	7	Mar 2001	x	│	x	x	x	x	x	
WPA	19	Mar 2002	x	x	│ x	x	x	x	x	x
SML	15	Apr 2002	x	x	│ x	x	x	x	x	x
NMLC	18	Feb 2003	x	x	x	│ x	x	x	x	x
IAF11	6	Feb 2003	x	x	x	│ x	x	x	x	x
NMLA	20	Mar 2003	x	x	x	│ x	x	x	x	x
NMLB	19	Mar 2003	x	x	x	│ x	x	x	x	x
IAF12	5	Mar 2004	x	x	x	x	│ x	x	x	x
HSB	18	Feb 2005	x			x	x	│ x	x	x
IDRMB	8	Feb 2005	x		x	x	x	│ x	x	x
IDRMC	9	Feb 2005	x		x	x	x	│ x	x	x
IAF13	8	Feb 2005				x	x	│ x	x	x
IAF14	11	Feb 2006	x	x	x	x	x	x	│ x	x
WS	6	Mar 2006		x	x	x	x	x	│ x	x

† "No. points" indicates the number of point-count survey stations included within each plot.

Southern cattail (*Typha domingensis*) was the most common emergent plant in our study plots, but California bulrush (*Schoenoplectus californicus*), chairmaker's bulrush (*Schoenoplectus americanus*), common reed (*Phragmites australis*), and other less common wetland plants were present in many study plots (see Conway and Sulzman 2007).

Field methods

We compared numbers of marsh birds detected before and after fire at 14 burn plots (8 that were burned as prescribed fires and 6 that burned as a result of incidental fires) and also at 15 control (non-burned) plots. We selected 1–2 control plots adjacent to each burn to account for any spatial variation in abundance. Fire management crews from the U.S. Fish and Wildlife Service and U.S. Bureau of Land Management conducted each of the prescribed fires. All fires occurred during the late winter or early spring (February–April) of 2001 to 2006 (1–4 burns/yr; Table 1). Study plots varied in size from 4 ha to 113 ha based on logistical constraints associated with each burn and the size of existing marshes. We did not monitor the fate of individual birds in response to the fire treatments, but all prescribed burns occurred prior to the peak breeding season, and rails were seen running or flying out of burn plots during several fires. Hence, we believe that resident marsh birds were only temporarily displaced (rather than killed) by the fires.

We conducted point-count surveys (with both passive and call-broadcast segments) during the breeding season (15 March–15 June) in both burn and control plots for 1–6 years ($\bar{x} = 2.6$ yr) pre-burn and for 2–5 years ($\bar{x} = 3.9$ yr) post-burn (Table 1). Hence, we had both temporal (pre- vs. post-burn) and spatial (burned vs. unburned) controls built into our sampling design. The number of years of survey data for pre- and post-burn varied among plots because we were only able to conduct 1–2 prescribed burns in most years due to the significant planning, permitting, and logistical effort required to conduct a large experimental fire in areas occupied by an endangered species. However, we believe that this aspect of the design was beneficial because we were interested in evaluating how abundance of birds changed as the number of years post-burn increased independent of any systematic trend in bird abundance across years. In other words, the staggered entry of new burns each year allowed us to evaluate whether birds increased in response to fire even if bird abundance was increasing or decreasing across time independent of our treatments. Time since burn was recorded in half-years because all burns occurred in the winter and all surveys occurred during the rail's breeding season (March–June). Hence, our first round of post-burn surveys at each plot occurred 1–3 months after the fire.

We established permanent survey points along the periphery of each plot at either 50- or 100-m intervals. We conducted 1–6 point-count surveys ($\bar{x} = 3.0$ surveys) per year from March to June at each study plot. We used existing survey protocols for wetland birds (Conway et al. 2002, Conway 2008). Each point-count survey included an initial passive detection period of 3–5 min followed by a period of call-broadcast of 3–6 minutes. The call-broadcast period consisted of a series of 1-min segments, each of which included 30 s of pre-recorded calls of the marsh birds thought to occur in that wetland based on prior surveys throughout the region (Conway et al. 2002, Conway and Sulzman 2007). Hence, the duration of the point-count surveys differed among plots, but was consistent throughout the duration of the study within a plot. We broadcast calls at 80–90 decibels (measured 1m from the speaker), and we used recordings of the common breeding calls of each species for the call-broadcast following Conway (2008). As per national protocols (Conway 2008), we conducted all surveys either between dawn and 09:00 hours or between 17:00 hours and dusk. Detection probability of our focal species did not differ between these two daily time intervals (Conway et al. 2004, Nadeau et al. 2008). We conducted surveys on days without rain and when wind speeds did not exceed 10 km/hr. We excluded birds detected at each point that we thought may have been detected at a previous survey point. We averaged the number of birds detected on a plot across the repeated surveys at that plot within a year, and compared the annual average during post-burn years to that of pre-burn years at each plot.

All surveyors attended two weeks of extensive field training prior to each survey season, and each surveyor took a hearing test (audiogram) to check their ability to hear subtle marsh bird vocalizations. We conducted periodic multiple-observer surveys (Nichols et al. 2000, Conway et al. 2004, Nadeau et al. 2008) each year to ensure that certain surveyors were not missing subtle calls. Fire alters vegetation structure and vegetation structure can affect detection probability of birds during surveys (Kubel and Yahner 2007, Pacifici et al. 2008). Hence, we wanted to examine whether detection probability during our surveys differed among treatments (control, pre-burn, and post-burn). As per national protocols (Conway 2008), all surveyors recorded whether they detected each bird within each 1-min interval of the survey. We used these data to examine whether detection probability differed among control, pre-burn, and post-burn plots using time-of-detection models (Alldredge et al. 2007).

To assess the effects of fire on plant species composition, we conducted 6 line transect surveys on each of 2 of our burn plots and resampled these transects each year for 6 years (2 years pre-burn and 4 years post-burn). The start of each transect was on the edge of the burn and the length of transects ranged from 45 m to 263 m ($\bar{x} = 157.2$ m). We laid out a rope along each transect during June of each year and recorded the plant species present within each 1-m interval for the entire length of each transect. We then calculated the percent of the total length of the transect occupied by each plant species as the number of 1-m segments in which a plant species was present on the transect divided by the number of 1-m segments on that transect. If more than one species was present in a 1-m segment, each species was attributed an equal portion of the segment. Dividing a segment where more than one species was present made it so that the percentages all sum to 100%, reduces confusion, and more appropriately represents species composition (because it doesn't under represent those plant species that are more commonly monotypic). Along these 12 line-transects, we also recorded a qualitative index of standing dead (decadent) vegetation at each of the 1-m intervals (with 0 indicating little or no decadent vegetation and 1 indicating moderate to substantial decadent vegetation).

Statistical analysis

We used the plot as the sampling unit in all of our analyses. We used linear mixed-model analyses to examine the effects of fire on abundance of each focal species. The response variable for each analysis was the average number of birds detected during year t at plot y minus the average number of birds detected during pre-burn years at plot y. For control plots, we used the year that the fire occurred in the adjacent burn plot as the burn year. We used a diagonal covariance structure with plot as a subject variable (random effect), treatment (burn or control) as a fixed effect, and the number of years post-burn as a repeated (fixed) effect in each model. We also included the interaction between treatment and years post-burn in all models because we were interested in whether fire effects might change as time since fire increased. This approach allowed us to test the hypothesis that bird abundance increased in response to fire by examining whether the number of birds detected was higher in post-burn years (relative to pre-burn years) on burned plots but not on control plots.

We used the *full closed-captures with heterogeneity* model structure in Program MARK (White and Burnham 1999) to examine whether detection probability of each of the five focal species differed among control, pre-burn, and post-burn plots. We evaluated 16 different models by allowing initial detection probability and probability of re-detection to vary by both time (across the 1-min intervals of the survey period) and treatment (control, pre-burn, and post-burn) (Farnsworth et al. 2005). We ranked candidate models by ascending Akaike's Information Criterion (AIC_c) values, adjusting for the small-sample bias (Burnham and Anderson 2002).

RESULTS

Burn severity and the effects on vegetation were all very similar among our burned study plots. Burning removed most of the live and dead plant material above the surface of the water or substrate. Wet, matted vegetative material at, and below, the waterline was not eliminated, charred bases of emergent plants often remained, and rhizomes appeared to be unaffected. Hence, marsh vegetation grew back rather quickly after the fires. In most burns, marsh vegetation was tall enough to attract rails within 1–3 months postfire. Vegetation height was typically similar to pre-burn levels within 1 year postfire, but was noticeably greener and lacked the dense understory of decadent vegetation that was present prefire. By year two, marsh vegetation in burned plots was difficult to distinguish from control plots (at least from a distance), but still had less dead vegetation in the understory.

We detected more Clapper Rails during post-burn years compared to pre-burn years on burn plots but not on control plots (Table 2, Fig. 2a). We saw some evidence that the positive effects of fire began to diminish as time since fire increased (interaction term in Table 2, Fig. 2a), even though our sample size declined as years post-burn increased. We also detected more Virginia Rails during post-burn years within burn

TABLE 2. Results of five linear mixed-model analyses examining the effects of fire on abundance of five species of marsh-dependent birds within the lower Colorado River basin of the southwestern United States.

Variable	Clapper Rails *F*	Clapper Rails *P*	Black Rails *F*	Black Rails *P*	Virginia Rails *F*	Virginia Rails *P*	Soras *F*	Soras *P*	Least Bitterns *F*	Least Bitterns *P*
Intercept	**11.5**	**0.001**	**10.4**	**0.002**	2.1	0.154	0.0	0.986	**4.2**	**0.045**
Treatment (burn or control)	**19.2**	**<0.001**	0.0	0.952	**6.0**	**0.017**	0.1	0.793	0.1	0.790
No. years post-burn	1.0	0.327	0.1	0.805	0.1	0.747	0.2	0.702	0.3	0.569
Treatment × no. years post-burn	3.8	0.075	0.9	0.334	1.5	0.228	0.8	0.391	0.5	0.493

Notes: The response variable was the increase in number of birds detected in post-burn years relative to the average of pre-burn years. Treatment (burn or control) was a fixed effect, plot was a random effect, and number of years post-burn was a repeated (fixed) effect in the models. Analyses were based on 29 plots (14 burned, 15 control), and the number of years sampled varied among plots from 1–6 years pre-burn and 2–5 years post-burn. We have included the sample sizes used in the analyses because they are more informative than the degrees-of-freedom values in linear mixed models (LMM). Boldface type highlights comparisons that were statistically different.

plots but not on control plots (Table 2, Fig. 2c), but we failed to detect an effect of fire on abundance of the other three focal species (Table 2, Fig. 2). We found support for models where both initial detection probability and probability of re-detection varied among the 1-min segments of the survey (Table 3). However, we found no evidence that detection probability differed between burn and control plots for any of the five species (Table 3). The species composition of the vegetation did not change noticeably as a result of the burns; most plots were dominated by southern cattail (or cattail and common reed) both before and after fire (Fig. 3). The amount of decadent vegetation was reduced as the result of the fires (Fig. 4).

DISCUSSION

Numbers of Clapper Rails and Virginia Rails increased following fire, and numbers of the other three focal species were unaffected. Emergent plants (cattail and bulrush) grew back quickly following the burns, an observation echoed by previous studies (Smith and Kadlec 1985*a*). We found no evidence that detection probability of any of the five bird species changed in response to fire. As expected, detection probability differed among the 1-minute segments of our surveys because call-broadcast is known to increase detection probability for all of our focal species (Conway and Gibbs 2001, 2005, Conway and Nadeau 2010). We detected more Least Bitterns in post-burn years on both burn and control plots. This pattern may be due to an increase in the ability of our field personnel to detect the more subtle calls of least bitterns as the study progressed. An increase in observer detection probability over time is a common bias in bird monitoring programs (Sauer et al. 1994, 1995, Kendall et al. 1996, Norvell et al. 2003) and can lead to biased conclusions in studies that include temporal, but not spatial, controls. We included both temporal and spatial controls and hence were able to detect, and account for, possible increases in observer ability to detect least bitterns.

Numbers of Clapper Rails and Virginia Rails may have increased in response to fire for one or more of the following reasons. First, dense vegetation at the water or mud surface may inhibit rail movement in undisturbed wetlands. Dense litter and decadent vegetation may reduce foraging efficiency of these birds by hindering movement. Indeed, fires reduced the amount of decadent vegetation in our study. Second, these birds seldom fly. They primarily walk to move from nest to foraging areas and between foraging areas. Marsh birds that occupy vegetation that has not been disturbed recently may be forced to walk on top of the dense mat of decadent vegetation (C. Conway, *personal observation*) to move from one location to another. This activity may expose birds to higher predation risk. And lastly, dense understory vegetation within these marshlands may reduce invertebrate food availability. Future research is needed to determine which of these (or other) mechanisms explains the increase in numbers of marsh birds postfire.

Natural disturbance agents are thought to have played a key role in the evolution and maintenance of biodiversity (Askins 2000). Indeed, allowing natural disturbance events (e.g., wildfire or floods) to proceed without intervention is being incorporated into many regional conservation plans (Johnson et al. 1998). However, our understanding of the importance of natural disturbance on vertebrates is limited due to our inability to experimentally mimic the full suite (and scale) of effects for many types of disturbance. Past studies evaluating the effects of disturbance have generally been of short duration, lacked appropriate controls, had few or no replicates, and were incidental (unplanned) (Finch et al. 1997). For example, we have only a limited understanding of the role of natural disturbance agents on the ecology of birds, and yet effective conservation strategies likely require the management of disturbance events to maintain biodiversity in many ecosystems (Brawn et al. 2001). Cyclical flooding events associated with large river systems are recognized as one of the most important sources of ecosystem disturbance, and floodplains of major rivers are among the most disturbance-prone of all ecosystems (Brawn et al. 2001). Many of the marsh-dependent organisms endemic to the lower reaches of large rivers may be adapted to the historical disturbance regime.

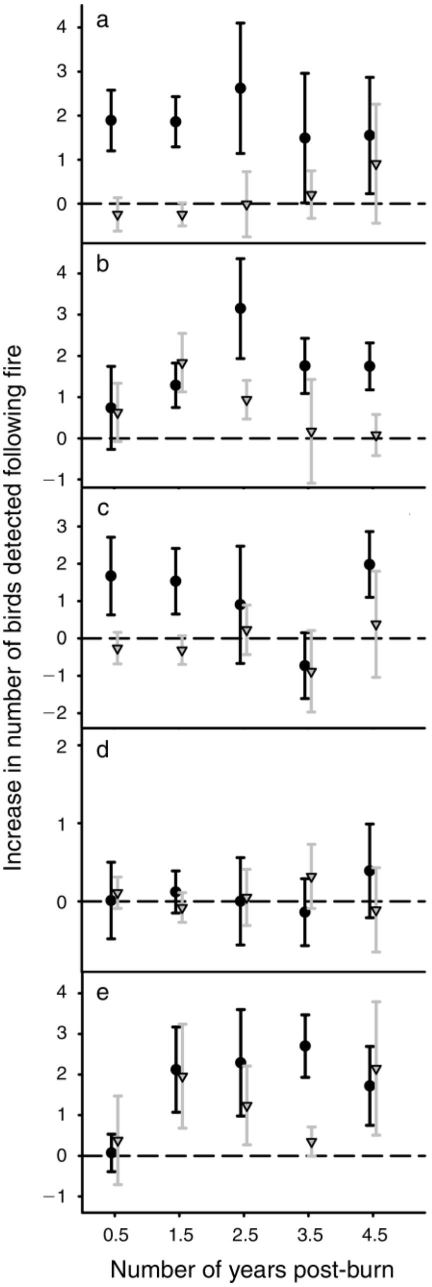

FIG. 2. Change in number of birds detected during post-burn years relative to the average number detected during pre-burn years on burn (circles) and control (triangles) plots for each of five focal species of marsh-dependent birds in the lower Colorado River basin of the southwestern United States: (a) Yuma Clapper Rails, (b) California Black Rails, (c) Virginia Rails, (d) Soras, and (e) Least Bitterns. The *y*-axis is the number of birds detected during each year post-burn minus the average number of birds detected across all years pre-burn. For control plots, we used the year that the adjacent treatment plot was burned as the "burn" year. Errors bars are standard errors based on averages across plots within a treatment; sample sizes from left to right are: 13, 13, 14, 14, 12, 11, 8, 5, 7, 5. Sample sizes decrease (and hence error bars tend to increase) from left to right due to the staggered entry of the burns (we only conducted 1–2 prescribed burns per year).

Ensuring persistence of the organisms that are well adapted to these periodic disturbance events has proven challenging to land managers and policy makers. We often have a limited ability to manage successional processes within an entire ecosystem through the application of disturbance (Brawn et al. 2001). For example, large-scale removal of water control structures and restoration of the historic flood regime is unlikely to occur in most large river systems around the world because the historic floodplain has often been developed for human uses and the nearby cities typically rely on stored or diverted water from the river. Consequently, floodplain habitats are among the most endangered ecosystems (Dynesius and Nilsson 1994). Damming of rivers in the western United States has greatly reduced the severity and frequency of flood disturbance, which has caused many bird species that depend on these systems to decline (Anderson et al. 1983, Rice et al. 1984, Strong and Bock 1990). Our results suggest that prescribed fire can be used to set back plant succession within emergent freshwater marshlands and help mimic the natural disturbance regime on the lower Colorado River. Hence, prescribed fire can be used periodically to create early-successional emergent vegetation in the

TABLE 3. List of competing models designed to evaluate whether initial detection probability (*p*) and probability of re-detection (*c*) of five marsh birds varied by either time (*t*; 1-minute segments within survey) or treatment group (*g*; control, pre-burn, or post-burn).

Species and model	ΔAIC_c	AIC_c weight	Model likelihood
Yuma Clapper Rail			
pi(*g*)*p*(*t*)*c*(*t*)	0.00	0.976	1.00
pi(.)*p*(*g* × *t*)*c*(*t*)	8.65	0.013	0.01
California Black Rail			
pi(.)*p*(*t*)*c*(*t*)	0.00	0.600	1.00
pi(*g*)*p*(*t*)*c*(*t*)	1.99	0.221	0.37
pi(*g*)*p*(*t*)*c*(*g* × *t*)	2.47	0.174	0.29
pi(.)*p*(*g* × *t*)*c*(*t*)	3.37	0.079	0.18
Virginia Rail			
pi(*g*)*p*(*t*)*c*(*t*)	0.00	0.427	1.00
pi(*g*)*p*(*t*)*c*(*g* × *t*)	1.74	0.179	0.42
pi(.)*p*(*t*)*c*(*t*)	1.96	0.160	0.37
pi(.)*p*(*g* × *t*)*c*(*t*)	3.37	0.079	0.18
pi(*g*)*p*(*g* × *t*)*c*(*t*)	3.37	0.079	0.18
pi(.)*p*(*t*)*c*(*g* × *t*)	3.78	0.064	0.15
pi(*g*)*p*(*g* × *t*)*c*(*g* × *t*)	7.20	0.012	0.03
Sora			
pi(*g*)*p*(*t*)*c*(*g*)	0.00	0.646	1.00
pi(*g*)*p*(*t*)*c*(.)	1.45	0.312	0.48
pi(*g*)*p*(*t*)*c*(*t*)	6.97	0.020	0.03
pi(*g*)*p*(*g* × *t*)*c*(.)	9.06	0.007	0.01
Least Bittern			
pi(*g*)*p*(*t*)*c*(*t*)	0.00	0.968	1.00
pi(.)*p*(*t*)*c*(*t*)	6.90	0.031	0.03

Notes: We used the full closed-captures heterogeneity (pi) model structure in Program MARK (White and Burnham 1999) and compared AIC_c for 32 candidate models per species (Burnham and Anderson 2002). Only models with $\Delta AIC_c < 10$ are listed.

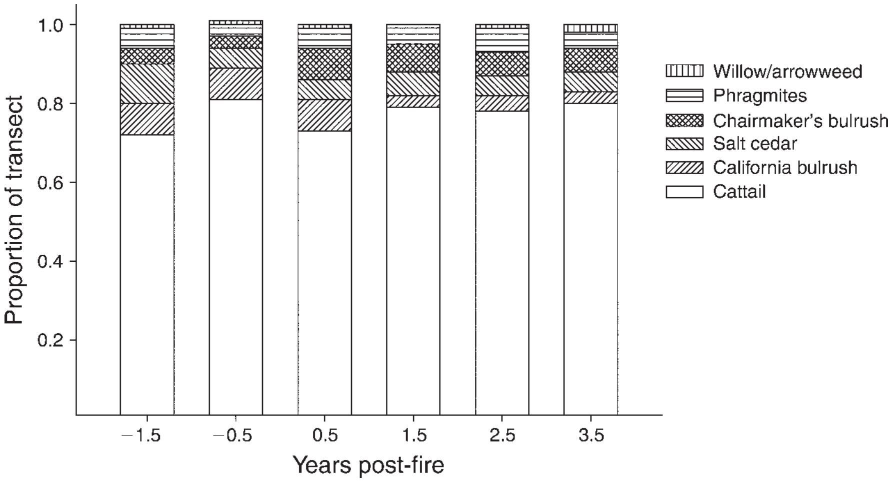

FIG. 3. Plant species composition at two burn plots for each of six consecutive years (two years pre-burn and four years post-burn).

absence of spring floods to help increase populations of endemic marsh-dependent birds.

Our results suggest that prescribed fire shows promise as a surrogate disturbance agent that might be used to increase marsh bird populations on the lower Colorado River, but additional studies are needed to evaluate the many other effects that repeated use of fire might have on the entire ecosystem. Little information is available regarding the historical fire regime within marshlands along the lower Colorado River or any similar desert riparian system in the region (Turner 1974). Natural fires were probably not common, despite the fact that emergent wetland plants accumulate litter rapidly and this decadent material is extremely flammable (U.S. Forest Service 2009). Most natural fires likely start via lightning strikes during the late summer monsoon season (July–September), and the incidence of lightning strikes on the lower Colorado River is relatively low (0.5–1.0 cloud-to-ground flashes$\cdot$km$^{-2}\cdot$yr^{-1}) compared to other areas in the United States (Vaisala's National Lightning Detection Network 2009). Moreover, lightning strikes have to ignite vegetation within the river corridor in order to create fire because the surrounding desert vegetation is too sparse to carry a fire. However, incidental human-caused fires are rather common in these marshes; >90% of the fires within the lower Colorado River corridor over the past five years were human-caused (U.S. Bureau of Land Management, *unpublished data*).

Introduction of prescribed fire as a management tool would not restore all of the disturbance processes produced by the large, annual spring floods that occurred historically on the lower Colorado River. Those floods likely redistributed sediment and nutrients, altered the course of the main river channel and modified the river banks, created oxbow lakes and backwaters, revitalized backwater depressions that were previously dry, promoted successional gradients in marshland plant communities, dispersed seeds long distances down the river corridor, and removed all decadent vegetation and scoured the substrate within some (but not all) marshes each year (Junk et al. 1989, Malanson 1993, Johnson 1994). In contrast, burning merely removes the aboveground plant material in the target marsh. Historic floods likely set succession back even further than these fires did. Moreover, historic spring floods and winter-season fires likely differ in numerous other ways including their effects on nutrient cycling and other species of wildlife. Future studies should examine the differences in how these two disturbance agents affect these other response variables. Finally, unless fires occur very regularly, decadent vegetation may continue to accumulate in a marsh, which may cause the marsh to eventually progress to a later successional stage. Fires will have to be frequent

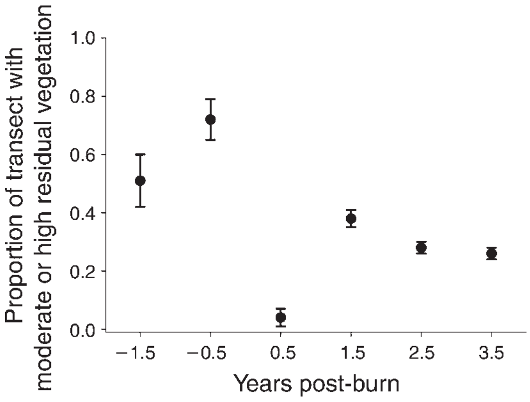

FIG. 4. The amount of residual vegetation decreased following fire. The *y*-axis is the percentage of 12 line-transects (six within each of two of our burn plots) that contained moderate to high amounts of residual vegetation during each of six consecutive years (two years pre-burn and four years post-burn).

enough to prevent succession and maintain optimal rail habitat over the long term. Hence, we need studies designed to examine the potential long-term impacts of repeated burning on plant community composition in the absence of the soil scouring and siltation services provided by spring floods.

Prescribed fire may not be beneficial to other marsh-dependent birds and may not be beneficial to the five species examined here in other wetland ecosystems. Prescribed fire is a common management practice in marshes in some areas of the United States, primarily on the gulf and southeast Atlantic coasts (Davidson 1992, Eddleman et al. 1994, Mitchell et al. 2006). Fire has been shown to increase plant diversity in salt marshes (de la Cruz and Hackney 1980, Ford and Grace 1998, Madanes et al. 2007), but to have no (or little) effect on plant diversity, salinity, or pH in other freshwater marshes (van der Toorn and Mook 1982, Smith and Kadlec 1985*b*, Janousek and Olson 1994, Taylor et al. 1994, Kostecke et al. 2004). We also detected no change in plant species composition following fire. Despite the common use of prescribed fire, very few studies have examined the effects of these fires on marsh birds using a rigorous BACI experimental design like we have here (Kirby et al. 1988, Walters et al. 2000, Mitchell et al. 2006). The few experimental studies that do exist only examined the effects of fire on saltmarsh sparrows (not rails or bitterns) and only for 1–2 years post-burn; relative densities of wintering sparrows returned to pre-burn levels one year post-burn and relative densities of breeding sparrows were higher in burned marshes compared to unburned marshes during the second breeding season post-burn (Gabrey et al. 1999, Gabrey and Afton 2000). Periodic but infrequent fires are thought to benefit breeding and wintering sparrows in Louisiana (Gabrey et al. 2001). Fire has traditionally been thought to benefit Cape Sable Seaside Sparrows (*Ammodramus maritimus mirabilis*) within subtropical wet grasslands in southern Florida (Taylor 1983), but recent correlative studies suggest that frequent fires are harmful to these birds (Curnutt et al. 1998, Walters et al. 2000, La Puma et al. 2007). Fires in saltmarshes have been shown to decrease abundance of numerous species of marsh-dependent birds in Argentina, including Dot-winged Crakes (*Porzana spiloptera*) and Speckled Crakes (*Coturnicops notata*) (Isacch et al. 2004). However, none of these past studies accounted for differences in detection probability between treatments as we did. In summary, fires have less obvious and more ambiguous effects on marsh birds within other wetland ecosystems. Hence, the utility of fire as a management application to replicate the historical disturbance regime on large river systems holds promise for restoring some populations of endemic species that are adapted to such disturbances, but needs to be evaluated experimentally in other river (and estuarine) systems. And even within the lower Colorado River ecosystem, many important questions remain. For example, we need studies designed to examine the effects of fire severity, fire size (i.e., the spatial scale at which fire affects marsh bird populations), seasonality of fires, different dispersion (spatial arrangement) of burned areas, and different disturbance agents (fire, mechanical, flood). Most importantly, we need information on the long-term effects of repeated fires on plant successional processes and the optimal disturbance interval (i.e., prescribed fire frequency) as these are extremely important attributes in all disturbance-prone ecosystems (Brawn et al. 2001).

Management Recommendations

Yuma Clapper Rails, California Black Rails, and Least Bitterns are considered priority species of concern in the Lower Colorado River Multi-Species Conservation Plan (LCR MSCP; U.S. Bureau of Reclamation 2006). The goals of the LCR MSCP for each of these species includes enhancing existing habitat, and restoring or creating habitat to provide additional breeding locations within the lower Colorado River basin. Our results suggest that prescribed fire can be used as an on-the-ground management tool to achieve several LCR MSCP goals. The results of this project may also aid efforts to de-list the Yuma Clapper Rail from the federal endangered species list. One of the requirements in the Yuma Clapper Rail recovery plan (U.S. Fish and Wildlife Service 1983) is to develop a proven method for habitat restoration; this project demonstrates that prescribed fire may serve that purpose. We suggest that regulatory agencies consider allowing periodic prescribed fires within marshes in the lower Colorado River basin, especially in marshes where Clapper Rail density has remained well below the 10-year average for several years and emergent vegetation appears decadent. Fires should probably be restricted to a small percentage (e.g. $\leq$15%) of the available local habitat for each species in any one year so that resident birds have suitable areas to go while vegetation recovers from the burns. Historically, the frequency with which succession of emergent vegetation was set back by spring floods likely varied greatly among marshes depending on their proximity to the main river channel and the seasonal dynamics of snow melt and runoff in the upper river basin. Although periodic fire appears to benefit (or not affect) marsh-dependent birds in the lower Colorado River basin, we still need additional information on the optimal frequency with which fire should be reapplied. Our results suggest that optimal burn frequency may vary among coexisting species. Hence, the frequency with which prescribed fire should be reapplied to individual marshes requires further study before we can recommend a detailed long-term fire plan that is optimal for all species endemic to the lower river basin. Moreover, we recommend initiating studies in concert with any long-term fire plan that would quantify the potential long-term effects of repeated fires on nutrient cycling in these marshes as part of a region-wide adaptive management plan.

Acknowledgments

The Fire Management Teams from the U.S. Fish and Wildlife Service and the U.S. Bureau of Land Management conducted all of the experimental prescribed burns. B. Raulston and J. Swett provided support for the first year of the project and were instrumental in getting the project started. Imperial and Havasu National Wildlife Refuges provided field housing, field vehicles, and boats. G. Finnegan of U.S. Bureau of Reclamation provided us with a boat each year. M. Kaib provided logistical support. Many technicians helped conduct surveys over the seven years, especially C. Sulzman, M. Vamstad, N. Bartok, J. Mohlmann, and M. Conway. C. Yde provided superb administrative support throughout the project.

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