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DEMOGRAPHICS OF SMALL MAMMALS USING ANTHROPOGENIC DESERT RIPARIAN HABITAT IN ARIZONA

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Abstract: Resource managers attempting to rehabilitate degraded desert riverine ecosystems must understand the effects of vegetation management on riparian wildlife. I used capture-recapture methods in February, June, and September 1992, and February 1993 to investigate demography of small mammals at a 4-ha site on the xerified lower Colorado River floodplain, 5 years after treatment to replace saltcedar (*Tamarix ramosissima*) with native woody plants. The site had become a mosaic of various vegetation types, including closed-canopy cottonwood/willow (*Populus fremontii*/*Salix nigra*), with no natural counterpart. I captured 9 of 15 native small mammal species potentially resident in local riparian habitats. Abundances of 7 of these showed a net increase over the year, although the rise was significant ($P < 0.05$) only for the deer mouse (*Peromyscus maniculatus*). Models incorporating estimates of recruitment via reproduction and survival were adequate to explain observed population dynamics, supporting the hypothesis that the site represents source habitat for most species. However, the site may have been a population sink for the deer mouse. Although not statistically analyzed, community biomass rose from 3,200 to 4,700 g/ha, reflecting contributions from the white-throated woodrat (*Neotoma albigula*), cactus mouse (*P. eremicus*), and Arizona cotton rat (*Sigmodon arizonae*). High biomass suggests the site has the potential to be important in local ecosystem functioning.

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Key words: Arizona, *Chaetodipus*, Colorado River, demography, desert, habitat restoration, *Neotoma*, *Peromyscus*, regulated river, riparian, rodents, saltcedar, *Sigmodon*, *Tamarix ramosissima*.

Riparian areas are hypothesized to be high quality habitats for many vertebrates (Johnson and Jones 1977, Brown and Johnson 1989), with the potential to serve as source areas in local or

regional population dynamics (Doyle 1990). High survival and/or production of juveniles among resident vertebrates should be especially pronounced in warm desert riparian sites, where the mix of high air temperature and soil moisture would promote high rates of primary and secondary production in a landscape generally noted for its low productivity.

The lower Colorado River supported some

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160,000 ha of riparian vegetation in 1894 (Mearns 1907 in Ohmart et al. 1988:24). However, a major contributor to this system's productivity has been eliminated: the annual flooding of the lowermost terrace of the floodplain (the "first bottom") that occurred during May and June as snowmelt progressed in the Rocky Mountains (Grinnell 1914). Dramatic shifts in composition and declines in the amount of natural riparian vegetation have occurred along this and other rivers of the North American desert Southwest since the early 1900s, reflecting interrelated changes in regional hydrology, water quality, wildfire frequency and intensity, land use practices, and invasion by an exotic plant, saltcedar (Ohmart et al. 1988). Federal and state agencies recently initiated programs to conserve and enhance wildlife resources along the lower Colorado River by replacing stands of saltcedar with native woody riparian species (Rosenberg et al. 1991, Pinkney 1992). Few evaluations of the kinds and numbers of wildlife within these areas of quasi-natural vegetation have been undertaken.

I collected data for 1 year on abundance and dynamics of small mammals using a 5-year-old revegetated riparian site on the lower Colorado River. I determined which species among the region's riparian small mammals used the revegetated site, and estimated values for survival and reproduction to test the hypothesis that this anthropogenic riparian system constituted a source habitat. I used a modeling rather than comparative approach to test my hypothesis; there was no pristine riparian habitat extant on the lower Colorado River, and I made no simultaneous evaluation of populations on unmanipulated habitat. I presume that any biophysical characteristics of the site leading to its operating as a population sink (Van Horne 1983, Pulliam and Danielson 1991) would not be completely masked even if conditions were unusually favorable during the study.

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STUDY AREA

I worked within a portion of the No Name Lake revegetation site, a 17-ha mitigation project located in the Colorado River Indian Reservation on the Arizona side of the Colorado River, 30 km south of Parker, Arizona. The area was cleared of saltcedar and most other vegetation by a contractor and replanted with native riparian trees and shrubs in 1987. Only a few large existing screwbean mesquite (*Prosopis pubescens*) and western black (Goodding) willow were retained. Fremont cottonwood and honey mesquite (*P. juliflora* var. *glandulosa*) made up 80% of the 6,100 transplants, with the remainder largely western black willow, blue palo verde (*Cercidium floridum*), quailbrush (*Atriplex lentiformis*), and California fan palm (*Washingtonia filifera*). Plant nomenclature follows Benson and Darow (1981). The species mix varied among areas within the site, but all transplants were provided with water through the 1988 growing season via drip irrigation systems.

Although survival of honey mesquite and palo verde was >75% in 1990 (Pinkney 1992), all palms and most of the 1,800 cottonwood and willow transplants died. Surviving cottonwood and willow trees, many approaching 12 m in height, formed a closed-canopy stand in a 0.5-ha depression that was inundated when adjacent cropland was irrigated. The understory within the grove varied from litter-covered open ground to dense quailbrush or arrowweed (*Pluchea sericea*) in 1992; there were few grasses or forbs, and essentially no large woody debris on the ground. I studied this cottonwood/willow grove and an adjacent xeric area in which most mesquite transplants survived because of the range of vegetation types represented. No Name Lake, a former Colorado River backwater that was an emergent marsh in 1992, formed the western border of the study site. The lake and study site were physically separated from the river by a dike.

The Colorado River floodplain surrounding the site (known as Parker Valley) was about 20 km wide; the nearest rocky outcrops or upland desert were about 8 km away, on the opposite side of the river. The climate was arid (1951–80 mean annual precipitation at Parker was 10.4

Table 1. Number of unique individuals captured (M_{t+1}), population estimates (\hat{N}), and associated confidence intervals (CI) for the most abundant small mammals at the No Name Lake revegetation site, Arizona, 1992–93.

Species	Session ^a	M_{t+1}	\hat{N}^b	95% CI	Model ^c
Cactus mouse	1	56	142AB	114–215	M(bh)
	2	89	139A	117–176	M(0)
	3	108	187AB	159–231	M(h)
	4	147	215B	189–256	M(h)
Deer mouse	1	25	31A	27–47	M(h)
	2	4	4		
	3	0	0		
	4	55	58B	55–72	M(h)
White-throated woodrat	1	25	31A	27–48	M(h)
	2	56	62B	57–85	M(h)
	3	51	58B	54–71	M(h)
	4	38	44AB	39–61	M(h)
Desert pocket mouse	1	4	4		
	2	17	17		
	3	34	46	39–67	M(h)
	4	12	12		

^a Trapping sessions 1–4 were 5–11 Feb, 2–8 Jun, and 25–29 Sep 1992 and 18–23 Feb 1993, respectively.

^b Estimates for species with same letter are not different ($P > 0.05$).

^c Model selected by program CAPTURE (Otis et al. 1978).

cm), with a bimodal rainfall pattern featuring peaks in August and January.

METHODS

Expected Species

I identified terrestrial small mammals that historically occupied riparian zone vegetation along the Arizona side of the Colorado River in the vicinity of the No Name Lake revegetation site, using Hoffmeister (1986:28). I included all species Hoffmeister associated with “Sonoran-Lower Colorado Desertscrub” and “Perennial Streams” vegetational communities, and then edited the list to conform to his range maps.

Trapping Protocol

I set up a grid of 201 trap stations (12 × 17, with 3 adjacent corner stations missing) during 10–16 January 1992. Stations were 15 m apart, resulting in the grid enclosing 3.89 ha. The grid contained almost the entire cottonwood/willow grove, as well as about 3 ha originally planted with cottonwood and mesquite that was a mosaic of dense arrowweed, quailbrush, or sparse arrowweed growing among surviving mesquite transplants. The grid also included a strip of older mesquite and saltcedar growing along the 4-m-high bank separating the xeric floodplain from No Name Lake, and a small area of the relatively dry fringe of the marsh that contained arrowweed, seepwillow (*Baccharis glutinosa*),

and cattail (*Typha latifolia*.) Shallow, standing water was occasionally noted within 10 m of some marsh stations, but permanent open water was >75 m away. I added 3 supplementary lines (27 trap stations) to the north end of the grid during trapping sessions 2 and 3 to provide additional data for estimating demographic parameters other than abundance.

I set Sherman live traps (7.6 × 8.9 × 33.0 cm), prebaited them for 1–3 days (doors locked open), and then ran them for 5–6 days at 4–5 month intervals over 1 year (Table 1). I conducted trapping sessions near dates of new moons to promote high capture probabilities for light-sensitive nocturnal species (Price et al. 1984). Further, several nights were at least partially overcast during winter (Feb) sessions, which would have promoted high capture probabilities at traps located in open or dense vegetation (Price et al. 1984, Bowers 1988). I placed a single trap within 50 cm of a pin flag marking the grid point. I placed a second trap next to the first at 19 stations located in cottonwood/willow or dense arrowweed during the last 4 days of the June session to evaluate whether trap density could be limiting the capture probabilities. In addition to setting Sherman traps, I ran 2–10 pitfall traps (plastic pails 30-cm diam × 60 cm deep) for most or all of each session, primarily in xeric habitat (150 trap nights total effort). I used oatmeal as bait in both trap types. I shaded traps during June and September sessions by

using Sherman aluminum trap tents (46 cm). Teams of 2–3 individuals checked traps each morning (starting at dawn) and afternoon, except when traps were closed during inclement weather. We rebaited traps as necessary and did not use ant repellent (Anderson and Ohmart 1977). Captured animals were given a unique mark by toe-clipping, weighed to the nearest 0.1 g, examined to determine reproductive status, age, and general condition, and then released at the capture site. We cleaned the traps with soap and water following each session. Field methods followed approved guidelines (ad hoc Comm. Acceptable Field Methods in Mammal. 1987).

Data Analysis

Analyses of population dynamics followed the robust design recommended by Pollock et al. (1990:56). I generated estimates of population size during each trapping session (\hat{N}) from box trap capture-recapture data, using a closed population model (program CAPTURE, Otis et al. 1978) in cases where the number of unique individuals captured (M_{t+1}) was >20 . I used only data from morning captures as input to CAPTURE. Animals dying during a session were not included in the input dataset, nor were animals solely captured on the supplementary lines. I increased CAPTURE-generated values of \hat{N} to account for deaths as suggested by Otis et al. (1978). I considered estimates having non-overlapping 95% confidence intervals to be different. For cases where $M_{t+1} \leq 20$, I used $\hat{N} = M_{t+1}$. I calculated density as population size divided by the area of the grid plus a 7.5-m-wide buffer strip (4.01 ha). I subjectively set the buffer strip width to be half the distance between trap stations.

I calculated sex and age ratios, estimates of mean sex- and age-specific mass, and proportions of females in various reproductive classes, using all captured individuals of known age and sex, including those captured on supplementary lines. I calculated species-specific biomass as the product of age/sex-specific density and the corresponding mean individual mass, summed over age/sex categories.

I generated survival estimates for periods between trapping sessions ($\hat{\phi}$, actually “persistence” because emigration and death could not be separated) by pooling session data (e.g., an individual was captured or not captured) and using the Jolly-Seber model for open popula-

tions (program JOLLYAGE, Pollock et al. 1990) in cases where initial population size was >20 . I tested for interspecific differences in $\hat{\phi}$, within time periods (seasons) and age classes, using a 2-tailed Z-test (Brownie et al. 1985:180) or the approximate 95% confidence intervals constructed by JOLLYAGE. I also used the latter to test for intraspecific differences in $\hat{\phi}$ between seasons, after converting $\hat{\phi}$ and associated upper and lower confidence limits to a common time unit (e.g., per day). I considered point estimates with non-overlapping confidence intervals to be different. For period 3 (between the third and fourth trapping sessions) and for cases where initial population size was <20 , I used an enumeration estimator ($\hat{\phi}^E$), calculated as the proportion of individuals alive at the end of session i that were known to be alive during session $i + 1$.

RESULTS

Small Mammal Richness

I judged that 14 of 15 small mammal species that historically occupied regional riparian habitats might have resided in the study site. I trapped 9 of these 14: cactus mouse, deer mouse, white-throated woodrat, Arizona cotton rat, desert pocket mouse (*Chaetodipus pencillatus*), Merriam’s kangaroo rat (*Dipodomys merriami*), southern grasshopper mouse (*Onychomys torridus*), western harvest mouse (*Reithrodontomys megalotis*), and round-tailed ground squirrel (*Spermophilus tereticaudus*). I also trapped 1 exotic (house mouse, *Mus musculus*). Species apparently absent from the site were the desert shrew (*Notiosorex crawfordi*), desert kangaroo rat (*D. deserti*), desert woodrat (*Neotoma lepida*), Botta’s pocket gopher (*Thomomys bottae*), little pocket mouse (*Perognathus longimembris*), and common muskrat (*Ondatra zibethicus*). Pocket gophers were present in agricultural fields adjacent to the study site but separated from it by a concrete-lined feeder canal; none was noted in the revegetated area.

Afternoon trap checks produced relatively few individuals, primarily white-throated woodrats and Arizona cotton rats. Nontarget species disturbed few traps; subadult desert cottontails (*Sylvilagus audubonii*) were caught in box traps on 16 occasions during sessions 2 and 3. Two-trap stations (run during session 2 only) contained animals in both traps in only 13 (17%) of 76 possible occurrences, and in all cases involved

Table 2. Survival estimates (and approx SEs) for adult ($\hat{\phi}_a$) and young ($\hat{\phi}_y$) of the most abundant small mammal species at the No Name Lake revegetation site, Arizona, generated using Model A2 in program JOLLYAGE (Pollock et al. 1990).

Period/species	$\hat{\phi}_a$	SE	$\hat{\phi}_y$	SE
8 Feb–4 Jun 1992				
Cactus mouse	0.693A ^a	0.074	0.619A	0.312
Deer mouse	0.130B	0.070		
White-throated woodrat	0.773A	0.089	1.000A	0.000
Arizona cotton rat	0.143B	0.132	0.000B	0.000
4 Jun–27 Sep 1992				
Cactus mouse	0.867A	0.118	0.723A	0.171
Deer mouse	0.000B	0.000		
White-throated woodrat	0.782A	0.065	0.939A	0.071
Arizona cotton rat	0.454A	0.150	1.000A	0.000

^a Estimates within an age class accompanied by the same letter have overlapping standardized (unit = /day) 95% CIs.

combinations of woodrats, cactus mice, and/or deer mice. I captured 11 individuals in pitfall traps, and although 5 individuals were trapped only in pitfalls, all 5 species captured in pitfalls also were trapped in box traps.

Population Dynamics

Cactus mouse was the most abundant rodent, reaching a peak of $\hat{N} = 215$ in the last session (Table 1). White-throated woodrat, deer mouse, and desert pocket mouse also were sufficiently abundant to enable estimation of \hat{N} with program CAPTURE in ≥ 1 trapping session. Estimates of capture probabilities generated by CAPTURE were all > 0.1 for these species, furnishing additional justification for use of CAPTURE in analysis (Otis et al. 1978:78). Values of \hat{N} for the cotton rat were 8, 16, 18, and 18 for trapping sessions 1–4, respectively. Corresponding values for \hat{N} for Merriam's kangaroo rat were 5, 8, 16, and 9; for the southern grasshopper mouse, 9, 3, 12, and 8; and for the western harvest mouse, 3, 5, 1, and 8. The house mouse was present (2–4 individuals) in all but session 1. One adult male round-tailed ground squirrel was captured in June.

Between trapping sessions 1 and 2, cactus mouse abundance stayed nearly constant, in contrast to a drop in deer mouse numbers and a doubling of the woodrat population (Table 1). The deer mouse was apparently absent by session 3 (late Sep), whereas the cactus mouse population showed a 35% increase over the June point estimate. The cactus mouse population increased between sessions 3 and 4, and deer mice reappeared in numbers exceeding those first estimated. Values of \hat{N} were higher in February 1993 than in February 1992 for 8 of the

10 species trapped, but 95% confidence intervals overlapped in 3 of 4 cases where comparison was possible. The exception was for deer mice. The lower confidence limit of \hat{N} for February 1993 exceeded the February 1992 point estimates for cactus mice and white-throated woodrats.

Survival

Cactus mouse survival was 70–90% for periods 1 and 2 (117 and 115 days, respectively; Table 2), with no difference between age classes in either period ($Z = 0.23$, $P = 0.82$, and $Z = 0.69$, $P = 0.50$, respectively). Values of $\hat{\phi}$ for adult cactus mice were not different from those for adult woodrats (period 1: $Z = 0.69$, $P = 0.50$; period 2: $Z = 0.63$, $P = 0.52$). The $\hat{\phi}$ value for adult deer mice was less than that for cactus mice ($Z = 5.53$, $P < 0.001$) or woodrats ($Z = 5.56$, $P < 0.001$) during period 1, but comparable with that for adult cotton rats ($Z = 0.09$, $P = 0.92$). Although the 95% confidence intervals for cotton rat survival in periods 1 and 2 overlapped (Table 2), $\hat{\phi}$ for period 2 (CI = $0.9932 \pm 0.0090/\text{day}$) was greater than the estimate for period 1 (0.9835/day). In contrast, deer mice disappeared from the site during period 2. Values of $\hat{\phi}^E$ for cactus mice, white-throated woodrats, and cotton rats for period 3 (146 days, 27 Sep 1992–21 Feb 1993) were 0.341, 0.636, and 0.333, respectively. Survival of desert pocket mice was relatively low each period ($\hat{\phi}^E = 0.25$, 0.14, and 0.18, respectively, for initial populations of 4, 21, and 57). Analogous estimates for kangaroo rats were 0.40 ($n = 5$), 0.71 ($n = 7$), and 0.14 ($n = 14$); values for southern grasshopper mice were 0.44 ($n = 9$), 0.67 ($n = 3$), and 0.42 ($n = 12$).

Table 3. Seasonal changes in sex/age structure (Ratio), number of reproductive females (RF), and biomass (g/ha) for small mammals at the No Name Lake revegetation site, Arizona, 1992-93.

Species	Feb 1992			Jun 1992			Sep 1992			Feb 1993		
	Ratio	RF	Biomass	Ratio	RF	Biomass	Ratio	RF	Biomass	Ratio	RF	Biomass
Cactus mouse	51:52:3 ^a	1 ^b	820	43:29:26	24	704	45:60:15	22	983	66:73:0	15	1,257
Deer mouse	10:13:2	1	283	2:2:0	1	29	0:0:0	0	0	24:29:0	11	372
White-throated woodrat	13:10:1	5	1,506	18:25:14	4	2,610	27:21:1	4	2,817	18:13:2	2	2,239
Arizona cotton rat	2:4:1	0	274	3:10:2	0	712	3:7:7	0	481	5:10:2	0	620
Desert pocket mouse	4:0:0	0	19	5:4:8	2	80	28:21:5	1	238	2:8:0	0	68
Merriam's kangaroo rat	3:2:0	0	45	3:3:1	0	64	7:6:3	1	125	4:5:0	1	89
Southern grasshopper mouse	5:4:0	0	238	2:1:0	0	82	4:5:2	1	63	4:3:0	0	50
Western harvest mouse	0:3:0	0	34	2:2:0	0	61	0:1:0	0	2	2:6:0	1	23
House mouse	0:0:0	1	17	1:0:1	1	17	2:1:1	0	21	0:0:2	10	10
Total biomass			3,219			4,359			4,730			4,728

^a Sex/age ratio is presented as numbers of unique individuals caught in the format: ad F:ad M:juv+young
^b No. F pregnant or lactating.

Reproduction and Recruitment

Subadult woodrats and adult females judged to be pregnant or lactating were present during all 4 trapping sessions (Table 3). Almost 40% of adult females were lactating in February 1992, and subadults made up 25% of the population in June. Other species showed little reproductive activity in February 1992, although several of these species did so in February 1993. I noted no pregnant or lactating cotton rats but judged 1 female to be in estrus during each of sessions 3 and 4. Determination of cotton rat reproductive status during processing was difficult. Subadult cotton rats were present during each session and made up 40% of the September population. I judged 2 of 4 adult female kangaroo rats present in February 1993 to be in estrus, and another was pregnant. Female grasshopper mice were in reproductive condition only in September, the only session in which subadults were trapped (Table 3). Subadults made up <20% of kangaroo rat and grasshopper mouse populations in September, when contributions from this age group were greatest, whereas at that same time subadults made up 40% of the cotton rat population. Subadult deer mice were only found in session 1, whereas subadult cactus mice made up 27% of the June subpopulation (Table 3). Although not statistically tested, for all species of mice, average mass of adult females was greater in June than in February, reflecting the presence of pregnant animals.

DISCUSSION

Community Structure

The small mammal community at the No Name Lake revegetation site contained 9 of 15 species considered potential residents of all forms of riparian habitat along the east bank of the lower Colorado River. Use of trapping protocols promoting high capture probabilities, evidence that trap density did not hinder capture of target species, and large trapping effort support the conclusion that 5 of 6 species not trapped were absent from the site. Additional pitfall trapping is needed to confirm the status of the desert shrew because they were known to reside in the area (4.3 km south; Howell 1976), the habitat seemed suitable, and they are not often captured in box traps (Armstrong and Jones 1972, Duncan and Corman 1991). Of the 5 species absent, only the Botta pocket gopher was known to reside in adjacent habitat.

The small mammal species most common on the site also were numerically dominant in a larger and more extensive study of lower Colorado River riparian habitats (Anderson and Ohmart 1984). Workers examined 6 community types along 450 km of the river, each of which had been affected by saltcedar invasion and/or river regulation but not intentional manipulation. Five species made up 97% of the 14,452 individuals they captured in monthly snap-trapping from 1974 through 1979: cactus mouse (59%), desert pocket mouse (12%), Merriam's kangaroo rat (10%), deer mouse (9%), and white-throated woodrat (7%). All other species that I captured also were trapped by Anderson and Ohmart (1984). Thus, the unique biophysical conditions within the revegetated area do not appear to have attracted novel, small mammal species or prevented colonization by any species commonly found in unmanipulated riparian habitat along the river.

Few studies of desert riparian small mammals provide density estimates suitable for comparison with values obtained at No Name Lake. However, at least 2 species appear to have been at high densities during the study. The densities for white-throated woodrats (8–15/ha, calculated from Table 1) are at the upper end of the range of values reported in the literature (Macedo and Mares 1988). Values based on the area actually occupied, rather than the entire grid, would be even higher, approaching 30/ha in June (Andersen, unpubl. data). Although the review by Veal and Caire (1979) provided no comparable data for cactus mice, other *Peromyscus* species show maximum densities of about 50/ha (French et al. 1976), similar to the February 1993 value for cactus mice.

Survival

Survival of adult cactus mice during period 1 (0.693), which was not different from survival for white-throated woodrats during the same period, would lead to an annual rate of 0.32/year if sustained through subsequent seasons. This annual rate was typical of a high survival rate in the classification scheme for small mammals presented by French et al. (1975). Indeed, cactus mice and woodrats continued to show similar or higher survival values during period 2. Only the low value of $\hat{\phi}^E$ for cactus mice during period 3 (0.341), which had a negative bias of unknown size (Pollock et al. 1990:32), prevented concluding both species demonstrat-

ed high annual survival. French et al. (1975) considered rates <0.02 /year to be low. This threshold was equivalent to a rate of about 0.28 for either period 1 or 2. Thus, deer mice and cotton rats showed low survival during period 1. Cotton rats demonstrated moderate survival during periods 2 and 3.

The product of the 3 $\hat{\phi}^E$ values obtained for Merriam's kangaroo rat (equivalent to 0.06, 0.34, and 0.01/yr) suggested an annual value (0.040/yr) somewhat lower than rates reported elsewhere (0.138, McClenaghan 1984; 0.241, Zeng and Brown 1987). Zeng and Brown (1987) suggested this species typically demonstrates high survival. The 3 $\hat{\phi}^E$ values for desert pocket mice suggested an annual rate of 0.006/year, substantially lower than values tabulated for congenics by French et al. (1976).

Population Dynamics and Habitat Quality

Population size increased over the year for 7 of 8 species, with a significant rise documented for deer mouse, and perhaps cactus mouse and white-throated woodrat (Table 1). However, for a habitat to be a population source area (and be designated high quality), any increase in abundance must be attributable to an excess of local reproduction over immigration. Thus, by demonstrating that reproductive output alone can account for shifts in population size and age structure under prevailing survival patterns, one can conclude that a habitat is capable of sustaining the population. In cases where reproductive output is more than sufficient, one can conclude that a habitat serves as a source area.

High survival of adult woodrats during period 1 (Table 2), coupled with recruitment (Table 3), resulted in woodrat density doubling (to $\hat{N} = 62$) and population biomass increasing by 73% (Tables 1 and 3). On the basis of the adult survival rate (0.773), the change in the adult segment of the population (determined using population size from Table 1 and age structure from Table 3) required recruitment of 24 individuals into the adult population during period 1. Given that white-throated woodrats produce a mean of 2 young/litter (Hoffmeister 1986) and survival of young is nearly 1.0 (Table 2), the requirement could be met by the estimated 17 (minimally 13) adult females present in February 1992, at least 7 of which were already pregnant or lactating (on the basis of ratios in Table 3). During period 2, recruitment of 20

individuals was needed, and the estimated adult female population from which those would have to be drawn was 20. Over the entire year, recruitment of 49 individuals into the adult population was needed. Because an estimated 17–24 adult females were present, at least 75% of which were in reproductive condition at one time or another prior to session 4, few or no immigrants are needed to explain woodrat dynamics. Given the observed population growth rate and high density present, the study site likely operated as a source habitat for woodrats, once recruits saturated the habitat.

Analogous analyses for cactus mouse (2–3 young/litter, >1 litter/yr), grasshopper mouse (2–4 young/litter), Merriam's kangaroo rat (2 young/litter), and cotton rat (possibly 10–12 young/litter) suggested that dynamics in their populations also can be explained without incorporating immigration. In the case of grasshopper mice, the observed population stability and its typical value (about 2.0/ha; McCarty 1975) suggested the study site sustained the population but did not serve as either source or sink.

Immigration may have been an important determinant of desert pocket mouse dynamics. I estimated 40 recruits were needed to account for the difference between June and September populations, a value above the contribution possible solely from the 5 adult females present in June.

The deer mouse population dropped to zero prior to September trapping and redeveloped as a result of recolonization prior to February 1993. The site may serve as a temporary refuge for this species (e.g., during periods when the adjacent agricultural fields are unsuitable). However, no individual marked in sessions 1 or 2 was recaptured in session 4. Thus, there is no evidence that an individual leaves and subsequently returns to the revegetated area. The site appears to serve as a population sink for this species.

I trapped a single ground squirrel and observed no others. Whereas round-tailed ground squirrel densities elsewhere exceeded 5/ha (Ernest and Mares 1987), Anderson and Ohmart's (1984) trapping data suggested these squirrels were rare or very localized in the lower Colorado River valley.

The apparent favorability of the study site for most small mammal species during the 1-year study may not be maintained over longer time

periods. Anderson and Ohmart (1984) documented dramatic annual fluctuations in populations of rodents in lower Colorado River riparian habitats, and the factors that affected the habitats they studied would likely also operate on revegetation sites such as No Name Lake. Anderson and Ohmart (1984) were unable to link the dynamics to interannual variation in precipitation, often considered a driving force in nonriparian desert rodent community dynamics (Brown and Zeng 1989).

Desert Riparian Ecosystem Productivity

Small mammal biomass at the revegetation site increased from 3,200 to 4,700 g/ha during the study, largely as a result of growth and reproduction. Grant and Birney (1979) reported that small mammal biomass within a New Mexican desert grassland ranged from 100 to 2,100 g/ha over several years, with a mean of 650 g/ha. The February 1993 value for No Name Lake approached the 5,200 g/ha peak value reported for all herbivorous mammals (except ungulates) residing in a montane meadow community (Andersen et al. 1980). I found no comparable values for other desert riparian habitats in the literature.

MANAGEMENT IMPLICATIONS

Revegetated desert riparian sites, even where plant establishment was only partially successful, may be ecologically valuable because of the high levels of small mammal biomass they support, independent of any contribution the site may make to maintenance of rodent biodiversity. High biomass and, presumably, productivity associated with the No Name Lake site suggested that resident rodents may be important as material processors within the ecosystem and as a prey base capable of supporting a large biomass in higher trophic levels. The apparent moderate-to-high quality of the study site for many native riparian small mammals needs to be confirmed over time and by examination of other sites where saltcedar is replaced with native species. Experimentation will be necessary to elucidate factors leading to source status. Also, the suitability of the site for small mammals should not divert attention from the broader ecological consequences of the project's failure to reach planting goals. Successful establishment

of quasi-natural vegetation along the lower Colorado River for avian habitat remains an overriding concern (Rosenberg et al. 1991) and may be necessary to restore the full functional role of the small mammal community in this desert riverine ecosystem.

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BLOOD-CLOTTING RESPONSE TEST FOR BROMADIOLONE RESISTANCE IN NORWAY RATS

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Abstract: Bromadiolone resistance in Norway rats (*Rattus norvegicus*) may reduce the efficacy of rodent control measures, and managers need a validated test to detect potential control problems. We developed a test to identify levels of resistance to the rodenticide bromadiolone in warfarin-resistant Norway rats. This test was based on changes in blood-clotting activity 4 days after administering a single dose of bromadiolone in conjunction with menadione sodium bisulphite (vitamin K₃). The test procedure identified the degree of bromadiolone resistance in laboratory and wild note that had anticoagulant resistance genes. In addition, results were available within 5 days, and susceptible animal could be humanely culled rather than dying of anticoagulant poisoning, as occurs with feeding tests. The test was based on an effective dose needed for 99% response (ED₉₉) for rats susceptible to all anticoagulants and results were determined separately for each sex. We report bromadiolone resistance in Norway rats for the first time in the United Kingdom and that rats in central southern England have a higher degree of resistance than do rats from Wales or Yorkshire.

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Key words: anticoagulant resistance tests, blood clotting, bromadiolone, Norway rat, *Rattus norvegicus*, United Kingdom.

Anticoagulants such as warfarin were first used for rodent control in the 1950s. Warfarin resistance in the Norway rat occurred within the same decade in the United Kingdom (Boyle 1960). A 6-day feeding test (Drummond and Wilson 1968) was initially used to identify warfarin-resistant Norway rats, but subsequently a 24-hour blood-clotting response (BCR) test was developed (Martin et al. 1979). Blood-clotting response tests are preferable to feeding tests because they are more accurate, more humane, quicker, less expensive, and indicate the degree of rodenticide resistance.

In 1975, difenacoum, the first of the second-generation anticoagulant rodenticides, was developed (Hadler et al. 1975). Difenacoum resistance in Norway rats was soon reported, how-

ever, in U.K. farm infestations (Redfern and Gill 1978). A feeding test was developed from combined data on warfarin-resistant and -susceptible rats to identify difenacoum-resistant rats (Redfern and Gill 1978) on the basis of the lethal feeding period needed for 98% mortality (LFP₉₈). We considered this test too stringent and subsequently developed a quicker difenacoum BCR test (Gill et al. 1993) to identify the degree of difenacoum resistance in Norway rats that had anticoagulant resistance genes. This test was based on changes in blood-clotting activity over 4 days after administering difenacoum in conjunction with menadione sodium bisulphite (MSB), a soluble salt of vitamin K₃ that is a water-insoluble chemical. We compared results from the newer test with the established feeding