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Monetary valuation of rare species and imperiled habitats as a basis for economically evaluating conservation approaches

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Abstract

Management actions directed towards the conservation of species or habitats are usually measured in resource improvement. Nevertheless, the decision to select and carry out such actions are rooted in the available funding. Therefore, to truly evaluate the benefit-costs of a conservation-directed management action, the resource improvement should be in the same metric as the expenditures. To this end, we describe here a variety of methods for attaching monetary values to rare species and habitats. We also give examples of applications with which we have been involved to demonstrate how such species and habitat valuations have allowed economic analyses of conservation approaches. The economic results helps to decide on how best to obtain the most from finite funding resources.

Resumen

Las acciones de manejo dirigidas a la conservación de especies o hábitat normalmente se miden con base en el mejoramiento de recursos. Sin embargo, las decisiones para elegir y ejecutar estas acciones están basadas en la disponibilidad de fondos. En consecuencia, para evaluar realmente el costo - beneficio de una acción de manejo para la conservación, el mejoramiento del recurso debe ser medido de la misma manera que los gastos. Con este fin, describimos una variedad de métodos para atribuir valores monetarios a especies y hábitats únicos. También, damos ejemplos de las aplicaciones que hemos usado para demostrar como estas evaluaciones de especies y habítats permiten el análisis económico de los métodos para la conservación. Los análisis económicos ayudan a decidir la mejor manera de aprovechar los fondos disponibles.

Introduction

Many endangered, threatened, or other species of special concern and their habitats, require management actions to aid their recovery. Funding is finite for recovery and conservation of species and habitats and must be carefully applied to maximize the positive impact on the protected resource. Analytical examination of the economics of management actions for resource (species or habitat) enhancement can provide managers with a logical working basis for selecting and implementing the most cost-effective conservation methodologies. While the direct costs for a conservation approach may be relatively easy to identify and quantify because they can be measured by the budgetary outlay for implementation, the rewards from those budgetary allocations are measured in terms of resource improvements, such as population growth or habitat recovery. To effectively evaluate the returns, the rewards from the expenditures must be in the same metric as the expenditures. That is, the resource improvement must also be monetarily valued. We describe some approaches that we have applied for monetarily valuing rare wildlife and habitat resources, and we review some of the conservation applications with which we have been involved.

Monetary Valuation of Rare Species

Determination of monetary values for rare species is not a straight-forward nor precise process. As an illustration, consider that values of endangered or threatened species have been deemed “incalculable” in U.S. Supreme Court case law (Tennessee Valley Authority vs. Hill 1978), the opinion going so far as to say “it would be difficult for a court to balance the loss of a sum certain - even $100 million - against a congressionally declared ‘incalculable’ value, even assuming we had the power to engage in such a weighing process, which we emphatically do not.” Despite that assessment, infinite or astronomically high monetary species valuations would be unlikely to be widely viewed as credible. Conservative and credible monetary values for rare species can be estimated through the variety of means that follow.

Contingent valuation is one method by which a value is assigned to a resource. Contingent valuation intends to measure people’s willingness to pay (WTP) for resources in a hypothetical market through the use of a survey instrument (e.g., Loomis and Walsh 1997). The respondent is asked to estimate the maximum amount he would pay to have a resource available. The payment method can be adjusted to fit the resource in question; examples include higher prices for natural area entrance fees or hunting and fishing licenses, higher trip costs, and taxes. WTP often varies greatly between payment methods. Question format can have a large influence on the results. Common formats include open-ended questions, payment cards, iterative bidding, and dichotomous choice and referenda (Loomis and Walsh 1997). Because the scenarios are hypothetical, the validity of the responses to a contingent valuation is unsure, and the results may not reflect the true WTP, either because people do not have a realistic sense of how much they would pay, or because they have incentives to dishonestly report their WTP (Loomis and Walsh 1997). To use contingent valuations of rare species in an economic analysis first requires that such survey values exist or can be generated, and that the data were obtained using statistically valid survey design principles, data collection procedures, and data analyses. Given the above, the results must be geographically and temporally relevant to the economic analyses at hand.

Legislatively designated values are another useful method for assigning societal values to resources (Engeman et. al in press; Bodenchuk et al. 2002). State wildlife and fisheries management agencies use estimates of economic values based on contributions to the economy by individual game species to derive their monetary values (Bodenchuk et al. 2002). These economic values serve as the basis for civil financial penalties for illegal kills resulting from such acts as poaching, environmental contamination, or other “takes” (Bodenchuk et al. 2002). However, rare and endangered species do not have civil financial penalties assigned in relation to their contributions to the economy as “renewable” resources, because they are rarely, if ever, exploited in a financially measurable fashion such as through the sale of hunting or fishing licenses and sportsman equipment.

While not exploited in an easily quantifiable sense, rare and endangered species are, however, almost universally protected with civil penalties set forth legislatively. More than likely, such species will have more than one value available from multiple enabling legislations (e.g., United States federal and individual state laws). Multiple applicable civil penalties pose a dilemma as to which to incorporate into an economic analysis. A conservative benefit-cost analysis is obtained when the minimal applicable value is employed. However, this could be a radical undervaluation for a species, especially when considering that all civil financial penalties from the different enabling legislations can apply simultaneously. Consider the example of predator depredations on marine turtle nests in Florida by Engeman et al. (2002). Their analyses chose the conservative route of applying a minimum legislative value of $100 from Florida statutes. However, the Florida Wildlife Code specified a value of $500 per life unit, and the federal Endangered Species Act (ESA) allows for civil penalties up to $25,000 per life unit. Thus, the monetary benefits accrued from the predator management approaches could have been as much as 250 times greater.

Breeding costs provide an empirical measure of value for a species. Captive breeding is not only a management strategy for assisting the recovery of rare species, but it also provides data for placing a value on a species. The use of captive breeding costs as a means for monetarily valuing rare species is a simple concept, because those monies spent to produce animals in captivity are empirically explicit demonstrations of a willingness to pay for new animals. The costs of captive breeding divided by the number of healthy individuals produced defines a value for the species (e.g., Bodenchuk et al. 2002). For example, the value calculated for black-footed ferret production (*Mustela nigripes*) in 1995 in this manner was $29,132 per animal (Bodenchuk et al. 2002). However, the valuing process is not quite as straight-forward as this seems. Sometimes, there are multiple captive breeding facilities for the same species, each with its own budget (e.g., Engeman et al. 2003b). A facility may remain in operation year-in and year-out, but its temporal budget and animal production may fluctuate substantially. Thus, budget and production variation among captive breeding sites for a particular species, and among years within a site, can result in substantial variation in the value for a particular species. The selection of a particular value for a benefit-cost or net benefit analysis must be carefully weighed against the objectives of the analysis. The most conservative analysis is obtained if the minimum cost per production of a healthy individual is used, whereas use of the maximum value provides the empirical peak expenditure to produce an individual of the species. Use of the median value for an individual provides an analysis representing the central tendency for valuing the species.

Monetary Valuation of Special Habitats

As with valuing rare species, credible valuation of special habitats is not straightforward. Special habitats such as wetlands have limited market value, and if such habitat is selectively protected, the market value diminishes further (King 1998). The use of contingent valuation surveys for special habitats, analogous to those applied to endangered animals is always a possibility, but they tend to be even more abstract appraisals of value (King 1998). Realistic cost estimates for restoring habitat to pristine condition (replacement costs) frequently are well in excess of the public's willingness-to-pay, and therefore, also do not represent a realistic valuation of wildlands. A defensible, logical, and applicable valuation for damaged habitat is to use expenditure data for permitted mitigation projects. Such data represent an empirical demonstration of willingness-to-pay value, and are most generally available for wetland habitats. The US dollar amounts per unit area spent in efforts to restore the various wetland habitat types has been presented by King (1998). The numbers represent the U.S. dollar amounts that environmental regulators, and to a degree elected governments, have allowed permit applicants to spend in attempts to replace lost wetland services and values (King 1998). Use of these figures, coupled with appropriate adjustments for annual rates of inflation (Zerbe and Dively 1994) leads to credible habitat valuations.

Example applications

Benefit-cost analyses of predator removal approaches for reducing losses of sea turtle nests

Historically, up to 95% of sea turtle nests at Hobe Sound National Wildlife Refuge (HSNWR), Florida have been destroyed by predation. In response, predator removal has been carried out since 1972 and was identified in a comprehensive Environmental Assessment (U.S. Fish and Wildlife Service 2000) as the only practical and legal approach for reducing nest predation on marine turtle nests at HSNWR by raccoons (*Procyon lotor*) and armadillos (*Dasypus novemcinctus*), and it is most important management program at the refuge (Bain et al. 1997). Over time, four approaches to predator removal had been applied that ranged from no removal to a predator removal contract with USDA/Wildlife Services coupled with predator monitoring using a passive tracking index (Engeman et al. 2003a). A benefit-cost analysis was conducted to compare the relative benefits of each predator removal approaches to its cost, and to each of the other approaches. Turtle reproductive data and predation data under each predator removal scenario were available and allowed estimation of the number of hatchlings that would have been lost to predation under each scenario. Predator management costs were known, therefore monetary values for hatchling sea turtles would allow the appropriate benefit-cost analyses to be conducted.

Contingent valuation and legislative values were the options considered for placing a value on hatchlings. Breeding costs for the sea turtle species nesting at HSNWR were not available. Whitehead (1992) in a contingent valuation survey had previously appraised marine turtle values at $32, however Engeman et al. (2003a) found those values to be inappropriate to generalize to the HSNWR situation due to severe survey design limitations in terms of the maximum monetary values that turtles could obtain, and use of those results would have been an extrapolation beyond the inference space of the data, both geographically and temporally. The survey was a small sample from North Carolina, whereas the Engeman et al. (2003a) study was in east-central Florida. In particular, the city bordering the Florida refuge, Jupiter Island, was considered the wealthiest in the U.S. (Nguyen 2000), thus making it unlikely that its residents would value turtles as low as in the Whitehead (1992) survey. Furthermore, the Whitehead (1992) results were approximately a decade earlier than the Engeman et al. (2002) economic analysis, making them temporally as well as geographically disjunct from the situation at hand. This exercise proved valuable as a cautionary lesson concerning the use of contingent valuations and led us (as already described) to apply a conservative, legislatively designated value of $100 from the Florida statutes, although higher values from other enabling legislations could have been used. Even so, the removal contract with USDA/Wildlife Services coupled with predator monitoring was found to have the highest benefit cost ratio, with a $5,000 contract resulting in conservatively estimated savings of $8.4 million in hatchling sea turtles (Engeman et al. 2002).

Hypothetical benefit-cost ratios for managing predators that threaten Puerto Rican parrots

The Puerto Rican parrot (*Amazona vittata*) (Figure 1) is one of the 10 most endangered birds in the world (U.S. Fish and Wildlife Service 1999), with only 30-40

Photo courtesy of Richard Engeman

Figure 1. The Puerto Rican parrot (*Amazona vittata*)

Photo courtesy of Richard Engeman

Figure 2. Feral swine(*Sus scrofa*) foraging and damaging valuable habitat.

birds comprising the single wild population. As with many endangered or locally rare species (Hecht and Nickerson 1999), predation has been identified as one of the factors limiting Puerto Rican parrot productivity in the wild (Snyder et al. 1987; Lindsey et al. 1994; U.S. Fish and Wildlife Service 1999). Parrot recovery efforts require many high-cost expenditures such as captive breeding, but the economic benefit from expenditures on predator management had not been analyzed.

To address this issue, we conducted an economic analysis of predator management for protecting Puerto Rican parrots (Engeman et al. 2003b). Five years of data on the production costs and the corresponding number of healthy fledglings produced from three highly managed populations (the wild and the two captive populations) were used to value Puerto Rican parrots. Resulting parrot valuations over years and populations ranged from $2,415 to $100,000 per individual. The median annual value from combining the expenditures each year for the three populations was $25,500 per parrot. Predator management costs were estimated from existing U.S. Department of Agriculture/Wildlife Services contracts for similar work in Puerto Rico. If median parrot values were applied, then only one parrot would have to be saved from predation every 2.6 years to allow the combined management for all predator species to be cost-effective. If the year of maximal parrot values (averaged over captive and wild populations) was used, then only one parrot saved every 4.2 years would make application of all predator management methods cost-effective. Use of the single highest per-parrot value from among years and populations would result in the combined application of all forms of predator management being cost-effective if only one parrot is preserved from predation every 11.8 years. Subsequently, predator management is now viewed as a component of the parrot recovery process that is unaffordable to omit.

Economically evaluating a structural method for reducing road kills of royal terns at bridges

Royal terns (*Sterna maxima*) in Florida are listed as a "species of special concern" by the Florida Committee on Rare and Endangered Plants and Animals (Egensteiner et al. 1996). Collisions with vehicles cause many royal tern road-kills at some coastal roads and bridges in Florida (Skoog 1982; Smith et al. 1994; Bard et al. 2002b). We examined the benefit-costs to royal tern conservation from a multi-year trial of a simple hazard reduction method applied to a bridge in east-central Florida (Shwiff et al. 2003), whereby metal poles were fastened vertically on both sides of the bridge to reduce the number of collisions between vehicles and birds by influencing them to fly well over bridge traffic (Bard et al. 2002a).

The benefit-cost analysis (BCA) of the structural modification involved estimating the monetary value of the benefits, measured in terns saved by reduced road-kills at bridge sites, versus the costs of making structural (i.e., erecting poles) modifications. Legislatively designated values from the Wildlife Code of the State of Florida (Chapter 39 F.A.C.) that specify up to a $500 fine for "take" were applied for a conservative analysis, and the U.S. Migratory Bird Treaty Act (16 U.S.C. 703-711), which specifies up to a $2,000 fine for "take" of any migratory bird, provided an upper range on tern values. The initial expenditure of $5,900 to erect the poles provided protection for 5 years (1995-1999). The five full years of protection cost an average of $1,180/yr. The average number of road-killed royal terns during this same period was 5.2 terns/yr, which was 14.2 terns/yr less than the average of 19.4 terns/yr for the 5 years before erection of the poles. Using the $500 per tern value, the average loss values before and after the structural modification program were \$48,500 and \$13,000. The corresponding values using the \$2,000 per tern value were \$194,000 and \$52,000. The average of 14.2 terns/year saved with a value of \$500 per tern produced an average annual savings of \$7,100. Over the 5 year period, the structural modifications provided a cumulative annual rate of return on the initial \$5,900 investment that increased from 20 % after year 1 to 502 % after five years.

Valuing Florida wetland habitat lost to feral swine damage

We carried out studies in two wetland habitat types in Florida whereby we estimated the amount and value of the habitat damaged through rooting by feral swine (*Sus scrofa*) (Figure 2). First, we monitored swine damage to native wet pine-flatwoods at three state parks from winter 2002 to winter 2003 (Engeman et al. 2003d). We also estimated the amount and value of swine damage to the last remnant of a formerly extensive basin marsh system now located only in Savannas Preserve State Park (Engeman et al. 2003c). While different sampling approaches were required to estimate damage in the different habitats, we used the same concept for attaching unit-area economic values to the habitat damage. For each study, we identified the dollar value for the appropriate wetland habitat category from each of the two studies in King (1998). The cost-per-unit area of swine damage in each case was calculated by multiplying the estimated proportion of area damaged by swine by the cost-per-unit area for habitat restoration.

The three parks where we examined damage to wet pine-flatwoods had different swine management histories and the damage patterns differed among them over time. The park in which swine were intensively removed in 2000 initially had the lowest habitat damage at 1.3%, but as a result of natural and artificial population growth it rose to 5.4% by the conclusion of the study, and was valued at \$19,193-36,498/ha. A park with no history of swine harvest had damage escalate from 2.6% to 6.4%, with an associated value of \$22,747-43,257/ha. Swine were managed as game animals in the third park prior to its inclusion into the state park system in 2000. Its proportion of area damaged decreased from 4.3% to 1.5%, valued at \$5,331 - \$10,138/ha. We attributed this decrease to human activities associated with development of the park's infrastructure causing dispersal of animals conditioned to avoid humans by hunting. Damage was highly scattered in each park, as evidenced by a much higher proportion of sampling sites showing damage than the actual proportion of land area damaged. The dispersed nature of small amounts of damage would tend to increase the effort for recovering habitat and make damage value estimates more conservative. Damage valuation estimates also were conservative because it was impossible to incorporate values for such contingencies as swine impact to state and federally listed endangered plants in the parks, some of which are found nowhere else in the world.

We found that swine damaged 19% of the exposed portion of the basin marsh in our study area. Seventy percent of the sample sites showed swine damage at the shoreline ecotone and 58% showed damage at the upland ecotone. The area damaged within our study site alone was valued between \$1,238,760 and \$4,036,290. In estimating the monetary values of the swine damage to the habitat we assumed standard costs for restoration. The periphery of the entire basin marsh would be about five times our study site. The cost of this contract was \$7,500, and represents only a minor fraction of the value of the swine damage to an average single ha of the exposed basin marsh, let alone to the synergistic value of the swine damage.

Benefit costs of removing feral cats to protect Key Largo wood rats

Worldwide, feral cats (*Felis catus*) are well-known to be highly destructive predators of native species. We are currently in the initial phases of data collection in Key Largo, Florida to document efficacy of feral cat removal efforts for protecting the highly endangered Key Largo woodrat (*Neotoma floridana smalli*). A companion component to the efficacy assessment will be to economically assess the cat removal efforts in terms of the dollar value of its impacts to the woodrat population. To do this, Key Largo woodrats will require valuation. Options for this include state and federal legislative values as for the sea turtle and royal tern examples, or if available, captive breeding costs from a breeding program now in its infancy. In this manner, the benefit-costs of feral cat removal as a Key Largo woodrat conservation tool can be valuated.

Summary

The ability to value rare wildlife or sensitive habitat resources provides a necessary and effectual tool for evaluating conservation approaches. Economic information and analyses can greatly assist managers on how most efficiently and effectively to allocate limited funds towards species conservation. Ultimately, many conservation funding decisions are made on a political level by people without high levels of training or expertise in biological sciences. Placing conservation issues in an economic context can greatly enlighten the political decision making process in an increasing economic arena.

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Use of roadkill data to index and relate raccoon activity at a heavily predated, highdensity marine turtle nesting beach

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Abstract

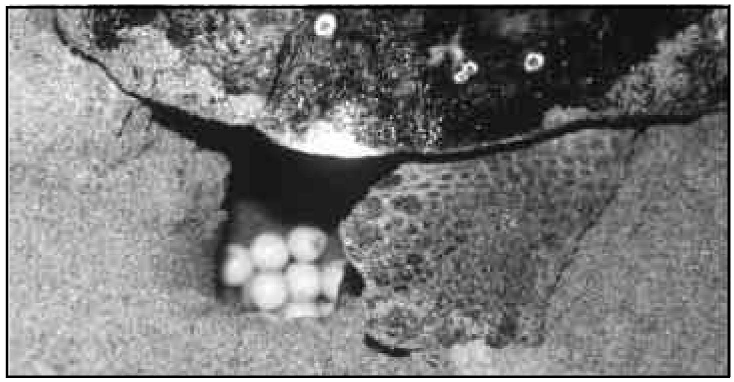
Four years of data from a high-density marine turtle nesting beach at John D. MacArthur Beach State Park, Florida were examined along with data on raccoon road-kills from adjacent roads, and data on park attendance (as an index of local traffic) to make inferences about raccoon activity patterns relative to turtle nesting. Raccoon road-kills were found to diminish substantially during turtle nesting, even though local traffic was constant or increasing. Opossums, the only other mammal consistently found as road-kills, did not show a decrease during turtle nesting season, but they are not known as a primary predator of turtle nests. We concluded that during turtle nesting raccoons are drawn to the beach to prey on the abundant food resource of turtle eggs, and they do not leave the beach until the end of turtle nesting season. High numbers of raccoon road-kills during the fall-winter, followed by a decrease in the spring around the start of turtle nesting season, might be used as indicators to initiate management actions to protect turtle nests.

Resumen

Cuatro años de datos recolectados de una playa de anidamiento de alta densidad de tortugas marinas en John D. MacArthur Beach State Park, Florida fueron analizados conjuntamente con registros de mapaches atropellados en carreteras contiguas y datos de visitación al parque (como un indicador del tráfico local) para inferir patrones de actividad del mapache con relación con el anidamiento de tortugas. Los mapaches atropellados se reducen sustancialmente durante la desova de tortugas, aunque el trafico local se mantuvo constante en aumento. La comadreja, el único otro mamífero encontrado consistentemente atropellado, no mostró una reducción en su población durante la desova de tortugas, pero no son conocidos como un depredador primario de los nidos de tortugas. Hemos concluido que durante la época de la desova de tortugas, los mapaches son atraídos a la playa por la abundancia de huevos de tortugas y no dejan la playa hasta el final de la temporada de la desova de tortugas. Los números elevados de mapaches atropellados durante el otoño-invierno, seguido por una reducción en la primavera alrededor del comienzo de la temporada de la desova de tortugas, puede ser usado como un indicador para iniciar acciones de manejo para protección de los nidos de tortugas.

Introduction

Predation is a critical threat to many endangered or even locally rare species (Hecht and Nickerson 1999), and predation losses can have an increased deleterious impact due to the compounding effects of habitat loss and altered predator communities (Reynolds and Tapper 1996). In this regard, raccoons *Procyon lotor* cause substantial destruction of marine turtle nests in Florida and throughout the southeastern United States (Stancyk 1982); thus, they exemplify an abundant native vertebrate that negatively impacts the conservation of endangered species (e.g., Garrott et al. 1993). While urbanization and development of coastal Florida have reduced the beach areas where marine turtles successfully nest, raccoons have prospered in the face of urbanization. They flourish in close association with humans where their populations often receive artificial support through refuse or direct feeding (Dickman 1987; Dickman and Doncaster 1987; Riley et al. 1998; Smith and Engeman 2002). Increased availability and concentration of food, den sites or other refuges may induce dense populations of wildlife species that inhabit urban environments (e.g., Dickman 1987; Dickman and Doncaster 1987; Riley et al.1998), and raccoons have been observed to achieve extraordinary densities (up to 238/km^2) in urban, coastal Florida (Smith and Engeman 2002). In addition, predators are known to recognize and key on high-density nesting areas (Lariviere and Messier 1998, Mroziak et al. 2000). Here, we examine four years of data from a high-density turtle nesting beach enclosed within an urban setting. We examine raccoon road-kill data from area roads during the same years to evaluate whether a raccoon migration to the high-density of nests is indicated.



Figures 1-4. This series of photos detail the lives of loggerhead turtles at John D. MacArthur Beach State Park, FL. Female loggerheads build a nest in the beach (top photo) and lay their eggs in the sand (second photo from the top). If raccoons or other predators find the nest, eggs will be eaten (second photo from the bottom), otherwise, hatchlings will emerge and head towards the ocean (botoom photo). Photos courtesy of Richard Engeman

Methods

Study site

John D. MacArthur Beach State Park (MBSP) is located on Singer Island in Palm Beach County, Florida, USA. It

Month	Mean number of nests deposited	Mean park attendance (1000s)	Mean # of road-kills	
			Raccoons	Opossums
January	0.00	7.653	5.50	0.50
February	0.00	9.098	3.25	0.50
March	0.00	12.608	1.25	1.00
April	2.50	11.280	1.25	0.75
May	213.75	8.071	1.75	0.25
June	518.50	6.344	0.25	0.25
July	485.25	8.777	0.50	0.50
August	106.50	7.551	0.75	0.50
September	1.75	5.121	2.50	1.25
October	0.00	4.816	3.25	1.00
November	0.00	5.166	8.75	1.75
December	0.00	6.362	6.75	0.25

Table 1. Yearly averages from 1995-1998 for marine turtle nest deposition (3 species combined), raccoon road-kills, opossum road-kills, and visitor attendance at John D. MacArthur Beach State Park, Florida.

consists of 65 tidal wetland/submerged ha, and 71 upland ha for a combined total of 136 ha. Terrestrial plant communities consist of maritime hammock (49 ha) and beach dune (9.3 ha). MBSP is encapsulated within the City of North Palm Beach, and is surrounded by suburban infrastructure to the north and south. The property is bordered to the east by the Atlantic Ocean, and the Intracoastal Waterway (a large bulkheaded estuary) truncates the entire western boundary. State Road A-1-A runs through MBSP parallel to the Intracoastal Waterway on the west side of Singer Island. This length of road is 2.6 km with a speed limit of 72 kph. The park also has another 1.1 km of infrastructure roads with a speed limit of 24 kph. No roads are immediately parallel to the beach on the Atlantic coast. Thus, wildlife from the beach would be unlikely to appear on the roads within a short time period.

There are 3 km of Atlantic Coast beach available for nesting by three threatened and endangered species of marine turtles (U.S. Fish and Wildlife Service 1994): loggerhead *Caretta caretta*, green *Chelonia mydas*, and leatherback *Dermochelys coriacea* turtles. Over the past 10 years, this span of beach has received an average of approximately 1,300 marine turtle nests each year (Desjardin et al. 2001).

Marine turtle nesting and road-kill surveys

During 1995-1998, MBSP rangers inspected the 3 km of beach each day from 1 March through 30 September. Surveys were initiated within 0.5 hr after sunrise and the number of new turtle nests was recorded each day, and those numbers were tabulated monthly.

A daily road-kill survey was conducted during 1995-1998, and consisted of slowly searching park and adjacent road surfaces for dead wildlife while driving ca. 8-24 kph (e.g., Smith et al. 1994; Bard et al. 2002; Shwiff et al. 2003; Smith et al. 2003). Surveys were initiated between 07:45-08:15 a.m. The numbers of each species observed as road-kills were recorded, and also tabulated monthly. To assess whether road-kills were a reflection of human traffic instead of reflecting a response to turtle nesting, we obtained park attendance data to index traffic volume on the roads in the area.

Data analyses

Several quantitative approaches were applied to the nesting and road-kill data to ascertain the existence of a relationship between turtle nesting and raccoon activity. The most direct approach was to examine the correlation between monthly nest deposition and road-kills. The number of nests currently in the beach each month might have provided a more refined variable to relate with raccoon activity, but this could not be calculated because nest removal rates due to hatching, predation, overwash, etc. were not available. Most months, turtle nesting was zero, but during the summer (nesting season), it ranged to over 650 nests/mo, making the nesting data non-normal. Therefore, Spearman's rank correlation (r) was used to measure the strength of relationship between turtle nesting and the other variables.

Another analysis compared average monthly road-kill rates between the times when turtle nests were being deposited and when they were not being deposited. This was carried out as a randomized block design where year was the blocking factor and it was analyzed as a mixed linear model (e.g., McLean et al 1991; Wolfinger et al. 1991) using SAS PROC MIXED, with a restricted maximum likelihood estimation (REML) procedure (Littell et al. 1996).

Comparative analyses were conducted where activity also was indexed by road-kills for other mammals. These data were analyzed in the same manner as that for the raccoons. These analyses provided an indication of whether raccoon activity patterns were typical for mammals, and therefore a function of other external forces, or whether raccoon activity stood out by itself relative to turtle nesting. Park attendance data were analyzed in the same fashion to see if traffic patterns in the area followed the same patterns as raccoon road-kills, or if raccoon road-kills could not be explained by traffic patterns.

Results

Over the four years, turtle nests were only deposited in April-September. Very few nests were deposited in April and September, but very large numbers were deposited May-August (Table 1). Thus, very few eggs were in the beach sand in April, but many remained in the sand in September from previous months of turtle nesting.

The results were striking for the analytical approaches used to relate turtle nesting to raccoon activity. Raccoon activity as indexed by road-kills was dramatically lower during months with turtle nesting than during non-nesting months ($F_{1,3} = 10.94$, $p = 0.04$). The only other mammal recorded more frequently than as incidental road-kills (i.e., > 5/yr, on average) were opossums *Didelphis virginiana*, which showed no difference between nesting months and non-nesting months ($F_{1,3} = 1.34$, $p > 0.3$). As would be expected, after viewing the above results, raccoon road-kills showed a negative rank correlation ($r = -0.60$, $p < 0.0001$) with turtle nest deposition, again indicating that when nest deposition rates were high, few raccoons were along the roads. In contrast, the correlation of opossum road-kills with turtle nesting was not distinguishable from 0 ($r = -0.17$, $p = 0.24$).

Park attendance was not strongly related to raccoon road-kills at $r = -0.22$ ($p = .14$). No differences were detected in park attendance between nesting and non-nesting months ($F_{1,3} = 0.45$, $p > 0.50$). Both attendance results indicate that the raccoon road-kill rate was not related to local area traffic, or if so, the relationship was very minor and opposite of what would be expected with fewer raccoon road-kills at times of higher traffic volume.

Discussion

The difference in raccoon road-kill rates between turtle nesting and non-nesting months was compelling. While we did not have data on traffic flows, park attendance data during the summer when few raccoons were being killed by traffic did not diminish when compared to fall-winter months when raccoon road-kills were highest. Furthermore, it would not be reasonable to expect traffic to decrease near a beach during summer holidays. In support of this, road-kills of opossums, only known to very rarely act as a primary predator of turtle nests (Woolard et al. in press), were not found to be less during turtle nesting season.

Our only practical explanation for these results is that raccoons were actively moving about the MBSP area until the beginning of turtle nesting. At that time they appeared attracted to the abundant food resource on the beach that thousands of nests of turtle eggs represent, as occurs commonly along the Atlantic coast of Florida (Stancyk 1982; Bain et al. 1997; Mroziak et al. 2000; Engeman et al. 2003). They would not leave the beach until that food resource diminished. Afterwards, they dispersed from the beach, and again were vulnerable to becoming road-kills. The relationship of raccoon road-kills to turtle nesting might be applied to assist marine turtle conservation at beaches with high nest predation. High numbers of road-kills during the fall-winter, followed by a decrease in raccoon road-kills in spring around the start of turtle nesting might be used as indicators to initiate management actions to protect turtle nests.

Evidence suggests that raccoon migrations to turtle nesting beaches may have a cultural ("learned") component (passed on from one generation to the next), because on some beaches most raccoon predation occurs on the night of egg deposition (Anderson 1981), while on others, predation rarely occurs then (Ehrhart and Witherington 1986; Engeman et al. 2003). A migration to a nesting beach that is culturally produced could well be lost over a few generations. For example, Engeman et al. (2003) demonstrated that a passive tracking system can be used to optimize predator management. As a consequence, predation on a high-density turtle nesting beach at Hobe Sound National Wildlife Refuge (HSNWR), 21 km north of MBSP, dropped from 42% to 29% in one year (Engeman et al. 2003). A further two years of this practice through 2002 reduced predation by raccoons and armadillos (*Dasypus novemcinctus*) on turtle nests to 9% (HSNWR, unpublished data). This suggests that a cultural cycle of turtle nest predation by raccoons at HSNWR may have been broken.

The chronology of the raccoon reproductive cycle, taken into consideration with our road-kill data, supports the premise of raccoons focusing their activities on the beach during turtle nesting season. Raccoon litters in Florida are typically born in March and April, with weaning from mid-May to July (Kern 2002). Thus, one would expect young of the year to inflate road-kill statistics during summer when turtles are nesting. However, that the opposite occurred could be attributed to the young accompanying mothers to the beach and also would suggest a cultural component to turtle nest predation.

Predation was the primary factor affecting the success of turtle nests at MBSP, with a depredation rate of 42.6% in 2001 (Desjardin et al. 2001). It is logical that similar predator management at MBSP as at nearby HSNWR could yield similar results. Engeman et al. (2002) demonstrated that a $5000 contract to manage predators during turtle nesting at HSNWR in 2000 yielded an $8.4 million return in marine turtle hatchlings using only a minimal monetary valuation for individual hatchlings. Investment in similar predation management strategies at MBSP might prove equally beneficial.

We can extrapolate in a logical fashion on how this might work at MBSP. If an average of 1,300 turtle nests are deposited annually at MBSP, then a 43% predation rate implies the loss of approximately 560 nests. With loggerhead turtles comprising approximately 98% of nests (Desjardin et al. 2001), an estimate of an average of 100 eggs/nest (Desjardin et al. 2001; Engeman et al. 2002) would be conservative. Thus, an average of at least 56,000 eggs would be lost to predation annually. Assuming a hatching rate similar to the 75% reported for HSNWR (Engeman et al. 2003) suggests an average net loss of 42,000 hatchlings/year at MBSP due to nest predation. Just halving the predation rate would produce an average of 21,000 more hatchlings/year. Because the MBSP beach is only 60% the length of the beach at HSNWR, it is logical to assume that expenditures at MBSP for the same level of predator management would be no more than that at HSNWR. Applying the same conservative turtle valuation as Engeman et al. (2002) suggests that a savings of over $2 million in turtle resources could result.

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Book Review
Saving Asia's Threatened Birds:
A Guide for Government and Civil Society

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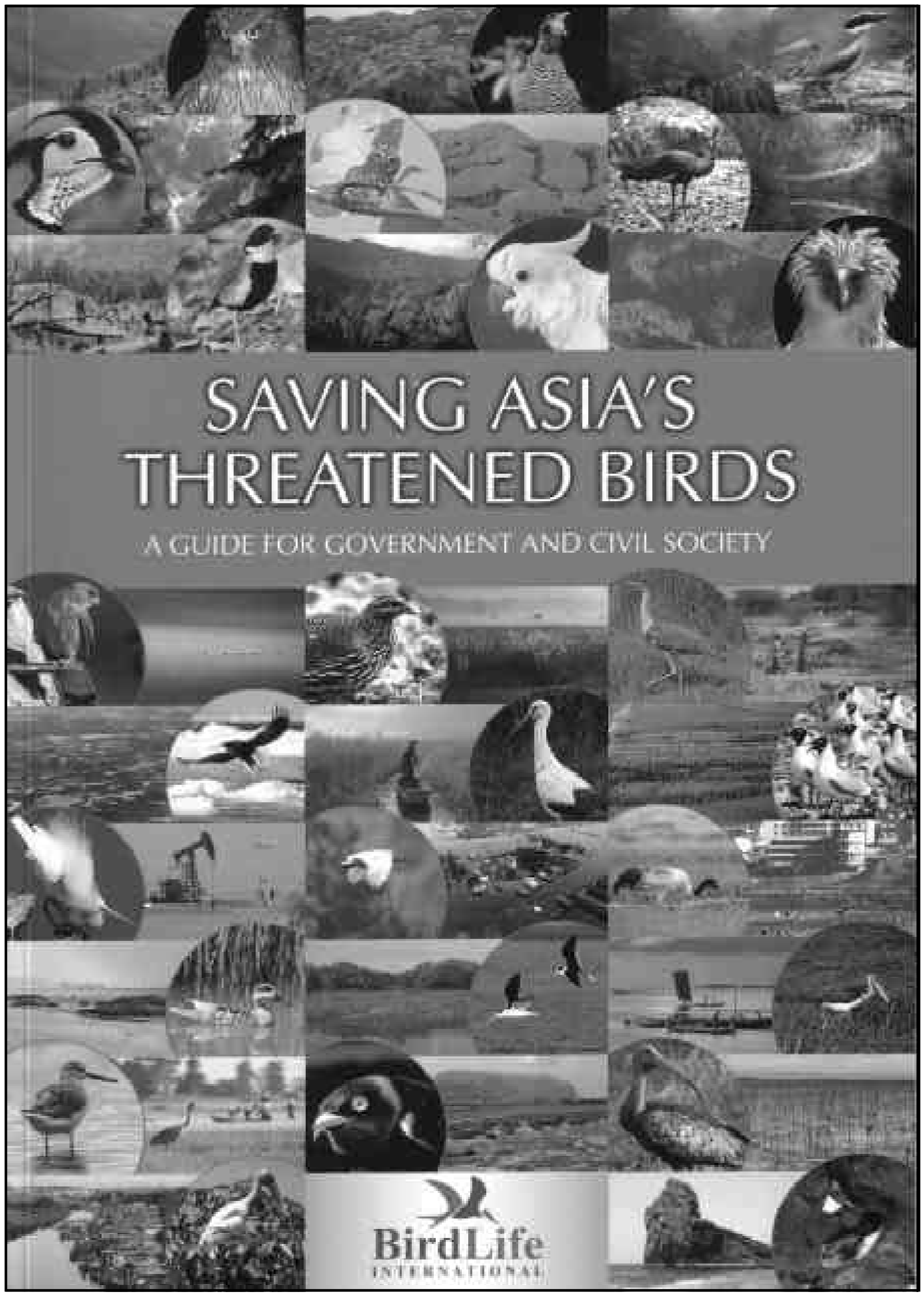

Saving Asia's Threatened Birds is an important work providing a much needed and novel synthesis of the complex issues concerning avian conservation in Asia. This book is the result of a cooperative undertaking between BirdLife International (BI) and the Critical Ecosystem Partnership Fund (CEPF). The text draws largely from the longitudinal avian studies of BI's core scientific endeavor, the Globally Threatened Species Programme. This program has collected and disseminated scientific avian research since the 1970's and is responsible for producing the international Bird Red Data Book. Funding for the writing of Saving Asia's Threatened Birds was provided by CEPF, an initiative of Conservation International (CI). CEPF is a collaborative alliance between CI, the World Bank, the Global Environment Facility, the MacArthur Foundation, and the Japanese Government. Perhaps commonalities between BI's Important Bird Areas (IBAs) and CI's Global Hotspots facilitated the book's development of and focus on systematic definitions of habitat regions rich in avian biodiversity and threatened by human development. It is evident from the acknowledgements that Saving Asia's Threatened Birds was a monumental endeavor, involving perhaps 100s of people directly, and 1000s indirectly via their contributions to the international Bird Red Data Books. This book is noteworthy for its concise presentation of an otherwise voluminous amount of material.

Saving Asia's Threatened Birds successfully presents the main findings of BI's work with Asian avifauna in a clearly written and visually rich manner. The main findings of the Bird Red Data Book are paired with discussions focusing on conservation issues and potential policy solutions relevant to 33 habitat regions in Asia. Maps, photographs, tables, and figures are used frequently and effectively, clearly linking complex and sometimes disparate data sources. The book is divided into four sections. "Asia: Birds, Habitats and People" presents a brief overview of the birds, habitats, and cultures of Asia. This first section concludes with short country- (or territory-) specific synopses including counts of threatened bird species, lists of the habitat regions per country, and page numbers linking to later chapters. The second section, "Asia's Threatened Birds and Their Habitats", explains the globally threatened avian species work developed by BI and includes definitions of the habitat regions used throughout the text. The third section, "Conservation Issues and Strategic Solutions", examines the conservation issues and their associated policy implications for individual habitat regions. The last and largest section, "Data Presentation", is devoted to a systematic synthesis of habitat region accounts. This well-organized section delivers a habitat-region-by-habitat-region chronicle of threatened bird species data, causes of habitat loss, and examination of policy implications.

The "Asia: Birds, Habitats and People" (pp 3-17) section of the book provides a brief overview of the biodiversity found in Asia's heterogeneous mix of habitats. A short summary identifies the economic, aesthetic, and cultural value of birds in Asia and how issues, such as growing human populations and developing economies, create pressures on habitats and biodiversity. The section ends with a series of country- (or territory-) level synopses. Each synopsis includes: a summary paragraph; color coded keys to both threatened bird species (Critically Endangered (CR), Endangered (EN), Vulnerable (VU)) and habitat regions (forest, grassland, wetlands); and page numbers linking to later chapters. These page numbers are important because most of the book is organized by habitat region and not by country.

"Asia's Threatened Birds and Their Habitats" (pp 18-21) introduces The Threatened birds of Asia: the BirdLife International Red Data Book, 2001, produced by BI in partnership with IUCN-The World Conservation Union. This two-volume Red Data Book generated detailed accounts of 323 threatened

Asian bird species. Information extracted for Saving Asia's Threatened Birds includes recommendations for avian conservation, identification of key habitats, and Important Bird Areas (IBAs). Analyses of distributions and habitat requirements of Asia's threatened birds produce a priority list of key habitat regions: nine major forest regions, three grassland regions, and 20 wetland regions. These key habitat regions overlap considerably with the priority regions identified by CI's Global Hotspots, BI's Endemic Bird Areas, and World Wildlife Fund's Global 200 Ecoregions (Table 1, p 20). Three full-page maps illustrate the Asian region with overlays of forest (Fig. 4, p 22), grassland (Fig. 5, p 23), and wetland (Fig. 6, p 24) habitat types. The maps include sidebar legends with color and alpha-numerically coded tabs corresponding to habitat type. Again, page references link the habitat regions identified on these maps to text found later in the book.

"Conservation Issues and Strategic Solutions" (pp 25-36) packs a wealth of information and may be of particular interest to individuals concerned with conservation policy. This section identifies human threats to Asia's avifauna and presents conservation initiatives to address these threats. Habitat loss and degradation is identified as the main threat to Asian birds. Table 1 (pp 26-27), an information-dense table, enumerates policy recommendations addressing direct causes of habitat loss by habitat region. This table links causes of habitat loss (i.e. timber extraction) to habitat region (forest) and policy recommendations (establish national forests, implement sustainable forest management practices, encourage forest certification; etc.). Hunting, wild bird trade, and invasive species issues are summarized in Table 2 (p 28) and discussion of indirect causes of biodiversity loss (increasing consumption, poverty, land tenure, etc.) appear on pages 28-29. An interesting policy section begins with Table 3 (p 31), which lists each Asian country's participation status in international agreements (such as the Convention on Biological Diversity, Ramsar, CITES, and others). The text focuses on each international agreement in turn, briefly summarizing the goals of the agreement and offering recommendations to strengthen existing agreements. The section ends with a series of tables connecting the Critical and Endangered bird species of Asia to: key habitat regions and human causes of habitat loss (Table 9, p 37); IBAs crucial to the survival of 66 species (Table 10, p 38); hunting, egg collection, and wild bird trade (Table 11, p 39); Asia's 'lost species' (avian species not recorded in recent decades) (Table 12, p 40); gaps in our understanding and knowledge (Table 13, p 40).

"Data Presentation" (p 41-240), which is the bulk of the book, condenses an abundance of information into well-organized and systematic habitat region accounts. Each habitat region is treated in a standardized format (explained on pp 41-42) that includes four sections. The "Habitat Overview" begins by identifying the globally threatened bird species occurring in the region, the habitat and altitudinal requirements of these species, and the countries and territories the region covers. A Summary Table identifies the number of species listed by IUCN Red List Category and the species' breeding/ non-breeding status in the region. This Summary Table links to Table 2, a larger table that includes information concerning species' distribution and population status. Subsequently, "Outstanding IBAs For Threatened Birds" identifies the outstanding IBAs and IBA-associated threatened bird species in the region. Table 1 summarizes each IBA's protected/listed status under international conventions; names the IBA island or territory location; and identifies associated threatened species and habitats. A map of the region is color coded for each habitat type, illustrating the geopolitical extent of each habitat region, and indicating the locations of IBAs and Endemic Bird areas."Current Status of Habitats and Threatened Species" reviews the current conditions of habitats, identifies current and past habitat threats, and lists conservation measures that are currently in place. Finally, “Conservation Issues and Strategic Solutions” is a large text section which examines: causes of habitat loss and degradation; protected areas coverage and management; human exploitation of threatened birds; and gaps in our knowledge concerning threatened birds. Table 3 succinctly summarizes the conservation issues and suggested strategic solutions particular to each habitat region.

Overall the book is visually lush and an easy-to-use practical reference for a variety of matters concerning the threatened birds of Asia. Saving Asia’s Threatened Birds has broad appeal and will be useful to scientists and non-scientists, policy makers, governmental and non-governmental organizations, and educational institutions and teachers.

The Impact of the Proposed Eddie Frost Commerce Park on *Speoplatyrhinus poulsoni*, the Alabama Cavefish, a Federally Endangered Species Restricted to Key Cave, Lauderdale County, Alabama

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Abstract

The Alabama cavefish, *Speoplatyrhinus poulsoni*, is one on of the most endangered species of fish in the world due to its low population size, restriction to a single cave (Key Cave), and threats to the cave's recharge area. The recharge area has historically been altered by deforestation and reservoir construction. There is a proposal to construct the Eddie Frost Commerce Park west of Florence, Alabama within part of the recharge area of Key Cave. The construction of this industrial park will have negative affects on the Key Cave aquifer and *S. poulsoni*, including unsuccessful reproduction from changes to flood cycles within the aquifer, degradation of groundwater, drainage and hydrologic alterations, and lower ground water levels. Environmental pollution is a major threat to the Alabama cavefish, and this threat will be magnified with the proposed industrial park, any growth surrounding the park, and pollutants from adjoining roadways. Proposals to divert pollutants out of Key Cave's recharge area and into the Cypress Creek drainage are flawed; dye-tracing tests indicate that the surface drainage divide between the Key Cave and Cypress Creek aquifers are meaningless with regards to subsurface water flow. One acute pollution event, even if detected by proposed monitoring stations, could not be stopped from reaching Key Cave, and could lead to the death of Alabama cavefish. Even if users of the industrial park are restricted to "clean" industry, the proposed park will likely disturb the critical habitat of *S. poulsoni* and may cause its extinction.

Resumen

El “pez de la cueva” de Alabama, *Speoplatyrhinus poulsoni*, es una de las especies de peces en mas peligro en el mundo debido a su baja población, su restricción a una sola cueva (Key Cave), y a las amenazas a la zona de recarga de la cueva. La zona de recarga ha sido alterada históricamente por deforestación y por la construcción de embalses. Actualmente, existe una propuesta para construir dentro de un parte de la zona de recarga de Key Cave el parque industrial Eddie Frost al oeste de Florence, Alabama.. La construcción de este parque industrial tendría un impacto negativo en el acuífero de Key Cave y S. poulsoni, incluyendo reproducción fracasada por cambios del ciclo de inundaciones dentro del acuífero, degradación del agua subterránea, alteraciones hidrológicos y de drenaje, y reducción de los niveles del agua subterránea. La polución del ambiente es una amenaza grande para el “pez de cueva” de Alabama que aumentara con el parque industrial propuesto, el desarrollo circundante al parque y agentes contaminadores de las carreteras contiguas. Las propuestas para desviar los agentes contaminadores fuera de la zona de recarga de Key Cave en el drenaje de Cypress Creek son defectuosas; pruebas de tintura adelantadas indican que la división del drenaje superficial entre los acuíferos de Key Cave y Cypress Creek no tiene sentido con respecto a la corriente de agua subterránea. Si hubiese un evento agudo de contaminación, aun cuando sea detectado por las estaciones de monitoreo propuestas, no podrá evitarse que drene hasta Key Cave y puede causar la extinción del pez de la cueva de Alabama. Incluso si los usuarios del parque industrial se les restringe como industria “limpia”, el parque propuesto perturbara muy probablemente al hábitat de S. poulsoni y causara su extinción.

Introduction

The Alabama cavefish, *Speoplatyrhinus poulsoni*, is one of the rarest and most endangered species of fish in the world (Cooper 1980; U.S. Fish and Wildlife Service 1982; 1985; 1988; 1990; Romero 1998; Romero and Paulson 2001; Trajano 2001). The U.S. Fish and Wildlife Service (USFWS) lists the Alabama cavefish as endangered because it is in danger of extinction (USFWS 1988), while the International Union for Conservation of Nature and Natural Resources (IUCN) lists it as critically endangered due to its extremely high risk of extinction in the immediate future (Hilton-Taylor 2000). The critical imperilment of this species is due to its low population size (less than 100 individuals), restriction to a single cave (Key Cave) in Lauderdale County, Alabama, and threats to the cave's recharge area (USFWS 1982; 1985; 1990; Romero 1998; Romero and Paulson 2001). These threats to the recharge area, outlined in three different approved versions of the USFWS Recovery Plan for *S. poulsoni* (1982; 1985; 1990), represent the official position of the Service with regards to protection of the Alabama cavefish under the Endangered Species Act. In the latest version (Second Revision 1990), the Recovery Plan lists numerous factors likely to cause the decline of *S. poulsoni*. All of these factors are associated with current and potential alterations of Key Cave's recharge area and include unsuccessful reproduction from alteration of flood cycles within the aquifer, degradation of groundwater, drainage and hydrologic alterations, and lower ground water levels (USFWS 1990). Planned industrial development is specifically listed as a threat to the recharge area of Key Cave, the critical habitat of *S. poulsoni*, both in the Recovery Plan and the initial listing of *S. poulsoni* as endangered and threatened wildlife (USFWS 1977; 1990). Additionally, the Recovery Plan states that survival of *S. poulsoni* is totally dependent on this recharge area, and adverse impacts must be eliminated if the species is to survive (USFWS 1990).

A recent study has examined the abundance, habitat preference, and distribution of *S. poulsoni* within Key Cave and surveyed numerous caves in the vicinity of Key Cave to search for other populations of *S. poulsoni* (Kuhajda and Mayden 2001). Results indicated that Alabama cavefish were present in five different pools within Key Cave and numbers observed were comparable to previous surveys in the 1970's and 1980's. However, the abundance of this species was extremely low, and no more than 12 individuals were seen on any one visit. Three size classes were present, indicating that recruitment is occurring. No other populations of *S. poulsoni* were discovered in surveys of neighboring caves, confirming that Key Cave is the only habitat worldwide for the Alabama cavefish. Although the Key Cave National Wildlife Refuge was established in a portion of the high recharge area of Key Cave, it was noted that threats to the groundwater continue from urbanization advancing into the remaining recharge area.

There is currently a proposal to construct the Eddie Frost Commerce Park west of Florence, Alabama (Price et al. 2001) (Figure 1). This proposed industrial park would be situated in part of the recharge area of Key Cave (Kidd et al. 2001). The construction of this industrial park would be in direct conflict with the stated objective of the Recovery Plan (USFWS 1982; 1985; 1990), which is to eliminate adverse impacts from the recharge area of Key Cave. Several issues regarding the location of this proposed industrial park and the negative affects it will have on the critical habitat of *S. poulsoni*, the Key Cave aquifer, are addressed below.

Alterations to Key Cave aquifer

An impermeable surface is planned on the 300 acres of the proposed industrial park plus additional impermeable surfaces from roadways serving the park and other associated development in the immediate vicinity. It is proposed that water from these surfaces will be diverted out of the recharge area of Key Cave and into the Cypress Creek drainage area to the southeast to prevent any contaminants in the runoff from entering the Key Cave aquifer (Price et al. 2001) (Figure 1). Reduced input of surface water in the recharge area of Key Cave could have a drastic impact on successful reproduction of *S. poulsoni*. Cavefish do not experience the typical seasonal cues (large change in water temperature and/or photoperiod) that other fishes use to co-

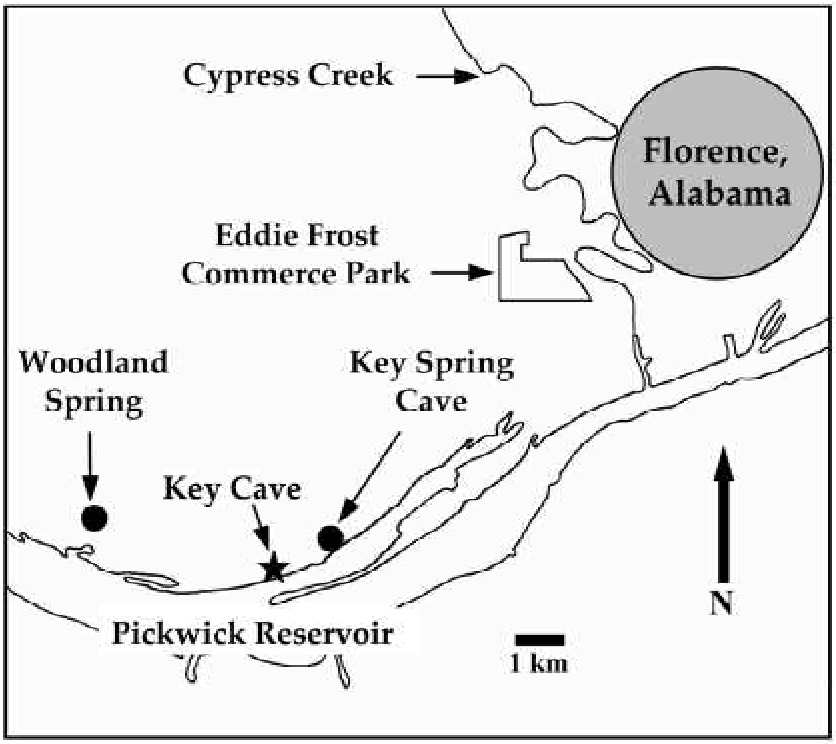

Figure 1. Recharge area for Key Cave and area near the western edge of Florence, Lauderdale County, Alabama.

ordinate maturation of gonads for synchronized spawning. The "seasonal" cue for cavefish is flooding of caves in the winter and spring which bring an increase in flow and a small temperature change. These are the proximate cues that signal hormonal changes and begin the maturation of gonads. The increase in nutrients associated with the higher flow gives the cavefish the food necessary to realize the production of gametes (Poulson 1963; Poulson and White 1969; USFWS 1982; 1985; 1990). Because population sizes of most cavefish, and *S. poulsoni* in particular, are extremely small, the need for synchronized spawning to assure that each sex is ready to spawn at the same time is paramount. Removal of surface waters from the recharge area of Key Cave will lower the amount of water entering Key Cave, and this may in turn disrupt the environmental cue necessary for successful reproduction. Only a portion of the adult population of cavefish spawns every year and their fecundity is quite low compared to other fishes (Poulson 1963). This compounds the threat that removal of water from the Key Cave recharge area will have on the spawning success of *S. poulsoni*. Because *S. poulsoni* is the most highly cave adapted amblyopsid in the world (Hobbs 1992; Poulson 2001), the life history traits of a small population size, infrequent reproduction, and low fecundity are amplified. This makes *S. poulsoni* even more susceptible to extirpation compared to other cavefish species (Noltie and Wicks 2001).

The proposed industrial park will also have negative impacts on the two other factors recognized by USFWS (1982; 1985; 1990) as likely to cause the decline of *S. poulsoni*, 1) drainage and hydrologic alterations and 2) lower ground water levels. The construction of the industrial park will create impenetrable surfaces, divert stormwater out of the Key Cave aquifer, and alter surface drainage patterns (Price et al. 2001). Direct removal of water from the Key Cave aquifer is also a potential threat. Waters pumped to the surface for industrial use have resulted in the lowering of water tables (Hobbs 1992). *Speoplatyrhinus poulsoni* lives in a zone of seasonal oscillation of the water table, and pools and lakes formed during high water become isolated during the dry season (Trajano 2001). Alabama cavefish in Key Cave become isolated in these pools. With drainage alterations and lower ground water levels, these isolated pools would be at a much larger risk of degradation (low oxygen levels, concentration of contaminants) or of completely drying out during the dry season. This would directly lead to the death of Alabama cavefish.

Concerns over the reduced input of water into the Key Cave recharge area resulting from the proposed industrial park are amplified when considering the reduction of water input that is already present in the recharge area. The native hardwood forest that once covered the recharge area has been replaced with agricultural land. During a rainfall event upon bare soil prominent in agricultural settings, compaction of the upper layer of soil occurs rapidly, and water is not as readily absorbed. This allows for more evaporation of rainwater than would have occurred if the ground was in a more natural, absorbent state (Hudson 1981). This decrease in absorbed water leads to less water entering the Key Cave aquifer, which can affect reproduction of *S. poulsoni* and can lead to inhospitable habitat within Key Cave during the dry season.

The biggest impact on water flow through Key Cave has been the replacement of the Tennessee River by the Pickwick Reservoir in 1938. Historically the outflow of Key Cave was Collier Spring, but this outflow is now almost completely under the reservoir, with only a small part of the former spring complex, Sometimes Spring, still visibly flowing close to the shores of the reservoir just downhill from the mouth of Key Cave. The submersion of Collier Spring decreased its outflow with the raised head elevation caused by the reservoir, thus increasing the flow of Woodland Spring (Figure 1). In concert with these changes in spring outflow, there is a change in flow of recharge water; much of the water that flowed through Key Cave now flows through the Woodland Spring aquifer, and the flows within Key Cave are reduced (Aley 1990). A misconception is that water from Pickwick Reservoir flows into Key Cave and has actually increased the flow of water in the aquifer. This is seldom, if ever, the case (Aley 1990). The reduction in flow within Key Cave may have already depressed the population size of this species through lower reproduction. Any further decrease in flow threatens the continued existence of this species.

The above-mentioned alterations currently present within the recharge area of Key Cave have diminished the natural flow of waters through the Key Cave aquifer. These adverse impacts must be eliminated if *S. poulsoni* is to survive (USFWS 1990), and any additional alterations to this aquifer, such as the proposed industrial park, may cause the demise of this species. For possibly hundreds of thousands of years *S. poulsoni* has evolved in the predictable environment of Key Cave. Because cave ecosystems are stable relative to surface habitats, minor disruptions can have severe consequences to its inhabitants. Additionally, the smaller a population, the less genetic diversity there is within that population to adapt to environmental change. These factors make *S. poulsoni* more susceptible to extinction than any other cavefish species (Noltie and Wicks 2001). All of these factors contribute to the absolute necessity for maintaining stability within the recharge area of Key Cave in the short-term and returning the recharge area to a healthy ecosystem in the long-term.

Additional aspects of Alabama cavefish biology

Cavefish are long-lived and do not reproduce annually, therefore decreased recruitment within a population due to changes in the cave's environment will not be immediately realized. Again, because *S. poulsoni* is so highly cave adapted, a long delay in observing a decreased population size is expected. The apparent stability of *S. poulsoni* over the last three decades does not necessarily mean that recruitment is stable. The Alabama cavefish may have been more plentiful in pre-settlement times. Today it is one of the rarest vertebrates in the United States, perhaps because of changes within the recharge area and aquifer of Key Cave over the last 100 years. Further alteration of the Key Cave recharge area may lead to the extinction of *S. poulsoni* (USFWS 1982; 1985; 1990).

There have been unfounded statements that *S. poulsoni* prefers a flow slower than what naturally occurs within Key Cave (Aley 1990). This has been attributed to the weak swimming abilities of *S. poulsoni* and its absence in Key Spring Cave, which shares the same aquifer as Key Cave but has higher flows (Figure 1). First, no swimming or flow-preference tests have ever been performed on *S. poulsoni,* so the assumption that it is a weak swimmer is pure speculation. Second, numerous epigean fishes, including piscivores, have been observed within Key Spring Cave (Kuhajda and Mayden 2001). There are more factors than just an increased water flow that contribute to the absence of *S. poulsoni* in this cave; cavefish are not adapted for an environment with predators (Poulson 1963). All scientific evidence indicates that the natural hydrology of the Key Cave aquifer must be kept as intact as possible for the continued existence of the endangered Alabama cavefish.

Pollution of Key Cave aquifer

The purpose of diverting stormwater out of the recharge area of Key Cave and into the Cypress Creek drainage area is to prevent any contaminants in the runoff of the industrial site from entering the Key Cave aquifer. The USFWS, in its Recovery Plans and initial listing for *S. poulsoni* (1977; 1982; 1985; 1990), has listed groundwater contamination as one of the factors likely to cause the decline of this species. Environmental pollution is a major threat for all species of hypogean (cave dwelling) fishes, with contamination from industrial runoff of particular concern (Proudlove 2001). Organic pollutants can increase nutrients in a cave ecosystem where low-nutrient conditions naturally occur; this can severely alter the natural food web in the cave system (Notenboom et al. 1994). Certain invertebrates can be eradicated from cave systems with introduced nutrients (Simon and Buikema 1997), which would have a direct impact on cave predators such as *S. poulsoni*. Increased nutrients can also cause an increase in pathogens (Brown et al 1998) and a decrease in oxygen (Marie and Pomel 1994; Simon and Buikema 1997). Toxins introduced into groundwater can become concentrated in the clays of caves and in traps within karst systems (Marie and Pomel 1994) and karst conduits can concentrate surface pollutants from non-point sources (Pasquarell and Boyer 1996). All of these factors put cave organisms at great risk with even low-level pollution. Bioaccumulation within cave organisms from even low levels of toxins is manifested due to their great longevity relative to surface organisms (Dickson et al. 1979; Hobbs 1992). The risk is even greater for cavefish, including *S. poulsoni*, because they are top-end predators. Chronic contamination of aquifers with toxins will cause accumulation through bio-concentration in these predators, and can lead to their loss from the community (USFWS 1982; 1985; 1990).

Few studies have examined the effects of organic and chemical pollution on groundwater ecosystems and cave organisms (Notenboom et al. 1994), but evidence exists for the direct effects pollution has on cavefish. Brown et al (1998) reported a 30% decline in a population of *Amblyopsis rosae* attributable to increased inorganic and organic compounds in the cave system. Pesticides were identified as the cause for “broken back syndrome” in a population of *A. spelaea* (Keith and Poulson 1981), and this condition may be widespread. Death of cavefish and cave sculpins has been attributed to acute pollution, and populations have been extirpated from entire cave systems due to chronic pollution (Quinlin 1983; Burr et al. 2001; Proudlove 2001). Pollution of groundwater can also be in the form of thermally altered runoff, especially from industrial enterprises (USFWS 1982). Thermal variation could alter the reproductive cycle of *S. poulsoni*, either eliminating recruitment, or producing offspring when environmental factors within the cave are not optimal for their growth and development (USFWS 1982; 1985). The threat of pollutants being introduced into the Key Cave aquifer comes not only from the proposed industrial park itself, but also from likely unrestricted growth surrounding the industrial park and from potential chemical spills on adjoining roadways.

Key Cave recharge area boundaries

Although a previous hydrogeologic study had determined that the proposed industrial park was within the recharge area of Key Cave (Aley 1990), a new study was completed in 2001 by the U.S. Geological Survey (USGS) (Kidd et al.). This study performed dye-tracer tests at three sites within or near the proposed Eddie Frost Commerce Park. Dye injected in well I-1, located just north of the proposed industrial park’s eastern half, was detected at only one site in Cypress Creek. Dye in well I-2, located in the northwest corner of the proposed park, was not detected at any sites. Dye from well I-3, located in the center of the western half of the proposed park, was detected at two sites in Key Cave and several other sites. There is cause for concern with several findings of this report and how these dye studies demonstrate that any development on this proposed industrial site could have a direct negative impact on Key Cave aquifer.

First, dye injected into well I-3 was not only recovered at numerous dye detection sites within the Tennessee River proper watershed, but also at a site well within the Cypress Creek watershed. If dyes (and pollutants) introduced into the environment within the Tennessee River watershed (as determined by the surface drainage divide) are later found within both watersheds, then conversely pollutants introduced within the Cypress Creek watershed will eventually be found within the Tennessee River watershed, and the Key Cave aquifer. In fact, it is common for sites in karst regions to contribute water to multiple aquifers (Aley 1990). It is obvious from the results of the dye-tracing test from well I-3 that the surface drainage divide is meaningless with regards to subsurface water flow, and that pollutants introduced into either watershed can end up in both watersheds.

Second, the concern of pollutants entering multiple aquifers is amplified by the relative flat nature of most of the proposed industrial park area (Price et al. 2001), further rendering any surface drainage divide meaningless. More dye-tracing tests need to be performed within the actual boundaries of the proposed industrial park, specifically within the Cypress Creek watershed, to determine the full eastward extent of the Tennessee River and Key Cave recharge areas. Only one well was injected with dye in the Cypress Creek surface drainage (I-1), and it was north of the proposed industrial park. The need for additional dye-trace tests is especially critical for the area of the two proposed detention ponds, which serve to store stormwater runoff and associated contaminants, and will be located on the south side of the park. This area is the most distant point in the park within the Cypress Creek watershed from test well I-1; therefore this area would be the least likely to have the same hydrogeography. Any industrial activity within any part of the proposed industrial park site poses a risk to the Key Cave aquifer through introduction of pollutants.

A third concern deals with construction of the industrial park and its roadways. Because the proposed site lies within a karst region, these construction activities could have disastrous effect on the Key Cave aquifer and *S. poulsoni*. Hobbs (1992) outlines hazards associated with heavy structural loading and urban development in areas with subterranean cavities. These include the formation of sinks (many in Alabama), large quantities of clay and silt being washed into caves due to highway construction], and recorded fish poisoning when diesel fuels were spilled at a construction site. Reduced flows of groundwater have also resulted from construction activities. Any one of these adverse affects could jeopardize the continued existence of *S. poulsoni*.

It is clear that there is imminent danger of contaminants from the Eddie Frost Commerce Park being introduced into the Key Cave recharge area regardless of any proposed engineering plans. All monitoring of the effectiveness of these plans to keep pollutants out of the Key Cave recharge area is meaningless. One acute event, even if detected by the proposed monitoring stations, could not be stopped from reaching Key Cave. Such an event would likely result in the death of individuals of *S. poulsoni*; acute pollution has resulted in the death of other hypogean fishes (Burr et al. 2001, Proudlove 2001). Because cave organisms in general have slow dispersal rates, the effect of a single pollution event could be irreversible (Strayer 1994) and, in the case of *S. poulsoni*, could cause its extinction. Even if all pollutants from the proposed industrial site were somehow diverted into the Cypress Creek watershed based on the surface drainage divide, there is little hydrogeographic evidence to show that these contaminants would be restricted to Cypress Creek (Kidd et al. 2001. In addition, if all pollutants were somehow successfully diverted out of the Key Cave recharge area, the removal of water could severely impact many aspects of the life history and threaten the continued existence of *S. poulsoni* as discussed above.

Summary

Speoplatyrhinus poulsoni, the Alabama cavefish, is restricted to a single cave system, Key Cave, and vigilant protection of this cave and its recharge area is paramount to the continued existence of this endangered species. The USFWS (1982; 1985; 1990) has developed three versions of a Recovery Plan for protecting and improving the critical habitat of this species, the Key Cave aquifer and its recharge area. The official position of the Service is outlined in these Recovery Plans, which state that factors likely to cause a decline of *S. poulsoni* include unsuccessful reproduction from alteration of flood cycles within the aquifer, degradation of groundwater, drainage and hydrologic alterations, and lower ground water levels. It is my opinion that the proposed Eddie Frost Commerce Park would have negative impacts on each and every one of these factors listed in the Recovery Plan for *S. poulsoni* (USFWS 1982; 1985; 1990). There is currently a Memorandum of Understanding and Conservation Plan between the USFWS and the City of Florence which attempts to minimize potential for groundwater contamination. Various measures include restricting tenants of the park to "clean" industry and the implementation of numerous monitoring activities. But even with these agreements in place, the impacts on the Key Cave recharge area from the proposed industrial park will likely disturb the critical habitat of *S. poulsoni* and may cause its extinction.

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Camera trapping *Priodontes maximus* in the dry forests of Santa Cruz, Bolivia.

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Abstract

During systematic camera trapping surveys conducted for jaguars, we collected photographs of giant armadillos at three of four dry forest (Chaco and Chiquitano) sites surveyed in eastern lowland Bolivia, thus extending the documented distribution of the species. The cumulative 30 camera trap records with time information suggest a highly nocturnal activity pattern. We identified individuals according to the distinct scale patterns, particularly the dividing line between dark and light scales on the carapace and hind legs. We estimated crude densities, ranging from 1-16 individuals/100 km^2 across sites and surveys, by dividing the number of individuals by the area enclosed by the camera traps. At one site the number of captures and recaptures was sufficient to estimate abundance using the software Capture, together with a survey area that includes a buffer area around the camera traps equivalent to half the mean maximum distance covered by individual animals observed at more than one camera trap location. Together these estimates suggest a population density of 5.77-6.28/100 km^2 for this site. Given the vast area (11,500 km^2) of similar habitat protected within the Kaa-Iya National Park, and preliminary evidence of the species in neighboring protected areas, the dry forests of eastern Santa Cruz may offer a unique stronghold for the long-term conservation of the species.

Resumen

A través de estudios sistemáticos de uso de trampas de cámaras por jaguares, hemos recogido fotos de armadillos grandes en tres de los cuatro sitios de los bosques secos (Chaco y Chiquitano) estudiados en la tierra baja oriental de Bolivia, aumentando así la distribución documentada de la especie. Los 30 archivos acumulados de las trampas de cámara, que incluyen información sobre la hora del día cuando fueron utilizadas, sugieren un patrón de actividad nocturna muy alto. Hemos identificado individuos de conformidad con las distintas configuraciones de las escamas, en particular, la línea divisoria entre las escamas oscuras y claras en el caparazón y patas posteriores. Estimamos las densidades brutas, que van de 1 a 16 individuos/100 km^2 a través de los sitios y estudios, dividiendo el número de individuos entre el área cubierta por las trampas de cámara. En uno de los sitios, el numero de individuos capturados y recapturados fue suficiente para estimar abundancia, usando el programa CAPTURE; y considerando una zona "buffer" alrededor las trampas de cámara equivalente a la mitad de la distancia máxima promedio cubierta por los animales observados en mas de una de las trampas de cámara. Como resultado, estimamos una densidad de población de 5.77 – 6.28 / 100 km^2 para este sitio. Tomando en cuenta la enorme zona (11,500 km^2) de hábitat protegido dentro del Parque Nacional Kaa-Iya, y la evidencia preliminar de especies en áreas protegidas cercanas, los bosques secos de Santa Cruz oriental pueden ofrecer un refugio único para la conservación de largo tiempo de esta especie.

Introduction

The giant armadillo, *Priodontes maximus,* is considered to be rare throughout its geographic distribution, and endangered (EN A1cd, IUCN 2002; Appendix I, CITES 2003) from continued habitat conversion and hunting. Its scarcity, combined with nocturnal and largely fossorial habits, have made it a difficult species to study in the field. It is known to live in a variety of Neotropical lowland habitats, from humid to dry forests and grasslands, and to feed on ants and termites, often destroying their large nests (Eisenberg and Redford 1999; Emmons and Feer 1999). Knowledge about its ecology derives mainly from indirect signs, sporadic sightings or dead animals, while its abundance and ranging behavior are largely unknown.

As a result of an intensive camera trapping study focused on jaguars, we obtained a number of automatically-triggered pictures of giant armadillos from the dry forests of Santa Cruz, Bolivia. In this article we describe the use of camera traps, present novel trapping data on giant armadillos, and discuss their contribution to the knowledge of activity patterns, abundance, ranging behavior, distribution and conservation status of this species.

Study area

1. Kaa-Iya del Gran Chaco National Park (see Figure 1): This 34,400 km^2 protected area covers the northern end of the Gran Chaco, including four principal landscape systems (Navarro and Fuentes 1999). The two purely Chacoan forest landscape systems are the Chaco alluvial plain forest (13,800 km^2) and the Chaco riverine forest (500 km^2). The two other landscape systems are transitional forests: the Chaco transitional landscape system (9,100 km^2) and the Chiquitano transitional landscape system (11,500 km^2).

1.a. During 1997, we established a field camp at Cerro Cortado (19° 31.60' S, 61° 18.60' W) in the Chaco alluvial plain landscape system, on the border between the Kaa-Iya National Park and the adjacent Izoceño indigenous territory. Annual precipitation at the site averages 500 mm. During the 6-8 month dry season, surface water disappears for extended periods. A single road runs through the study site, unused for over a decade until we reopened it to establish our research camp. We opened a grid of 2-4 km study trails off of the road. The area is not subject to hunting or livestock pressure.

1.b. Towards the northern end of the Chiquitano transitional landscape system we established a field camp in 2001 at Tucavaca (18° 30.97' S, 60° 48.62' W), on the Bolivia-Brazil gas pipeline and 85 km south of the town of San José de Chiquitos. Annual precipitation at the site averages 800 mm. During the six month dry season, surface water disappears for extended periods. Existing roads include the gas pipeline itself (30 m-wide right-of-way, with a 3-6 m-wide road to one side or in the center), a gravel road north to San José, and an overgrown road south to Paraguay. We opened a square grid of 5 km study trails, enclosing a 100 km^2 study area centered on the field camp and the gas pipeline, and added in 2002 an inlaid 2 km x 2 km grid of trails. Scrub patches remain where the forest was burned roughly 30 years ago, and the area is not subject to hunting or livestock pressure.

1.c. Also in 2001, we established a second field camp towards the southern end of the same landscape system at Ravelo (19° 17.72' S, 60° 37.23' W), near the Paraguayan border. Annual precipitation at the site averages an estimated 650 mm, but, unlike the previous site, water points (springs, lagoons) persist year-round in all but the driest years. The springs cluster around the Cerro San Miguel that rises 500 m above the surrounding plain, while a salt pan also lies within the study area. A single road crosses the area, from the city of Roboré to the northeast, passing through Ravelo military outpost, on to Paraguay. Several overgrown roads also exist, unused for over 10 years: one leads west to the large salt pans within the Kaa-Iya National Park and from there north to Tucavaca and San José, others were opened in a grid of oil exploration lines. We reopened several of these roads as footpaths/study trails, as well as cutting additional new study trails 3-5 km long to cover the study area. The dozen soldiers at the Ravelo military outpost maintain a small number of cattle (30) and several donkeys, while the nearest cattle ranch 15 km to the southeast at Palmar de las Islas maintains roughly 300 cattle. Livestock is not fenced in and therefore strays between Ravelo and Palmar along the main road.

2. The San Miguelito Private Reserve comprises approximately 25 km^2 within a

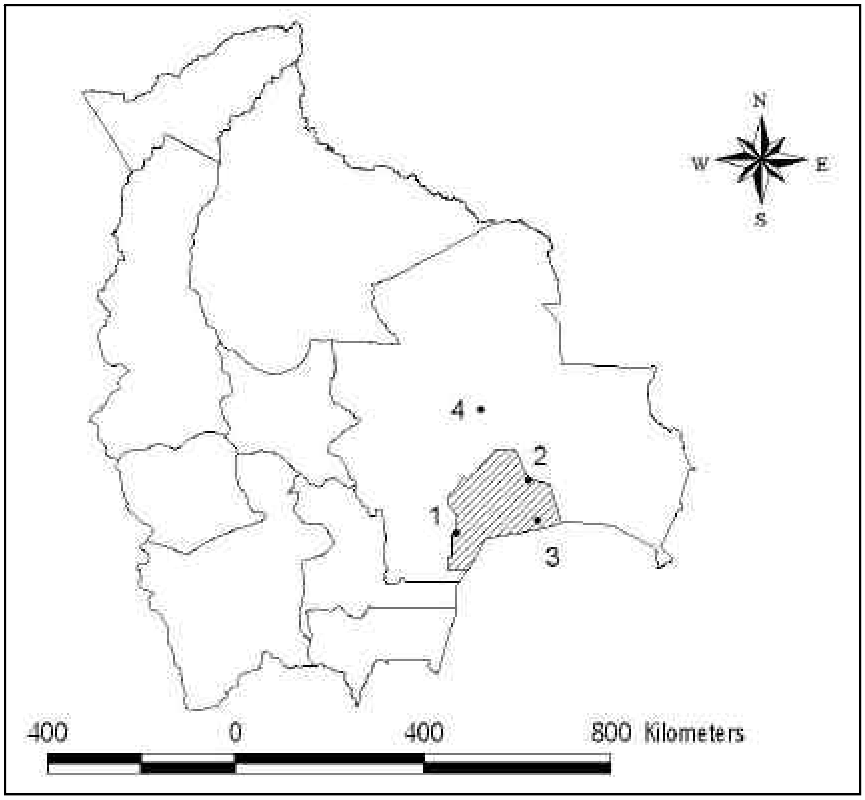

Figure 1. Dry forest protected areas (A= Kaa-Iya del Gran Chaco National Park, B=Otuquis National Park, C=San Matías Integrated Management Area, D=Valle Tucavaca Municipal Reserve) and study sites (1=Cerro Cortado, 2=Tucavaca, 3=Ravelo, 4=San Miguelito Private Reserve) in southern Santa Cruz Department, Bolivia.

400 km^2 cattle ranching property 200 km to the east of Santa Cruz, and north of the Kaa-Iya National Park (17° 05.52' S, 61° 47.32' W). The landscape system is Chiquitano dry forest, with an average annual rainfall from 1000 to 1500 mm, and vegetation types ranging from sub-humid forest to 'cerrado' woodland (Rumiz et al. 2002). Cattle ranching is the principal economic activity outside the private reserve itself, with patches of forest cleared for pasture. The ranch maintains a system of roads through the reserve, in addition to which we opened a number of study trails 1-3 km in length. A small river runs through the private reserve, and several permanent springs along the 200 m escarpment running parallel to the river, as well as artificial ponds, provide surface water for wildlife.

Methods

The methodology at all sites consisted of a systematic camera trap survey, whose primary objective was to survey jaguars (*Panthera onca*) and to estimate their population densities (Maffei et al. 2002, in press, under review, Silver et al. in press). We used Camtrakker® and Trailmaster® camera traps with passive and active infrared detection systems. Sensors detect movement or changes in temperature from the presence of an animal, and activate the camera trigger. The cameras automatically recorded the date and hour on each photograph. Cameras were active continuously (24 hours a day), although a few were reprogrammed to avoid self-firing during the hottest hours of the day. We set them in pairs facing each other across a trail/road in order to simultaneously photograph both sides of any animal passing between them along the trail/road, with a distance of 1-2 km between camera sets. They were checked for correct functioning every 3-7 days, and reloaded with film (ASA 200 or 100) or batteries if necessary. All stations with at least one active camera per night were added up to calculate sampling effort in 'trap-nights'.

At Tucavaca, during eight months (May-December, 2001), we rotated 12 camera traps among sites on the study trails and the gas pipeline, for a total of 2,520 trap-nights. During an intensive 60-day survey period (19 January-20 March 2002), we installed 32 pairs of camera traps on the same study trails and pipeline road, for a total of 1,920 trap-nights. We conducted a second intensive 60-day survey from 12 April-12 June 2003 with 26 pairs of camera traps using the western half of the 10 x 10 km grid, the inlaid 2 x 2 km grid of trails, and the pipeline road, for a total of 1,560 trap nights.

In four 50 x 30 m grids where fresh giant armadillo burrows and tracks were observed, we set an additional seven cameras during three months (April-June 2002) at the burrows and along the trails themselves, for a total of 654 trap nights. Following the second intensive survey, we have continued monitoring trails with these seven camera traps (270 trap nights to August 2003).

At Ravelo, we also rotated 14 camera traps among sites on the study trails and at seasonal ponds, for a total of 1,248 trap-nights. During an intensive 60-day survey period (10 February-10 April 2003), we installed 37 pairs of camera traps on the same study trails, as well as the main road, at seasonal ponds, and a salt pan, for a total of 2,170 trap-nights. We conducted a second intensive survey using 22 pairs of camera traps across the same area (September-November 2003), for a total of 1,320 trap-nights.

At San Miguelito we conducted an intensive 60-day survey (20 September-20 November 2002), installing 22 pairs of camera traps on existing roads and study trails. We installed four pairs of cameras along the edge of the river, one pair at a salt lick, and one pair at a spring. Trapping effort totaled 1,695 trap nights.

We registered the date and camera trap location on each roll of exposed film and processed film in batches. All relevant pictures were scanned and printed, associating in pairs photographs taken by paired cameras of each single event (same animal, date, hour, and location). To calculate capture frequency we divided the number of captures by the sampling effort with the result expressed as captures per 1,000 trap-nights.

Individual animals were identified by looking at paired photos to detect particular features or marks and by comparing them with photos from other locations and dates. The minimum number of different individuals per site was determined for each survey period and used to estimate abundance. A crude estimate of density was obtained by dividing this number of individuals by the area enclosed by the camera traps.

A more elaborated analysis was performed with the program CAPTURE (Rexstad and Burnham 1991) for the single site with sufficient observations. Assuming a closed population during each sampling period, we estimated abundance based on captures and recaptures of identified individuals. In order to estimate the population density we divide the estimated abundance by the effective sample area. The effective sample area included a circular buffer around each station, whose radius was half the mean maximum distance among multiple captures of individuals at the site.

Results

Camera trap records and activity patterns

After a total effort of over 17,000 trap-nights, distributed across four sites (Table 1), we obtained 33 records of *Priodontes maximus*. As previous hunter reports andlack of animal signs had suggested, no record of *Priodontes* was obtained in Cerro Cortado. Cumulatively, Tucavaca rendered 29 records, San Miguelito 3, and Ravelo 1. The highest capture frequencies come from Tucavaca, but are only 4-6 per 1,000 trap nights. Combining records across sites, camera trap records indicate that giant armadillo activity patterns are decidedly nocturnal, beginning at 22:00, with only one day-time record (3%) to date (Figure 2).

Site	Trap nights	Area (km^2)	Captures	Individuals	Capt/ 1000 t-n	*Ind/ 100 km^2	Months
Cerro I	2280	49	0	0	0,00	0,00	2
Cerro II	1680	49	0	0	0,00	0,00	2
Tucavaca Pilot	2520	50	10	8	3,97	16,00	8
Tucavaca I	1920	130	10	6	5,21	4,61	2
Tucavaca II	1560	60	7	4	4,49	6,67	2
Tucavaca Burrows	654	20	1	1	1,53	5,00	4
Tucavaca Other	270	8	1	1	3,70	12,50	1
Ravelo Pilot	1248	40	0	0	0,00	0,00	8
Ravelo I	2160	102	1	1	0,46	0,98	2
Ravelo II	1320	101	0	0	0,00	0,00	2
San Miguelito	1695	24	3	2	1,77	8,33	2
TOTAL	17037		33	22			

Table 1. Camera trapping effort and records of Priodontes maximus at four dry forest sites in Santa Cruz Department, Bolivia. Note: *Density estimates are crude, based on individuals observed and survey area enclosed by camera traps.

Individual identification and abundance

The unique scale patterns permitted the identification of individual animals, while the genitalia were infrequently shown and did not permit us to sex every individual. The dividing line between dark and light scales on the carapace and on the hind legs was particularly noteworthy, as was the number of light scales per row from the lower edge of the carapace up to the dividing line (Figure 3). Based on these differences, we identified eighteen individuals at Tucavaca (8 males, 1 female, and 9 unsexed), and two unsexed individuals at San Miguelito. The sole individual photographed at Ravelo was a female.

Comparing the number of individuals identified during each survey period with the area enclosed by the camera traps, we estimate crude densities of 5-7 animals/100 km^2 at Tucavaca (Table 1). Though fewer individuals were observed at San Miguelito, the smaller survey area suggests a slightly higher population density of 8 animals/100 km^2.

Combining all Tucavaca records during 28 months, we estimate a minimum abundance of 18 different individuals in a study area of 130 km^2, suggesting a density 14/100 km^2 if all animals survived and remained in the area. However, most individuals were observed on a single occasion, with only four individuals observed 2-3 times each over 1-3 month periods, and a single animal observed four times over a 7-month period.

Ranging patterns and density estimation with CAPTURE

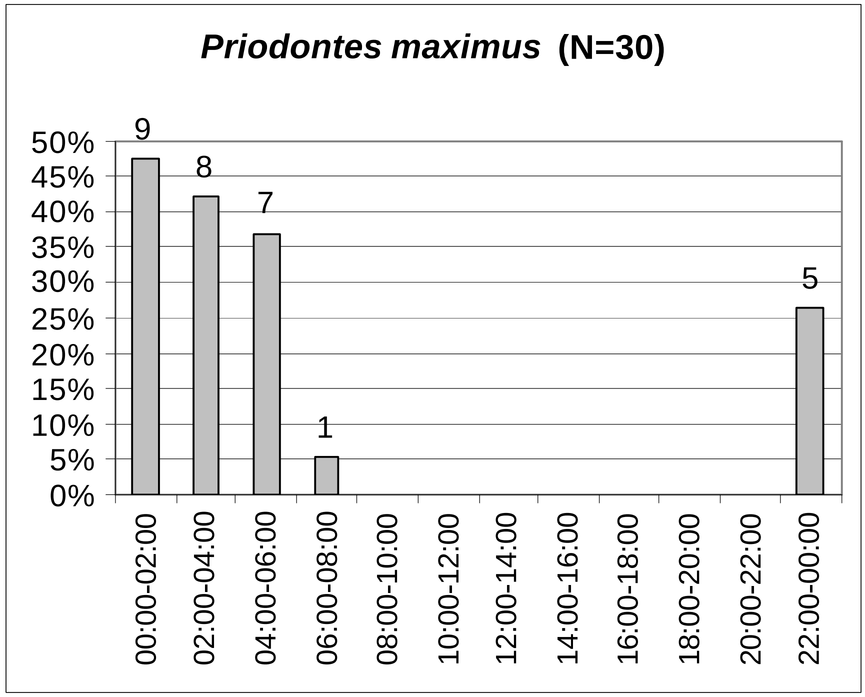

Figure 2. Activity patterns from observations at all three sites (Tucavaca = 26, San Miguelito = 3, Ravelo = 1).

Figure 3a. Two photographs of the same individual, 101M, taken 7.5 km and 7 months apart. Photos courtesy of Andrew Noss

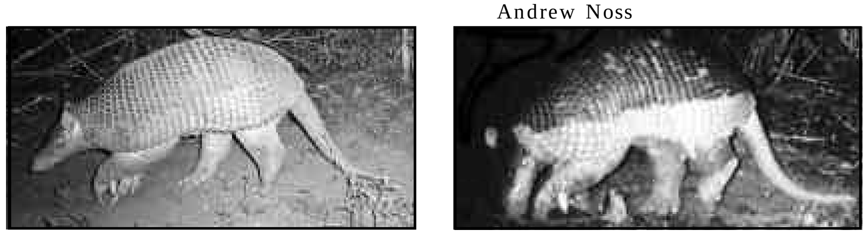

Figure 3b. Two other individuals, 103 and 110. Photos courtesy of Andrew Noss.

The small number of individuals "captured" (N=6) and "recaptured" (N=2) during the first systematic survey at Tucavaca was sufficient to estimate a population abundance of 12 (± 20.83) individuals using the program CAPTURE's heterogeneity model M(h). As a proxy for home range, which is unknown, we used the mean maximum observed distance (3.73 km, range 0.7-7.5 km), for individuals observed at more than one location during the entire camera trapping period (N=4). We divided this average in half in order to estimate a 1.86 buffer around camera trapping points during the systematic survey, and in turn to determine a discontinuous survey area of 191 km^2 (Figure 4). Finally, to estimate population density, we divided the population abundance by the survey area, obtaining a value of 6.28/100 km^2 (SE ± 2.76).

During the second systematic survey at Tucavaca we recorded only four individuals with three recaptures. The estimated population abundance using CAPTURE M(h) is 6 (± 2.12) individuals, though the data are ill-conditioned. Only one individual was photographed at two sites, separated by a distance of 2.3 km. Combining the mean maximum distance with those from the previous survey, we estimate a buffer of 1.72 km (N=5), and a nearly continuous survey area of 104 km^2. The resulting density estimate from this second survey is 5.77/100 km^2 (SE ± 1.95).

The small number of captures at San Miguelito does not permit an abundance estimate using CAPTURE. For the single individual observed twice, the distance between observations was 1.34 km. Applying instead the more conservative 1.22 km average buffer observed at Tucavaca to San Miguelito, we have an approximate survey area of 48 km^2 with two observed individuals, and a tentatively estimated population density around 4 animals/100 km^2.

Multiple records of six individuals at Tucavaca showed maximum movements of 2-7.5 km for three males, 1 km for a female, and 0.7 km for an unsexed individual. For the single (unsexed) individual observed twice at San Miguelito, the distance between observations was 1.34 km. For two male individuals at Tucavaca, we are also able to estimate minimum observed home ranges based on the minimum convex polygon uniting the points (N=3) where each animal was recorded: 3 km^2 for the first animal from February-March 2002, and 15 km^2 for the second animal between August 2001 and March 2002 (Figure 4). Within this range we observed at least two other individuals during the same 7-month period. Out of the 20 locations where we observed giant armadillos, three registered multiple individuals: a maximum of three individuals during a two-month survey period, and four individuals over 28 months. These cases included spatial overlap among 2-3 males,as well as between a male and a female individual.

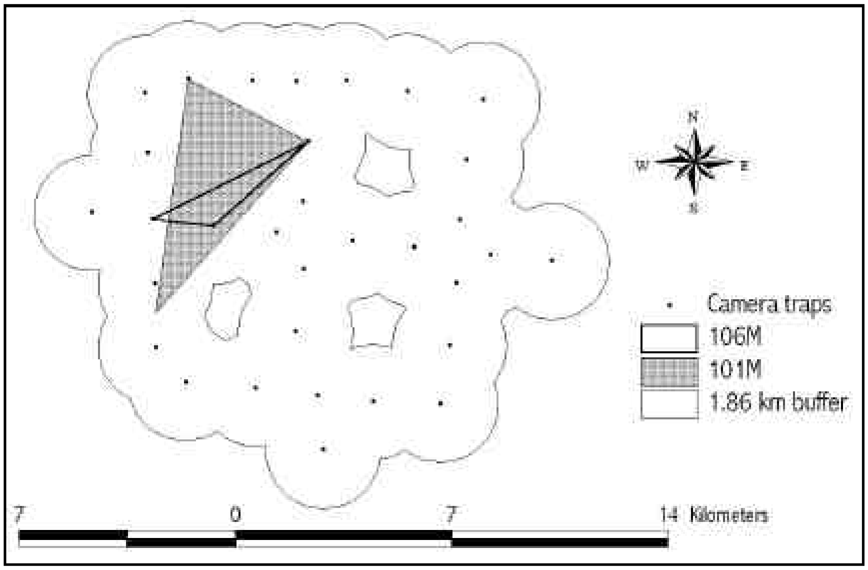

Figure 4. Camera trap array, survey area Tucavaca I, and minimum observed range of two individual males over a 7-month period.

These multiple observations, and indeed all observations of giant armadillos at Tucavaca, occurred both on study trails as well as on the gas pipeline right-of-way road. No animals were recorded revisiting open burrows. At San Miguelito and Ravelo, all observations occurred on roads, with none at camera traps located on study trails, at springs, or salt licks.

Discussion

Camera trapping as a method to study giant armadillos

Camera trapping provides a new method for monitoring *Priodontes maximus*, permitting the identification of individuals and a description of their ranging behavior, and in turn the estimation of population densities. Although the systematic sampling reported here was designed to survey jaguars (trap spacing 1-2 km, location on roads—Silver et al. in press; Maffei et al. in press), it succeeded in obtaining important records of the rare giant armadillo. The sampling effort was high (~15,000 trap nights), representing perhaps an exaggerated cost for a single species study, but a cost-efficient method if considered as part of a wider survey. The intent to increase capture probabilities by targeting areas where burrows and tracks had been observed failed, with only one photograph of a giant armadillo resulting through this effort. Ideally, density estimates should be derived from surveys designed according to species specific information at the particular site (like home range) to determine camera trap spacing and continuous survey areas, and behavior to identify high probability locations for camera placement. We therefore consider our results on giant armadillo densities to be preliminary and tentative. However, as for many species at many sites, we lacked detailed range and behavioral information on giant armadillos in dry forests.

With respect to activity patterns, the number of observations is low, but 24-hour monitoring with camera traps would appear to be an objective method for assessing activity patterns (unless the animals only use roads and study trails at certain periods), and the data are consistent with the nocturnal habits reported in the literature (Eisenberg and Redford 1999; Emmons and Feer 1999; Parera 2002).

Individual identification based on differences in scale patterns from photos proved reliable . Though even at Tucavaca, where we registered the most observations, the number of captures and recaptures was small during 60-day survey periods, it was sufficient to estimate a population abundance using CAPTURE. With a small number of individuals captured at multiple locations, the estimate of the buffer and the effective survey area is also tentative. Nevertheless, the similarity between the two density estimates at Tucavaca, based on surveys over a year apart, in dry versus wet seasons, and based on different camera trap lay-outs and survey areas, would appear to validate the method.

Capture frequencies per 1,000 trap-nights might also be used to estimate abundance, particularly when comparing systematic survey efforts, as they coincide more or less with the number of individuals observed per site and per survey effort. However, the area sampled would presumably affect the number of individuals observed, for example comparing San Miguelito with Tucavaca, and comparisons across sites should be made with caution. Trap location may influence capture success by species (Rumiz et al. 2003; Maffei et al. in press): studies with different target species may emphasize different trap locations, in turn affecting capture rates of both target and non-target species. More specifically, capture rates resulting from systematic sampling geared towards species with dissimilar habits (e.g. jaguar vs. smaller carnivores) or directed to species-specific resources (active burrows, fruiting trees, salt licks) should not be compared out of context. Moreover, the rarer the species, the greater the sampling effort necessary to obtain sufficient records for density estimates.

Camera trapping data on giant armadillo ecology

These records represent an important range extension for the species in Bolivia, compared to museum records compiled by Anderson (1997) and Eisenberg and Redford (1999), and hunting and track records (Wallace and Painter in press) from more humid lowland Amazonian forests to the north. Our camera trapping records confirm the species' presence in dry forests that include the Chiquitano forests as well as their transitional forms to Chaco forests. Our surveys also appear to confirm its absence in the driest Chaco alluvial plains at Cerro Cortado, with nearly 4,000 trap nights of effort surpassing the 1,000 trap nights that Carbone et al. (2001) estimate necessary to confirm absence. However, our single capture in over 4,700 trap nights at Ravelo suggests that animals occurring at very low densities may not be detected even with high levels of camera trapping effort, as indicated by Jennelle et al. (2002).

Giant armadillos evidently use both roads and trails, though only for short distances according to track observations. The limited information provided by camera trapping on ranging patterns and individual spacing suggests that several individuals may use the same area, not showing strict territoriality between males and females or among males. The maximum observed distance that a single individual traveled and the observed range described, 7.5 km and 15 km^2, are noteworthy and the first such records in Chaco or Chiquitano dry forests. If an individual may move 7.5 km as recorded, it is almost certain that its home range will overlap with those of other individuals, given the density estimates derived from systematic surveys of about 5-6 individuals per 100 km^2 in dry forests where they are not hunted. However, the small number of observations permits only tentative conclusions regarding giant armadillo ecology that require confirmation using alternative methods such as radio-telemetry.

Conservation status of giant armadillos in the dry forests of Santa Cruz

Priodontes maximus is known to occur in most of the protected areas from the humid lowlands and lower Andean slopes of Bolivia (Tarifa 1996). More recent hunting records and observations of animal sign suggest that it is still present in other natural areas of the Bolivian Amazon (e.g. Rumiz and Herrera 2000; Rumiz et al. 2001; Wallace and Painter in press). At the same time, local reports indicate that it is disappearing due to hunting and habitat conversion.

By confirming the presence of *Priodontes* populations in the dry forest landscapes of southern Santa Cruz, the known extension of protected habitat for this species increases considerably. Within the vast Kaa-Iya del Gran Chaco National Park, the Chiquitano transitional forest (as represented at the Tucavaca and Ravelo sites) covers approximately 11,500 km^2. In addition, the Chaco transitional landscape (not sampled within the Park, but similar to that of San Miguelito) represents about 9,100 km^2 of protected habitat. In comparison, *Priodontes* reports from the Argentine Chaco dry forest are limited to relatively small protected areas (86-1142 km^2) that do not offer sufficient protection in the long term (Porini 2001). Though reported from the Brazilian Cerrado (Silveira et al. 1999), its conservation status there is unknown.

Three other newly established protected areas in southern Santa Cruz (ANMI [Integrated Management Area] San Matias, AMNI Otuquis, and Reserva Municipal Valle de Tucavaca, Figure 1) would appear to protect giant armadillos according to observed field signs or unpublished reports of *Priodontes* (Museo de Historia Natural NKM mammal database). Although hunting and other human activities occur around and sometimes within these protected areas (Arispe and Rumiz 2002), the dry forest landscapes of Santa Cruz are likely to prove a unique stronghold for the long term survival of the species. We will continue to evaluate the conservation status of giant armadillos across additional sites in the protected areas of Santa Cruz and to monitor populations and individuals identified to date.

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News from Zoos

U.S. Hatched Andean Condors Return To Colombia

Four Andean condors recently returned to South America as part of a successful international collaborative program that reintroduces the endangered Andean condor to its native home range in Colombia. For more than 14 years, the Zoological Society of San Diego has been working with South American government agencies and conservation organizations to send Andean condors to both Colombia and Venezuela.

"The Andean condor program is international in scope and involves a number of U.S. zoos as well as federal and local agencies in South America," said Michael Mace, curator of birds for the San Diego Zoo's Wild Animal Park and the American Zoo and Aquarium Association's (AZA) Andean condor Species Survival Program (SSP) coordinator. "One of the most rewarding aspects for a zoo is to be able to participate in release programs where, as in this case, birds are specifically reared and returned to the wild."

One male and three female Andean condors from the San Diego Zoo, Los Angeles Zoo, Dallas Zoo and a condor offspring from a Cincinnati Zoo breeding pair on loan to the World Bird Sanctuary in Missouri were flown out of Miami and were to be received by CORPOBOYACA, a natural resources management agency, early Wednesday morning with the help of the Colombian federal agency Ministerio de Medioambiente de Colombia.

The Andean condor, found throughout the Andes from Colombia to Tierra del Fuego, Argentina, is threatened in its northern range and has become rare in Venezuela and Colombia. These two countries developed the reintroduction program to release captive-bred birds from conservation organizations such as North American zoos.

"The Los Angeles Zoo is proud to be part of the Andean condor Species Survival Program," said Susie Kasielke, Los Angeles Zoo curator of birds. "The young male Andean condor slated for release is the 33rd bird coming from the Los Angeles Zoo, 17 of which were hatched here," said Kasielke, who is also the Andean condor North American studbook keeper. "Nineteen facilities have produced eggs and chicks for release to the wild. The real measure of success is that some of the birds are now successfully reproducing in the wild."

The Zoological Society of San Diego has sent 30 Andean condors to South America from the San Diego Zoo and the Wild Animal Park over the last 10 years. Twenty-two of those birds were sent to Colombia. Three of the Andean condors sent to Colombia this month stayed at the Cincinnati Zoo's off-site breeding facility in preparation for transfer and release into the wild. After their arrival in Colombia, the birds will spend two weeks in a holding facility to acclimate to their new surroundings before their release into a wilderness area 50 miles northeast of Bogota.

Youth Art Aids Tree Kangaroo Conservation Program

Roger Williams Park Zoo, in collaboration with the Rhode Island Art Education Association, recently hosted the 3rd Annual Papua New Guinea Youth Art Exchange exhibit from 6 March through 31 April. The exhibit featured artwork and letters shared through the Papua New Guinea Art Exchange, a program facilitated by Rhode Island art teachers in conjunction with the Zoo's Tree Kangaroo Conservation Program (TKCP), which is based in Papua New Guinea (PNG).

Through the Exchange, local Rhode Island students and students from PNG learned about the similarities and differences in the culture, lifestyle, wildlife and conservation issues of the US and PNG. In beautiful postcards and drawings, Rhode Island children shared a slice of local life and PNG students responded with their versions of the same - stories of their "tsing-tsings" (culturally symbolic dances), descriptions of the mountains and rainforests, and pictures of tree kangaroos.

The goal of the exchange, according to Zoo Education Curator Robbie Fearn, is to create a universal understanding of environmental stewardship and respect for neighbors in faraway lands. "Science does not save species, people do," says Fearn. "The exchange of art and ideas between students in the US and in Papua New Guinea creates this caring connection that will translate into protection of wild places."

The TKCP is an award-winning conservation program based at Roger Williams Park Zoo that is dedicated to saving the Matschie's tree kangaroo from extinction. To date, the program is responsible for establishing more than 75,000 acres of PNG rainforest as a conservation area.

IMLS Awards More than $2 Million to Museums for Critical Conservation

The Institute of Museum and Library Services (IMLS) recently announced the 66 recipients of the 2004 Conservation Project Support. 66 grants were awarded, totaling $2,406,478. Recipients will match the grants with an additional $3,877,531. This year IMLS received 186 applications for a wide range of projects, including conservation treatment, training, surveys, and public education. The recipients included several AZA-accredited zoo and aquariums, and their projects are listed below:

Chicago Zoological Society - Brookfield, IL - $75,000

To develop recommendations about appropriate male social units and socialization to improve long-term captive management of bottlenose dolphins. This will be achieved through examination of the social behavior and social relationships among adult and maturing males within the captive population of Atlantic bottlenose dolphins (*Tursiops truncates*) combined with behavioral data on maturing dolphins in the wild.

Cincinnati Zoo and Botanical Garden - Cincinnati, OH - $75,000

To collaborate with the Center for Plant Conservation (CPC) to address reproductive and conservation problems of 40 highly endangered plant species held in the CPC's National Collection of Endangered Plants in order to revive populations and provide material for research and germplasm storage. Applicant Match: $173,904.

International Crane Foundation - Baraboo, WI - $54,610

$47,450 to monitor and improve the social and physical environment of the Foundation's captive breeding population of whooping cranes. The purchase of video equipment will enable ICF to better manage its breeding Whooping Cranes to increase the number of offspring produced for reintroduction. The equipment will also be used to conduct limited applied research to improve the management of these birds. $7,160 to develop an exhibit, outreach programs, and for Web site enhancement to share with visitors and the general public recorded images of the cranes.

Lincoln Park Zoological Gardens - Chicago, IL - $74,633

To develop new stand-alone analytical database software that will combine appropriate features of existing database software (SPARKS, SPARK-plug) with new features designed to accept and manipulate data exported from ZIMS (Zoo Information Management System). The need for this new software is critical because existing software is outmoded and new institutional records-keeping software will require major changes in how population management data are processed.

Toledo Zoo - Toledo, OH - $55,759

$46,152 to study the development of conservation husbandry and breeding strategies that can be used for three endangered butterfly species: Mitchell's satyr (*Neonympha mitchelli*), purplish copper (*Lycaena helloides*), and swamp metalmark, (*Calephelis muticum*). $9,607 to create a curriculum on the conservation of these species and work with state organizations to create a DVD for the public.

Information for News and Zoos is provided by the American Zoo and Aquarium Association