the CD, and they did so for an average of 6.8 hours and viewed 69% of its content. All read the print curriculum, spent a mean of 15.6 hours doing so, and read 93% of it. An average of 10.7 other NGO staff and volunteers also used the curriculum. Consultants communicated with NGOs a mean of 5.9 times (range from 1 to 17), each lasting 30 min.

The CD was evaluated as very useful by 30 directors (72%); the print materials were evaluated as such by 37 directors (88%); and the distance consultation, by 28 (67%). Of the 42 experimental condition NGOs, 37 held staff meetings to plan new programs based on the model, 34 chose target populations, and 31 identified community venues for carrying out the intervention. Grants were written to funders by 24 NGOs (57%) to support intervention implementation.

At follow-up, 18 of the 42 experimental condition NGOs (43%) but only 7 of the 41 controls (17%) had developed a new HIV prevention program based on the model that was disseminated [$\chi_{M\text{-}H}^2 = 5.28$, $P = 0.022$, odds ratio (OR) = 4.0, and confidence intervals (CI) = 1.3, 12.0]. POL core elements were incorporated into an existing prevention program by 23 (55%) experimental NGOs compared with 11 (27%) control NGOs ($\chi_{M\text{-}H}^2 = 7.14$, $P = 0.008$, OR = 4.6, and CI = 1.6, 13.4). Nearly twice as many NGOs in the technology transfer condition ($n = 27$, 64%) as controls ($n = 14$, 34%) either developed a new program or modified an existing program based on the disseminated model ($\chi_{M\text{-}H}^2 = 7.70$, $P = 0.006$, OR = 5.0, and CI = 1.7, 14.5).

Successful transfer of the disseminated intervention should also be reflected in increased incorporation of its core elements. Across all study NGOs, 67 candidate HIV prevention programs offered at both baseline and follow-up could be matched. Experimental and control NGOs had respective means of 1.03 and 1.35 core elements in their baseline programs. At follow-up, experimental NGOs incorporated significantly more core elements ($M = 2.81$) in candidate prevention programs than control NGOs [$M = 1.90$, $F(1,61) = 4.27$, and $P = 0.04$]. New programs developed by experimental NGOs at follow-up incorporated a mean of 2.69 POL core elements.

NGOs also often networked with other in-country providers to share information they gained during project participation. Of the 42 experimental NGOs, 23 (55%) gave copies of the CD or print curriculum to other community-based providers. Twelve NGOs shared the CD curriculum with a total of 81 other organizations, and 21 NGOs gave print manuals to 143 other organizations. Twelve experimental NGOs (29%) held training sessions on the model for a total of 73 other in-country agencies, and 19 (45%) met with their governments about the model. The intervention was incorporated into government HIV prevention strategic plans in 26% of the NGO countries.

When new intervention approaches are found efficacious in the research arena, they must be moved into the hands of service providers. It is important to distinguish between our interactive curriculum and passive uses of Web sites to simply supply information. We also believe that the personalized cultural consultation offered in this project was critical for helping providers tailor and implement the new method. Advantages of Internet- and computer-based dissemination methods include potential cost-effectiveness for reaching very large numbers of international providers, ability to produce new training packages whenever new advances are made, equity of opportunity for training by organizations even in resource-poor countries, and availability of on-demand training.

Our outcomes relied on director reports of NGO programs, which could be biased. However, 18 of the experimental NGOs (43%) spontaneously shared with us materials, photographs, and funding applications that corroborate their reports. It is clear that most NGOs did not adopt the dissemination intervention in its entirety and that their use of core elements was selective. Lack of funding was an implementation barrier cited by 25 of the 42 experimental NGOs (60%), and providers most often incorporated core elements that did not require additional cost or personnel. We do not know whether the omission of certain elements adversely impacted the effectiveness of programs implemented by the NGOs. Future research should study how service providers use and adapt science-based models and the effectiveness of these interventions when carried out by providers in the field. Our findings show that advanced communication technologies can link service providers worldwide with training, technical assistance, and consultation in the use of new approaches originating in the research arena (*10*).

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Supporting Online Material
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Materials and Methods
Tables S1 to S4

26 May 2004; accepted 13 August 2004

Avian Extinction and Mammalian Introductions on Oceanic Islands

Tim M. Blackburn,[1]* Phillip Cassey,[1] Richard P. Duncan,[2] Karl L. Evans,[3] Kevin J. Gaston[3]

The arrival of humans on oceanic islands has precipitated a wave of extinctions among the islands' native birds. Nevertheless, the magnitude of this extinction event varies markedly between avifaunas. We show that the probability that a bird species has been extirpated from each of 220 oceanic islands is positively correlated with the number of exotic predatory mammal species established on those islands after European colonization and that the effect of these predators is greater on island endemic species. In contrast, the proportions of currently threatened species are independent of the numbers of exotic mammalian predator species, suggesting that the principal threat to island birds has changed through time as species susceptible to exotic predators have been driven extinct.

The colonization of each land mass by humans has broadly coincided with an increase in the extinction rate of the native biota (*1–6*). The oceanic island bird species lost to extinction after human colonization are estimated to number in the hundreds to thousands (*3*, *7–10*). Although the exact causes of these extinctions are debated (*11–15*), human colonization is typically associated with habitat destruction and fragmentation and with other processes that can eliminate species, including overexploitation of populations (*5*, *16*).

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These processes may also be driven by the nonnative predator and herbivore species that humans introduce. In particular, the introduction of mammalian predators has caused the extinction of many populations and species of oceanic island birds (*3*, *17–20*) that, having evolved in their absence, lack appropriate antipredator responses.

The probability that a bird species has been extirpated from an island varies substantially among islands (Fig. 1A). This variation has normally been ascribed to differences in the characteristics of individual islands, such as area, isolation, elevational range, or date of human colonization (*9*, *21–24*). Isolation may increase extinction susceptibility, because the avifaunas of isolated islands are likely to have evolved for longer in the absence of predators, and so their species are more likely to react naïvely to exotic mammalian predators when they arrive. Large islands support larger populations of native bird species, which are thus less prone to extinction (*25*). Larger islands (or those with a greater elevational range) are also more likely to provide refugia from forces that promote extinction. Islands colonized longer ago may have more unrecorded prehistoric extinctions and so appear to have fewer extinctions, because only species that are relatively resistant to extinction remain [the filter effect (*21*)]. However, islands also vary substantially in the number of exotic mammal species, from none to >20 species. This variation may also drive variation in extinction probability. Here, we test this hypothesis using data from 220 oceanic islands worldwide (*26*) (table S1). Most mammalian introductions have occurred in the period after European colonization of an island, so we focus on bird extinctions that have occurred in this historic period. This increases the reliability of our results because historic extinctions are well documented, whereas prehistoric extinctions suffer from the incompleteness of the (sub)fossil record.

Our results show that islands with more exotic mammal predator species have lost a greater proportion of their avifauna since European colonization. The probability of extinction from an island's avifauna since European colonization increases with the number of exotic mammal predator species introduced (estimate ± SE = 0.24 ± 0.05, $P < 0.01$; the estimate is a linear parameter estimate from a general linear mixed model with binomial errors). The numbers of exotic mammalian predators and herbivores on an island are highly positively correlated (Pearson's $r = 0.67$, $n = 220$ islands, $P < 0.01$); however, the probability of extinction is unrelated to changes in the number of exotic mammal herbivore species (estimate ± SE = 0.02 ± 0.07, $P = 0.51$). This suggests that the predator relationship is not a simple consequence of more extensive environmental modification that leads to both more native extinction and exotic establishment. Introduced mammal predators are known to have caused specific island bird extinctions, but our results are consistent with introduced predators being a major cause of bird extinctions on oceanic islands around the world. Islands with more exotic predator species have suffered correspondingly greater losses.

We restrict analysis to historic extinctions, yet some exotic mammal species were established on islands before European colonization (e.g., the Pacific rat *Rattus exulans*), and many bird extinctions were prehistoric. A significant effect of predator species number is still observed when the analysis is expanded to include prehistoric mammal introductions and bird extinctions (multiple regression, predators: estimate ± SE = 0.24 ± 0.06, $P < 0.01$; herbivores: estimate ± SE = 0.01 ± 0.07, $P = 0.81$).

The apparent influence of predators could be an artifact of colinearity between the number of predator species introduced and biogeographic variables that have previously been shown to relate to extinction probability across islands, such as area or isolation (*9*, *21–24*). The number of predator species introduced to an island does indeed covary positively with island size ($r = 0.63$, $n = 197$ islands, $P < 0.01$) and maximum elevation ($r = 0.42$, $n = 183$, $P < 0.01$). However, because extinction probability should be lower on large and elevationally diverse islands, negative relationships would be expected if this colinearity were causing the predator effect. Moreover, multiple regression analyses that include island characteristics and number of exotic mammal predator species reveal that the effect of predators is robust to the inclusion of other variables that might determine avian extinction probability and that predator species number is the strongest predictor (Table 1).

Previous studies have found that islands with more exotic bird species have lost more native bird species (*27*). This relationship is argued to be an indirect consequence of extensive environmental modification that increases the habitat available for exotic bird species and negatively affects native

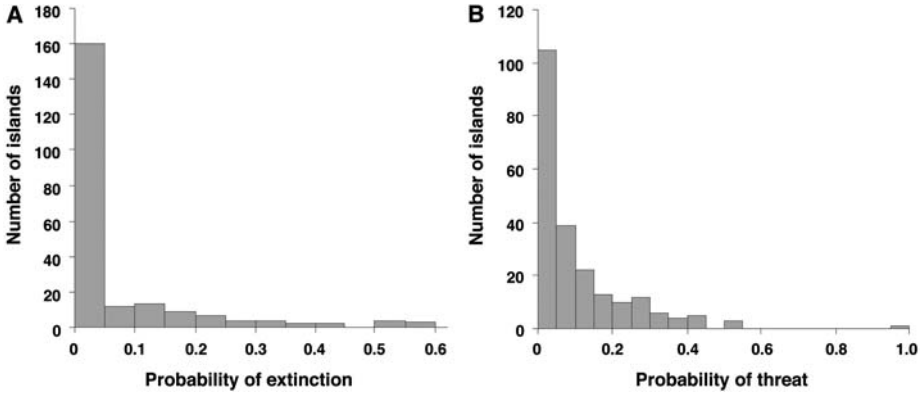

Fig. 1. Interisland variability in the probability of historic extinction and current threat for bird species. (**A**) The frequency distribution across islands ($n = 220$ islands) of the probability that a species in the historic fauna has become extinct from an island (the proportion extinct). Here and throughout the paper, extinction refers to the loss of a species from an island and so does not necessarily equate to global extinction. (**B**) The frequency distribution across islands ($n = 220$) of the probability that a species in the extant fauna is threatened with extinction (the proportion threatened). Threat refers to the risk of global extinction (*30*).

Table 1. The minimum adequate multivariate model (MAM) for historical probability of extinction in native island bird species. The MAM was derived by backward deletion of nonsignificant terms from a full model that also included island isolation, latitude, and human colonization date. Adding the number of exotic bird species to the full model does not alter the final MAM. The estimate is a linear parameter estimate.

Predictor	Estimate	SE	*t* value
Intercept	–4.25	0.67	
Number of introduced predatory mammal species	0.41	0.07	5.75***
Log (area)	–0.70	0.16	–4.40***
Log (maximum elevation)	0.69	0.27	2.48*

*$P < 0.05$. **$P < 0.01$. ***$P < 0.001$.

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species. Numbers of exotic bird species and native bird extinctions are also positively correlated across islands ($r = 0.62$, $n = 220$ islands, $P < 0.01$), as are numbers of both exotic bird and mammal species ($r = 0.65$, $n = 220$ islands, $P < 0.01$). However, the proportion of bird species extirpated from an island is independent of the number of exotic bird species in a multiple regression that includes the number of exotic mammalian predator species (Table 1).

Islands with more exotic predatory mammals suffered higher probabilities of bird extinction presumably because more diverse predator assemblages target a wider range of prey species or increase the overall predation rate on each species. However, alternative possibilities are that extinction has been driven by the range of predator types or by the presence of one or two particularly damaging predators, such as rats or cats (*17*, *19*, *20*, *28*), with the probability of these species being present on an island increasing with predator assemblage size [the sampling effect (*29*)]. We assessed these possibilities by comparing the fit to the data of different models in which the assemblage of introduced mammals was characterized in different ways (Table 2). Characterizing an assemblage by the number of predator species clearly provided the best fit to the data (the probability that this model was the best fitting out of the candidate set is 0.77), better than models in which the assemblage is characterized by the number of all mammal species or by the specific presence or absence of rats and/or cats. It was also a better fit than the number of taxonomic orders of predators introduced, which we used as a metric of number of predator types. These results strongly imply that each successive addition of an exotic predator acts to eliminate an additional proportion of an island's avifauna.

The impacts of mammalian predators ought to be greater on bird species endemic to islands, because these are more likely to have evolved in the absence of predators. For a given number of exotic predators, the probability of extinction is higher for island endemics than for species with continental populations (endemism: estimate ± SE = 1.55 ± 0.28, $P < 0.01$). Moreover, although higher extinction probability for species with continental populations is associated with decreasing island area (estimate ± SE = –0.84 ± 0.22, $P < 0.01$), increasing island elevation (estimate ± SE = 0.79 ± 0.32, $P < 0.01$), and the number of predatory mammal species (estimate ± SE = 0.35 ± 0.07, $P < 0.01$), extinction probability for island endemics is solely associated with the number of predatory mammal species (estimate ± SE = 0.16 ± 0.05, $P < 0.01$).

The probability that a species in the extant native avifauna is threatened with extinction also varies substantially across islands in our data (Fig. 1B). Is this variation also explained by the numbers of exotic mammal predator species? Certainly, the probability of threat is positively correlated with the probability that a species has already been extirpated from an island ($r = 0.43$, $n = 220$ islands, $P < 0.01$), and the proportion threatened is positively related to the total number of exotic mammal species (estimate ± SE = 0.04 ± 0.01, $P < 0.01$). However, in contrast to extinctions, there is no significant relationship between the probability of current threat and the number of exotic predators, although there is for the number of exotic herbivores (herbivores: estimate ± SE = 0.08 ± 0.03, $P < 0.05$; predators: estimate ± SE = 0.04 ± 0.05, $P = 0.36$). Moreover, in multiple regression, the probability of threat is significantly related only to biogeographic features of the islands (Table 3), with variation in the probability of threat across islands independent of the number of exotic mammal predator species.

The introduction of mammalian predators has been a major cause of bird extinctions on oceanic islands worldwide. Each successive predator introduction increases the number of species lost (Table 1), and island endemic species have suffered the most. In contrast, the proportion of an island's avifauna currently threatened with extinction is unrelated to the current number of exotic mammal species, even though many individual bird species are at risk from such predators (*30*), and islands with many extinctions in the past are also those with high levels of current threat. However, the current threat to native birds is higher on larger, more elevated, isolated islands that have been colonized more recently, consistent with the well-known filter effect (*21*, *31*). Presumably, most species susceptible to the current assemblages of exotic predators have already been driven extinct, especially on smaller, more easily colonized islands. However, these results do not imply that there is no longer any danger to island birds from exotic predatory mammals, because they relate to the current communities of exotic mammals on these islands. The establishment of each additional predator species is predicted to lead to additional extinctions among the native birds. This implies that exotic predators remain one of the major threats to island avifaunas, given that most islands currently have few predators (and some have none) and are likely to suffer progressively more extinctions if and when additional predators colonize. The likely consequences of future introductions are clearly presaged by the losses of the past.

Table 2. Comparison of models for the probability of historical bird extinction on oceanic islands (response variable) with the assemblage of introduced mammals characterized in different ways. Predictor variables in each model are $\log_{10}$ area, $\log_{10}$ elevation, and the variable in the first column; thus, the first model in this table is identical to that in Table 1, and subsequent models replace the number of predators in that model with the variable in the first column. Akaike's Information Criterion (AIC) values were calculated with PROC NLMIXED in SAS (*32*). ΔAIC is the difference between AIC values for each model and that with the lowest AIC (in this case, number of predators). *W* is the model's Akaike weight, which is the relative probability that the model is the best fit to the data of those tested (*33*).

Predictor	AIC	ΔAIC	*W*
Number of exotic predator species	335.9	0.0	0.77
Number of exotic herbivore species	341.0	5.1	0.06
Number of exotic mammal species	339.7	3.8	0.12
Number of exotic mammal orders	341.9	6.0	0.04
Rats, presence or absence	348.9	13.0	<0.01
Cats, presence or absence	345.3	9.4	0.01
Rats and cats, presence or absence	347.4	11.5	<0.01

Table 3. The MAM for probability of threat in extant island bird species. The MAM was derived by backward deletion of nonsignificant terms from a full model that also included the number of exotic predator species, the number of exotic herbivore species, and the number of exotic bird species. The estimate is a linear parameter estimate.

Predictor	Estimate	SE	*t* value
Intercept	–4.83	1.32	
Log (area)	0.19	0.09	2.21*
Log (maximum elevation)	0.39	0.17	2.24*
Log (years since colonization)	–0.76	0.24	–3.15***
Log (isolation)	1.06	0.31	3.38***

*$P < 0.05$. **$P < 0.01$. ***$P < 0.001$.

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The Impact of United States Recreational Fisheries on Marine Fish Populations

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We evaluated the commercial and recreational fishery landings over the past 22 years, first at the national level, then for populations of concern (those that are overfished or experiencing overfishing), and finally by region. Recreational landings in 2002 account for 4% of total marine fish landed in the United States. With large industrial fisheries excluded (e.g., menhaden and pollock), the recreational component rises to 10%. Among populations of concern, recreational landings in 2002 account for 23% of the total nationwide, rising to 38% in the South Atlantic and 64% in the Gulf of Mexico. Moreover, it affects many of the most-valued overfished species—including red drum, bocaccio, and red snapper—all of which are taken primarily in the recreational fishery.

Many of the ecological and political problems associated with fishing in U.S. waters historically have been attributed to foreign fishers (*1*, *2*). This perspective led to the passage of the Magnuson Act nearly 30 years ago to eliminate foreign competition, which set in motion a wave of expansion for U.S. commercial fishing fleets. By 1996, it was clear that removing the foreign fleets had not resulted in sufficient conservation (*3*), and amendments to the Magnuson Act more strongly emphasized reducing the fishing pressure of domestic fleets.

In the years following the amendment, the public focused on stock depletion, bycatch, and habitat damage caused by commercial fisheries (*4*, *5*) but paid little attention to the recreational sector. The perception that recreational fishing had little influence on stock declines derived from estimates that it contributed only 2% to U.S. landings (*6*). However, marine recreational fishing effort has increased by over 20% in the past 20 years (*7*), rivaling commercial fisheries for many major fish stocks, including summer flounder (*Paralichthys dentatus*), scup (*Stenotomus chrysops*), and red snapper (*Lutjanus campechanus*) (*8*).

We examined data from the National Marine Fisheries Service (NMFS) online databases (*9*), because we assumed that these readily accessible data sets were used to produce the existing estimates of recreational landings. Using these data, we produced a similar estimate. However, substantial inconsistencies in the online databases cloud the relevance of the number, such as the inclusion of commercially caught freshwater species and the exclusion of recreational data sets, such as data from the southeastern headboat sector (table S1).

We developed a comprehensive landings database (*10*) with data provided by the Marine Recreational Fisheries Statistics Survey (MRFSS), NMFS science centers and fishery management councils (FMCs), multistate marine fisheries commissions, and state natural resource agencies (table S2). We included landings data only and did not include fish discarded at sea either as regulatory discards (for commercial and recreational fisheries) or as a result of catch-and-release (exclusively a recreational fishing practice). After standardizing the data to allow for reasonable comparisons of these diverse data sets (tables S1 to S3), we assimilated a 22-year (1981 to 2002) time series of commercial and recreational landings.

We conducted analyses for the continental United States at national and regional levels, the latter based on the management jurisdictions of the following FMCs: Northeast (combining Northeast and Mid-Atlantic FMCs, Maine through Virginia), South Atlantic (*11*) (North Carolina through the east coast of Florida), Gulf of Mexico (the west coast of Florida through Texas), and Pacific (Washington through California, including Alaska only in the nationwide comparisons).

The nationwide analyses included three successively smaller groups of species: all federally managed marine fish; all marine fish, excluding walleye pollock (*Theragra chalcogramma*, used to produce frozen fish products) and menhaden (*Brevoortia tyrannus* and *Brevoortia patronus*, used almost exclusively to produce fish meal); and all "populations of concern" [i.e., those populations listed by NMFS (*12*) as either overfished or experiencing overfishing]. Menhaden and pollock were excluded because they have little or no recreational value and they are not considered overfished (*12*), although they comprise more than half of all U.S. fisheries landings: pollock landings approximate 1.8 million metric tons

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Avian Extinction and Mammalian Introductions on Oceanic Islands

Tim M. Blackburn, Phillip Cassey, Richard P. Duncan, Karl L. Evans and Kevin J. Gaston (September 23, 2004)

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