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Ecological and economic analysis of poaching of the greater one-horned rhinoceros (*Rhinoceros unicornis*) in Nepal

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Abstract. Nepal's greater one-horned rhinoceros (*Rhinoceros unicornis*) faces serious threats from poaching. Poaching of these rhinos is a complex problem, influenced by such diverse factors as the price of rhino horn on the international market, local socioeconomic factors, and the population dynamics of the species. Few studies have attempted to address this complexity. In this study, we model the poaching and population dynamics of the one-horned rhinoceros within an integrated framework of ecological, socioeconomic, political, and legal dimensions. The poaching model for rhinos in Royal Chitwan National Park (RCNP) in Nepal is combined with the population model for the species within a simulation framework and explored under various alternative policy scenarios with differing external socioeconomic and political conditions as well as internal policy response. We predict that, under the current (2003–2005) rhino conservation strategy, poaching would continue to be a major threat to the rhino population in RCNP. Furthermore, the internal policy response must begin to consider external factors such as socioeconomic conditions within the park buffer zone to be more effective in the long run. Finally, we find that, for long-run control, antipoaching policies should be directed at increasing the opportunity costs of poaching by creating better alternative economic opportunities, and at antipoaching enforcement.

Key words: antipoaching policies; economic incentives; Poisson regression; rhino conservation; Rhinoceros unicornis; *Royal Chitwan National Park, Nepal; simulation modeling.*

Introduction

Royal Chitwan National Park (RCNP), located in the low-lying Terai region of southern Nepal, provides habitat for most of the highly endangered greater one-horned rhinoceros (*Rhinoceros unicornis*) in the country. Although the rhino population in RCNP increased steadily during early years after the park's establishment, the growth rate has declined in recent years. The estimated rhino population in the Chitwan Valley in 1972, a year before RCNP was established, was, at a maximum, 147. The subsequent maximum estimated populations are 310 (in 1978), 376 (in 1988), and 460 (in 1994), the 1988 and 1994 estimates coming from official censuses (see Martin and Vigne 1996). Moreover, the latest census has revealed a startling population decline from an estimated 544 rhinos in 2000 to just 372 rhinos in 2005 (DNPWC 2001, Thapaliya 2005). This is worrisome given that a majority of the RCNP's budget is spent on protecting rhinos. Furthermore, the one-horned rhinoceros is protected as an endangered species under Appendix I of CITES, the Convention on International Trade in Endangered Species of fauna and flora, which has placed a ban on all international trade in live rhinos and rhino parts. Despite being considered one of the most successful rhino conservation programs in the world (Martin and Vigne 1996), the rhino population in RCNP clearly faces serious threats, particularly from poaching (Fig. 1).

Conservation of rhinos in Nepal started as early as 1940, when Rana rulers established armed rhino patrolling units to protect rhinos and other rare wildlife in the Terai region for their own exclusive hunts (Maskey 1998, Gurung and Guragain 2000). An influx into the region, coupled with the government's Integrated Agriculture Program to increase agricultural production in Chitwan, resulted in the systematic clearing of forests and concomitant rhino habitat destruction (Gurung and Guragain 2000). Settlers also killed wild animals, including rhinos, to reduce nuisance. By 1968, only ~100 rhinos remained in the Chitwan Valley (Kemf and van Strien 2002). In response, RCNP was created in 1973 from a small conservation area that had existed since 1958 (Maskey 1998, Gurung and Guragain 2000). The establishment of the RCNP was a major step toward conserving the rhinoceros in Nepal within their natural habitat, subsequently enhanced with the introduction of the Royal Nepalese Army (RNA), Antipoaching Units (APUs), and use of informants and reward systems to combat poaching (HMGN/DNPWC 2003*b*).

Poaching of the one-horned rhinos in Nepal can be viewed, in part, as an economic problem involving local poachers, middlemen, and buyers whose behavior is

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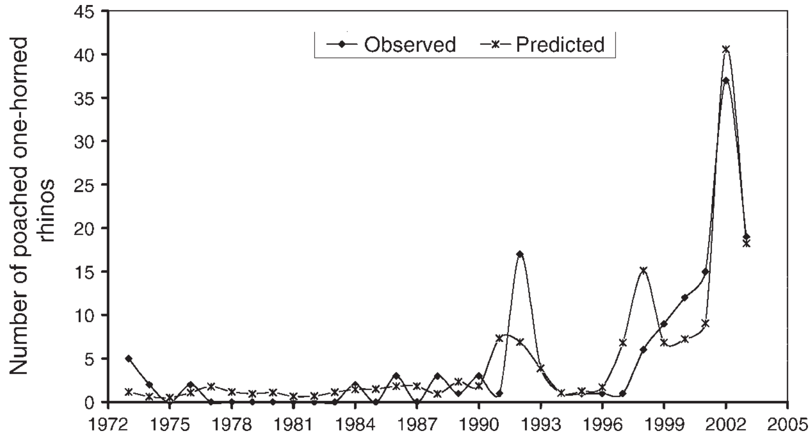

FIG. 1. Actual (circles) and fitted (crosses) values for the number of one-horned rhinos (*Rhinoceros unicornis*) poached in Royal Chitwan National Park (RCNP), Nepal, from 1973 to 2003.

influenced by the international (black) market price for rhino horn. Moreover, poaching at the local level is affected by local socioeconomic factors and incentives, such as the opportunity costs of poaching and penalties. Several studies have assessed the effectiveness of antipoaching enforcement in controlling poaching and conserving park rhinos in Nepal (see for example, Martin 1996, 1998, 2001, 2004, Martin and Vigne 1996, Maskey 1998, Gurung and Guragain 2000, Adhikari 2002, Yonzon 2002). However, none have systematically analyzed the reasons behind the poaching problem or the factors that might have influenced poaching levels in Nepal over the years. Furthermore, existing studies have either investigated poaching in isolation, mostly qualitatively from a socioeconomic and legal perspective (as just described), or have analyzed the one-horned rhino's population and ecology from a biological perspective (e.g., Laurie 1978, Dinerstein and Price 1991). Analogous poaching circumstances in Africa have been studied within the context of an integrated framework combining socioeconomic, political, legal, and ecological factors (e.g., Milner-Gulland and Leader-Williams 1992*a*, *b*, Skonhoft and Solstad 1996, Jachmann and Billiouw 1997, Bulte and van Kooten 1999*a*). However, these studies are mostly theoretical in nature, with little integrated empirical work. We believe that similar integrated research on Nepal's rhinos complemented with strong empirical analyses could lead to improved policy formulation. But, to date, no fully integrated studies have been conducted on the rhinos in Nepal. While other factors, such as habitat loss or human settlement, have likely contributed to the rhinos' decline, poaching has been shown to have a major negative effect on rhino numbers (Rothley et al. 2004).

This study aimed to fill that gap by assessing the poaching and population dynamics of the one-horned rhinoceros in RCNP within an integrated framework of ecological, socioeconomic, political, and legal dimensions. We began by building and empirically estimating a retrospective poaching model for RCNP covering the years 1973–2003. This poaching model was integrated with a population dynamics model predicting rhino numbers over the same period (Rothley et al. 2004). The integrated poaching/population model was then used to predict the effect of alternative antipoaching policies on the level of poaching and on the future rhino population.

SIMULATION MODEL

Grant et al. (1997) define simulation as "the process of using a model to mimic, or trace through step by step, the behavior of the system" under investigation. Simulating a natural resource management issue, such as rhino conservation, under various external conditions and internal policy responses could provide important insights into how the system behaves to potential policy responses. Our simulation model for the RCNP rhinos integrates an existing population dynamics model with a novel poaching behavior model, which is described in the next section, *Population dynamics*. Developing an integrated ecological and economic model allowed us to analyze the impacts of alternative rhino conservation and antipoaching policies on the level of poaching and, ultimately, on the Royal Chitwan National Park (RCNP) rhino population.

However, data limitations prompted us to take an innovative approach in constructing our integrated model. The poaching behavior model incorporates a high level of detail with its inclusion of numerous particular agents that drive poaching intensity, such as the probability of detection, capture, and conviction, alternative economic opportunities, the penalties when caught and convicted, and rhino horn prices. In contrast, the rhino population dynamics model contains considerably fewer specifics regarding the mechanisms that alter population density, and instead is parameterized with demographic data collected or estimated from previous rhino surveys. These two models are joined to

Table 1. Parameter values for the rhinoceros population model (data from Rothley et al. [2004: Fig. 3]).

Parameter description	Code	Values used in the model
Birth rate (no. rhinos per year)	b	0.34
Death rates (no. rhinos per year)		
Adults	d_A	0.029
Subadults	d_{SA}	0.022
Calves	d_C	0.028
Initial stage-class distribution for starting population		
Adults	A_0	65%
Subadults	SA_0	14%
Calves	C_0	21%
Carrying capacity of habitat (no. rhinos)	K	1000
Male–female ratio (percentage females in population)	f	58%

form the basis for our reduced-form poaching function that is estimated through regression. Practical reasons motivated this hybrid mechanistic/empirical modeling approach, as we had sufficient time-series data for an estimated model of poaching but only enough data for a simulation model of population. Confronted with this mismatch in data availability, we constructed a linked model which we would argue is an innovative solution to our data limitations, and a preferable alternative to inaction given the perilous predicament of the greater one-horned rhinos. We also note that the rhino population model did indeed forecast the unfortunate drop in rhino numbers in Royal Chitwan Park that was later confirmed in a subsequent population survey.

Population dynamics

Rothley et al. (2004) created a discrete, stage-class model to describe the population trajectory of the RCNP rhinos using a simple, logistic assumption of density-dependent regulation on their annual birth rate and including the effects of poaching and translocations. The model can be summarized by the following pseudocode:

$$A_t = A_{t-1} + (\text{maturing subadults} - \text{adult deaths} - \text{adults poached} - \text{adults translocated}) \times \Delta t$$

$$SA_t = SA_{t-1} + (\text{maturing calves} - \text{maturing subadults} - \text{subadult deaths} - \text{subadults translocated}) \times \Delta t$$

$$C_t = C_{t-1} + (\text{births} - \text{maturing calves} - \text{calf deaths} - \text{calves translocated}) \times \Delta t$$

where A_t, SA_t, and C_t are the number of adults (seven years or older), subadults (between four and six years old), and calves (less than four years old), respectively, at time period t, with Δt being the time step (in years). Note that only adult females breed and there is an approximate 3.5-year intercalving interval. Parameter estimates for birth and death rates and the male–female ratio were obtained from prior studies (natural deaths are treated separately from poaching and result most often from intraspecific encounters and tigers). An appropriate carrying capacity for the RCNP population was estimated by comparing model predictions to actual field counts. Table 1 lists the parameter values used in the population model.

The results presented and discussed in this paper are based on one of the three best-fit population models described in Rothley et al. (2004). However, we explored the poaching and population trajectories using the other two best-fit models in our simulation model. Since these runs provided similar results to those presented in this paper, we have not presented them. Furthermore, the points of discussion and conclusions would not change if either of the other two best-fit models were used instead of the one used in this paper.

Official censuses of the RCNP rhinos were performed every 5–6 years starting in 1988. Martin and Vigne (1996) independently compiled rhino population estimates for 1972, 1978, 1988, and 1994, whereas DNPWC (2001) provides the rhino population from the 2000 census. While this model is relatively crude given the complex relationships between rhinos and their predators, competitors, and habitat, its predictions closely tracked the field estimates for the years available. The model also correctly predicted the precipitous decline in RCNP rhino numbers between 2000 and 2004, reported in 2005 (Thapaliya 2005). This decline has mainly been blamed on the decreased security on the park leading to increasing poaching, with a reported annual mortality of 18 rhinos compared to the annual birth rate of just about four rhinos (Martin 2004, Chapagain 2005*a*, *b*, Thapaliya 2005).

Poaching behavior

Generally, poaching behavior has been studied within the context of theories of incentives to participate in illegal activities, and within the context of antipoaching effectiveness (e.g., Leader-Williams et al. 1990, Milner-Gulland and Leader-Williams 1992*a*, Leader-Williams and Milner-Gulland 1993, Jachmann and Billiouw 1997). These investigations have emphasized the socioeconomic factors, such as alternative economic opportunities, affecting poaching levels. Leader-Williams et al. (1990) concluded that rhino number declines result from problems originating outside the protected areas, such as increasing international rhino horn prices and a decline in alternative economic opportunities for local people living in neighboring areas. They further report that "law enforcement units were effective in capturing poachers, but were too small to provide protection to large populations of rhino and elephant."

The poaching model described in this section builds on earlier studies in hypothesizing a set of influences on the decision to poach. Based on these works, we would expect that (1) a rise in the probability of punishment or stricter punishment, (2) a fall in profits from poaching, or (3) a higher opportunity cost of poaching due to economic opportunities elsewhere, reduces poaching levels. We test these expectations with our empirical model of rhino poaching in the RCNP by exploring the predicted effects of (1) antipoaching measures that determine the probability of being caught and convicted, (2) penalties when caught, (3) the opportunity costs, (4) the direct costs that determine profit levels, and (5) rhino horn prices on the international (black) market.

The model.—Since poaching at the local level can be characterized as de facto open access, we use general functional relationships to derive a reduced-form poaching model, which is then estimated as an ad hoc model using available time-series data. In some respects our model follows the simple open-access model used by Bulte and van Kooten (1999*a*, *b*) to analyze the effects of an ivory-trade ban on poaching.

Although rhino management is under a state-property-rights regime, we assume that poachers act as if there is de facto open access governing their industry, since they operate outside legal property rights. A simple model for poaching industry profits under open access and the threat of capture and conviction can be represented as

$$\pi_t = p_t h(B_t, E_t, X_t) - \theta(B_t, E_t)[F_t + p_t h_t] - c(A_t)E_t \quad (1)$$

where π_t is poaching profits for year t; p_t is the gross price per rhino horn received by the local poachers; $h(B, E, X)$ is the poaching harvest function, measured as number of horns harvested; B_t is the antipoaching efforts; E_t is poaching effort; X_t is the number of the poached animal; $\theta(B, E)$ is the combined probability of detection, capture, and conviction, expressed as a function of poaching effort and antipoaching enforcement effort; F_t is the fine upon conviction and/or proxy for the value of time, if incarcerated; and $c(A_t)$ is the cost of poaching effort, expressed as a function of alternative economic opportunities, A. Note that if captured, the poachers must pay the fine and forego the benefits from the harvest in their possession.

If open-access profits are assumed to be zero in each time period (as a result of free entry and exit) then the equilibrium level of effort (E_t^*) can be derived for each time period by setting Eq. 1 equal to zero and solving for E_t^*. For a given set of parameters and observations on the variables in Eq. 1, we get

$$E_t^* = E^*(B_t, X_t, A_t, F_t, p_t). \quad (2)$$

Eq. 2 defines the equilibrium level of poaching effort. By substituting this equilibrium effort level into the harvest function $h(B, E, X)$, the following reduced-form poaching function is obtained (Skonhoft and Solstad 1998):

$$\begin{aligned} h_t &= h(E^*(B_t, X_t, A_t, F_t, p_t), B_t, X_t) \\ &= h(B_t, X_t, A_t, F_t, p_t) \quad \text{when } E_t^* > 0. \end{aligned} \quad (3)$$

Eq. 3 shows that the reduced-form poaching function depends on antipoaching effort, the rhino population, alternative economic opportunities, penalties when caught and convicted, and rhino horn prices. These variables cover the key economic determinants of poaching as hypothesized in earlier studies, allowing us to test empirically their impacts on poaching.

A final point concerns the price of rhino horn, p_t. The local price is unlikely to be determined locally, given an international black market for rhino horn; therefore, its price locally can be treated as fixed. However, we should recognize that this "fixed" price may vary over time due to changing international market conditions. Ideally, we would estimate a full price–quantity model, where the quantity demanded is expressed as a function of the international price of rhino horn, income (gross domestic product [GDP]), and other factors. This relationship can be inverted to give the international price as a function of the other variables. Then it would be possible to predict changes in the international price and modify the local price (normally expressed as a fixed percentage of the international price) as needed. Data limitations prevented us from taking this approach, although we retain a measure of this more general approach in our empirical modeling, as described in the next section.

Data and estimation of reduced-form poaching function.—The dependent variable in our estimation of Eq. 3 was the number of rhinos poached each year in RCNP for the period 1973–2003. Poaching data were available from various studies (Martin and Vigne 1996, Maskey 1998, Dhakal 2002) and from Department of National Parks and Wildlife Conservation (DNPWC) Annual Reports (DNPWC 1999, 2000, 2001, 2002, 2003). Annual rhino population estimates were obtained from the model in Rothley et al. (2004), as described in *Simulation model: Population dynamics*. The number of APUs active within the Chitwan valley since their establishment in 1993 was used as the sole index of antipoaching effort, B_t, because access to other information on antipoaching efforts, such as number of Royal Nepalese Army (RNA) personnel inside the park or their antipoaching patrols, was restricted for security reasons. However, these factors (i.e., RNA antipoaching efforts) likely did not vary significantly over the period. The opportunity cost of poaching was captured using per capita GDP for Nepal due to the lack of time-series data on local economic indicators or unskilled wage rates for the required period. (The GDP per capita for Nepal in real terms was obtained from the UN Statistics Division web site.)[4] Because data on actual fines imposed on convicted poachers over the years were not available, we used the maximum fines set by the law adjusted for inflation.

Ideally, we would have collected price data based on payments to local poachers for poached rhino horn and expressed these payments as a proportion of the international black market price. However, time series for the recent price of rhino horn and for quantities of rhino horn supplied to the international market do not exist, mainly because this trade has been illegal since the early 1970s. Instead, we capture the impact of the international horn price on poaching in Nepal using per capita GDP in East Asia as a proxy for the inverse demand relationship determining the international price. (We also tested the GDP per capita of Hong Kong as a proxy for price; however, this variable did not perform as well as GDP per capita for East Asia.) We use East Asia, since this region is the major consumer of rhino horn from Nepal. We use GDP since other researchers have found that the demand for horn in these countries is primarily income driven (Milner-Gulland 1993). Clearly, in taking this approach there are some limitations, such as an implicit assumption that the quantity of horn supplied to the black market is constant. (To see this, assume a demand relationship, $q_t = f(p_t, M_t)$, where q is the quantity of horn supplied to the market, p is price, and M is income, measured as GDP. The inverse demand relationship is $p_t = f^{-1}(q_t, M_t)$. For price to be determined solely by income, q would need to be held constant.)

Time series data on dependent and independent variables used in the estimation of the reduced form poaching function are presented in Table 2.

It is also important to note that while the equilibrium assumptions were made to derive the reduced-form poaching function (in effect to understand what factors could come into a reduced-form poaching equation), the estimation of the poaching function and subsequent simulation of the poaching and population of the rhinos are not performed to come up with an "equilibrium" figure for poaching or for the rhino population. Our aim is to use these models to predict the dynamic quality of the state variables (i.e., poaching, rhino population) under differing external conditions (prices, economic opportunities) and internal policy responses (number of APUs).

The reduced-form poaching function was estimated with LIMDEP 7.0 (Econometric Software, Plainview, New York, USA) using a Poisson regression method to account for the count-data nature of the dependent variable. The Poisson regression model assumes the equality of sample mean and sample variance for the dependent variable (i.e., the number of rhinos poached). However, in this study, the variance was more than 13 times the mean of the number of rhinos poached between 1973–2003, indicating overdispersion. An alternative to the Poisson regression model that relaxes the assumption of equality in mean and variance is the Negative binomial model. Both the Poisson and Negative binomial models were estimated using LIMDEP 7.0 (Econometric Software 2002), and the most suitable model (the Negative binomial model) was selected after (1) tests for overdispersion and (2) tests for the model itself against the alternative. The estimation results from the Negative binomial regression model are presented in Table 3.

Coefficients on all explanatory variables in the poaching model have expected signs (for example, the coefficient on APU is negative; Table 3). Further, the results show that the coefficients on APU, real per capita GDP for Nepal (GDPC_NEP), and the dummy for years affected by Maoist insurgency (MAOIST) are all significant at 1%. The coefficient on the exploitable rhino population (POPN[−1]) is significant within 10% ($P = 0.0536$). The coefficient on the real per capita GDP for East Asia (GDP_EA) is just outside the 10% significant level ($P = 0.1018$). However, the coefficient on the inflation-adjusted maximum fine (REAL_PEN) is highly insignificant ($P = 0.7038$).

Integrated model and the policy scenarios

The population dynamics and poaching models were integrated in a simulation model framework using STELLA 5.1.1 (High Performance Systems 1998). The population dynamics model provided rhino population density as an input to the poaching model. Actual poaching data were used as input to the population model for the period 1973–2003 (*Simulation model: Data and estimation of reduced-form poaching function*). After 2003, the poaching level was estimated from the poaching model and then fed into the population dynamics model, providing a prediction for the rhino population in the subsequent year.

For the simulated period (2004–2018), we varied the model's exogenous variables together to create five plausible scenarios. We employed the scenario approach as an alternative to a full sensitivity analysis, where each model variable (and potentially the assumed form of each relationship between the variables) is systematically altered and the model response is observed. We justify this approach in several ways. First, our models were constructed from components where each was examined for mistakes and illogical predictions before the next was constructed. Second, given the large number of variables in our models, a full, factorial sensitivity analysis would

[4] ⟨http://unstats.un.org/unsd/snaama/dnltransfer.asp?fID=11⟩

TABLE 2. Dependent and independent variable data used in the estimation of the poaching model.

Year	No. rhinos poached	Population (no. rhinos)	Penalty for poaching (Nrs)†	Real penalty (Nrs)‡	No. APUs	Maoist insurgency	GDP East Asia (US$)	GDP Nepal (US$)
1973	5	231	15 000	1626.90	0	no	564	83
1974	2	241	15 000	1357.47	0	no	611	103
1975	0	253	15 000	1261.56	0	no	659	112
1976	2	267	15 000	1302.08	0	no	709	102
1977	0	279	15 000	1184.83	0	no	846	99
1978	0	293	15 000	1103.75	0	no	1138	114
1979	0	305	15 000	1066.10	0	no	1219	127
1980	0	317	15 000	929.37	0	no	1292	131
1981	0	330	15 000	836.12	0	no	1369	146
1982	0	344	15 000	748.50	0	no	1285	151
1983	0	358	15 000	666.07	0	no	1378	146
1984	2	373	15 000	647.67	0	no	1441	148
1985	0	384	15 000	599.52	0	no	1492	154
1986	3	400	15 000	503.69	0	no	2006	155
1987	0	398	15 000	454.82	0	no	2384	169
1988	3	408	15 000	417.36	0	no	2862	186
1989	1	419	15 000	383.44	0	no	2908	180
1990	3	434	15 000	354.27	0	no	2925	189
1991	1	444	15 000	306.56	0	no	3274	169
1992	17	434	15 000	261.69	0	no	3579	179
1993	4	433	100 000	1622.59	2	no	4075	177
1994	1	442	100 000	1497.68	5	no	4371	197
1995	1	457	100 000	1391.59	5	no	4868	202
1996	1	472	100 000	1274.05	6	no	4534	205
1997	1	488	100 000	1224.89	7	yes	4275	220
1998	6	503	100 000	1101.20	8	yes	3882	203
1999	9	513	100 000	1024.80	8	yes	4309	218
2000	12	516	100 000	1000.00	8	yes	4590	227
2001	15	499	100 000	973.80	8	yes	4172	228
2002	37	495	100 000	945.18	0	yes	4122	220
2003	19	461	100 000	894.13	0	yes	4444	233

† Nr stands for Nepal Rupee.
‡ Inflation-adjusted maximum fine.

be impractical and difficult to interpret. Finally, the full sensitivity analysis would dictate a multitude of variable value combinations that would be impossible or highly improbable. The strength of our scenario approach is that we focus our attention on a limited number of realistic variable combinations having particular meaning and importance for the real world management scenario at hand. The weaknesses are that less model testing is performed, mistakes or variable combinations that highlight model irregularities may be missed, and novel variable combinations unlikely to be generated as scenarios based on existing conditions and assumptions but yielding desirable outcomes will not be produced. We also do not explore the sensitivity of our model to uncertainty or produce a complete ranking of the effect size for individual parameters.

TABLE 3. Negative binomial regression estimates for reduced-form poaching model for Royal Chitwan National Park (RCNP), Nepal.

Explanatory variable	Mean (SD)	Negative binomial model: Coefficient (SE)	Negative binomial model: P	Expected sign
Constant		0.4501 (2.6990)	0.8676	
POPN[−1]	385.35 (91.758)	0.0189 (0.0098)	0.0536	+
APU	1.8387 (3.1101)	−0.2130 (0.0578)	0.0002	−
REAL_PEN	934.2333 (397.1678)	−0.0002 (0.0006)	0.7038	−
GDPC_NEP	166.8709 (43.424)	−0.0526 (0.0173)	0.0025	−
GDPC_EA[−1]	2502.4516 (1509.5733)	0.0007 (0.0004)	0.1018	+
MAOIST	0.2258 (0.425)	2.3371 (0.5823)	0.0001	+

Notes: The dependent variable is the number of rhinos poached per year. POPN[−1] is population of rhinos in the park lagged by a year. APU is the number of antipoaching units. REAL_PEN is the inflation-adjusted maximum fine (in Nepal Rupees). GDPC_NEP is the real per capita GDP of Nepal. GDPC_EA[−1] is the real per capita GDP for East Asia lagged by a year. MAOIST is the dummy for years affected by Maoist insurgency. The sign indicates the expected effect of the independent variables on the level of poaching (the dependent variable), so a negative sign indicates the expected negative effect (reduction) on poaching of a particular variable.

Table 4. Baseline and alternative assumptions used in establishing the policy scenarios for the gross domestic product (GDP) in Nepal and East Asia and presence/absence of the Maoist insurgency in Nepal.

External factors	Assumptions		
	Baseline	Alternative 1	Alternative 2
GDP per capita Nepal and Maoist insurgency (GDPC_NEP)	growth with ongoing Maoist insurgency (2%)	low growth (2%) with ongoing Maoist insurgency	high growth (4%) without Maoist insurgency, 2009 onward
GDP per capita East Asia (GDPC_EA)	historical mean growth rate, 1973–2003 (4.2%)	low growth (3%)	high growth (6%)

In the baseline (Scenario 1), we assumed the current (i.e., year 2003) policy to continue throughout the simulation period. In terms of external conditions, the growth rate in East Asia was assumed to be at 4.2%, which represented a mean growth rate for the poaching model estimation period 1973–2003. The economic growth rate in Nepal was assumed to be 2%, combined with a continued Maoist insurgency. The key assumptions concerning GDP in Nepal and East Asia for the baseline (Scenario 1) and four alternative policy scenarios (Scenarios 2, 3, 4, and 5) are presented in Table 4, while full descriptions of the five scenarios are contained in Table 5. Scenario 3 represented the "best case" for rhino conservation, as the GDP per capita in East Asia, which shows a positive impact on the level of poaching (i.e., increases poaching), was assumed to be growing at a lower rate (3%), and the GDP per capita in Nepal, which shows a negative impact on the level of poaching in RCNP (i.e., decreases poaching), was assumed to be growing at a higher rate (4%). In contrast, Scenario 4 represented the "worst case" for rhino conservation, as in this scenario the GDP per capita in East Asia was considered to be growing at a higher rate (6%), while GDP in Nepal was set at a lower rate (2%). The other two alternative scenarios (i.e., Scenarios 2 and 5) represented intermediate cases where both the GDP per capita in East Asia and Nepal were considered to be growing either at a lower or a higher rate, thereby balancing the positive/negative impacts of each factor.

The integrated model was then simulated from the initial year using the four alternative scenarios, each with the number of APUs set at 0, 5, 10, or 15, for a total of 17 model runs (1 baseline + 16 alternative scenarios).

Results

Scenario 1 (baseline) results in the highest poaching figures on average, and during most of the simulation years, compared to any other simulation runs, except for Scenario 4 (worst case) with the number of Antipoaching Units (APUs) set at 0 and 5 (Table 6). Furthermore, mean numbers of poached rhino under

Table 5. Policy scenarios used in the simulations, based on the GDP assumptions and the internal policy response (i.e., the number of APUs deployed).

Scenario	Description
Scenario 1	
Baseline	This scenario assumes that the politico-economic condition in Nepal post 1996 continues throughout the simulation period (i.e., Nepal GDP growth rate at 2% and continuing Maoist insurgency). Furthermore, it assumes the growth in East Asian economy at the historical mean (i.e., 4.2%).
Scenario 2	
LG-EA + LG-NEP	The assumption regarding the politico-economic condition in Nepal in this scenario is the same as in the baseline. In addition, this scenario assumes a low growth rate in East Asia of 3%.
Scenario 3	
Best case: LG-EA + HG-NEP	This can be considered the "best case" scenario. It assumes a low growth rate in East Asia, coupled with high growth rate in Nepal (4%) from the year 2009 onward and the solution of the Maoist problem.
Scenario 4	
Worst case: HG-EA + LG-NEP	This can be considered the "worst case" scenario. It assumes a high growth rate in East Asia (6%). However, the growth rate in Nepal is assumed to remain low (2%) with the continuation of the Maoist insurgency.
Scenario 5	
HG-EA + HG-NEP	This scenario assumes a high growth rate in East Asia, as well as a high growth rate in Nepal from 2009 on with the Maoist insurgency problem resolved.

Notes: Each of the alternative scenarios was simulated over the period 2004–2018 with Antipoaching Units (APUs) at four different levels: 0, 5, 10, and 15 units. The baseline scenario was simulated over the same period with APUs at 0 (i.e., same level as in the base year, 2003). Key to abbreviations: LG, low growth; HG, high growth; EA, East Asia; NEP, Nepal.

TABLE 6. Simulated poaching and population figures for the rhinoceros in RCNP for every fifth year of the simulation period, and the mean poaching and population figures for the entire simulation period.

Scenarios, no. APUs	Poaching and population (no. greater one-horned rhinos)					
	2003 (Base year)		2008		2013	
	Poached	Population	Poached	Population	Poached	Population
Scenario 1						
Baseline						
0	19	461	7	480	10	519
Scenario 2						
(LG-EA + LG-NEP)						
0	19	461	6	481	7	528
5	19	461	3	499	4	560
10	19	461	1	505	2	576
15	19	461	0	508	1	583
Scenario 3						
Best case: LG-EA + HG-NEP						
0	19	461	6	481	0	553
5	19	461	3	499	0	574
10	19	461	1	505	0	582
15	19	461	0	508	0	585
Scenario 4						
Worst case: HG-EA + LG-NEP						
0	19	461	9	479	15	501
5	19	461	4	498	11	545
10	19	461	1	505	6	569
15	19	461	1	508	2	580
Scenario 5						
HG-EA + HG-NEP						
0	19	461	9	479	1	547
5	19	461	4	498	0	571
10	19	461	1	505	0	581
15	19	461	1	508	0	584

Notes: Non-integer values have been rounded. Key to abbreviations: LG, low growth; EA, East Asia; NEP, Nepal; HG, high growth.

Scenarios 2, 3, and 5 are predicted to be significantly lower than those under the baseline for all four APU policy options (t test for differences in means, $P < 0.001$, df = 14). In addition, Scenarios 2, 3, and 5 predict a significantly higher mean rhino population for the simulation period under all four APU policy options, compared to the baseline (t test for differences in means, $P < 0.01$, df = 14). Simulated trajectories for numbers of rhinos poached and the rhino population, under the four APU policy options, are shown in Figs. 2–5.

Under the policy option with no APUs, the number of rhinos poached is predicted to increase from the first year of the simulation. However, for Scenarios 3 and 5, poaching drops from the year 2009 onward (Fig. 2A). Scenario 2 (with low economic growth in both Nepal and East Asia) predicts a relatively stable level of poaching with a mean of nine rhinos poached per year (minimum = 6, maximum = 10). In contrast, Scenario 4 (with low economic growth in Nepal and high growth in East Asia) predicts the highest level of poaching among all the scenarios including the baseline (Fig. 2A). The mean number of rhinos poached under Scenario 4 (mean = 13, minimum = 6, and maximum = 10) is significantly higher than that under the baseline (t test for differences in means, $P = 0.002$, df = 14). The rhino population under Scenario 4 with no APUs is predicted to be significantly lower than that under the baseline (t test for differences in means, $P = 0.0004$, df = 14; Fig. 2A).

A policy option with five APUs produces similar results over the simulation period as no APUs (Fig. 3A, B). The main difference being that under all the alternative scenarios the poaching level drops to fewer than four rhinos per year for the first 4–5 years under this policy. After that initial drop, however, poaching picks up sharply under Scenario 4 (worst case), reaching a figure of 19 rhinos poached by the end of the simulation period (Fig. 3A). On average, poaching under Scenario 4 with five APUs (mean = 10, minimum = 3, maximum = 19) is not significantly different from the baseline (t test for differences in means, $P = 0.80$, df = 14); however, the mean rhino population under Scenario 4 is significantly higher than that of the baseline scenario (t test for differences in means, $P < 0.001$, df = 14).

A policy response consisting of 10 APUs is effective in controlling poaching completely under Scenarios 3 and 5, and keeping poaching below four rhinos per year throughout the simulation period under Scenario 2 (Fig. 4A). Under Scenario 4, this policy option reduces poaching to below four rhinos per year for the first seven years

TABLE 6. Extended.

Poaching and population (no. greater one-horned rhinos)			
2018		Mean (2004–2018)	
Poached	Population	Poached	Population
11	544	9	501
7	568	7	508
5	608	4	533
3	633	2	545
1	645	1	551
0	624	2	529
0	642	1	543
0	649	0	550
0	651	0	552
20	492	19	485
19	546	9	517
15	595	5	537
9	627	2	547
0	617	3	523
0	638	1	542
0	647	0	549
0	651	0	552

of the simulation, but poaching rises from 2011, reaching a peak of 16 rhinos at the end of the simulation period, which is higher than that under the baseline (Fig. 4A). Under this policy option, the mean number of rhinos poached over the simulation period is significantly lower, and the mean rhino population significantly higher, for all scenarios compared to the baseline (t test for differences in means, $P < 0.01$, df $= 14$).

Finally, the policy response with 15 APUs predicts successful control of poaching for Scenarios 2, 3, and 5 over the entire simulation period (Fig. 5A). For Scenario 4 (worst case), this policy keeps poaching below four rhinos per year for the first 11 years of the simulation period. After this point, poaching increases gradually until it reaches nine rhinos at the end of the simulation period (Fig. 5A). As with the 10-APUs policy option, the mean number of rhinos poached over the simulation period is significantly lower, and the mean rhino population significantly higher, when the alternative scenarios are compared to the baseline (t test for differences in means, $P < 0.01$, df $= 14$).

DISCUSSION

Our baseline rhino conservation strategy simulation shows that poaching would continue to be a major threat to the Royal Chitwan National Park (RCNP) rhino population over the simulation period, 2004–2018, with a mean of nine rhinos (minimum = 6, maximum = 11) poached per year. Poaching of a mean of nine rhinos per year would preclude recovery of the population to a size that can be supported by the park (Rothley et al. 2004), given that the rhino population is decreasing at present.

It is important to note also that the simulation model assumes fixed growth in the external factors driving the simulation model (e.g., gross domestic product [GDP] per capita of Nepal and East Asia), leading to much smoother poaching and population trajectories than might be expected under the normal annual variation that occurs with these explanatory variables. The actual outcome of the current rhino conservation strategy could be much worse, especially in terms of the rhino population, if one or more of the external variables were to change abruptly in the "wrong" direction. For example, if there is a sharp increase in per capita GDP in East Asia, leading to a higher demand for rhino horn, or a sharp decline in per capita GDP in Nepal that lowers the opportunity cost of poaching, poaching would likely increase significantly. The large decline in the population of rhino at RCNP between the last two censuses demonstrates the short-term volatility in a population of a species under poaching stress. Although most of the poaching in RCNP between the 2000 and 2005 censuses has been blamed on the decline in security within and around the park, it is noteworthy that the original APU system was disbanded in 2001, severely inhibiting information gathering and antipoaching enforcement (Adhikari 2002). Furthermore, political instability arising from the Maoist insurgency not only weakened the antipoaching efforts but also affected adversely the country's economy. This development may have been partly responsible for the observed increase in poaching, since it would reduce the opportunity costs of poaching.

The poaching and population trajectories from the simulation of alternative scenarios highlight some important factors that may influence the level of poaching in RCNP. At a certain level of policy response (i.e., the number of APUs), changes in economic conditions in East Asia or Nepal are predicted to have distinct effects on the poaching levels. It is not surprising that the "best case" scenario, characterized by high economic growth in Nepal and low growth in East Asia, predicts the lowest levels of poaching for all APU assumptions. However, Scenario 5, with high economic growth in Nepal and East Asia predicts levels of poaching very similar to the "best case." This shows that the increase in poaching in response to high demand for rhino horn (as a result of high economic growth in the East Asian region) could be more than compensated for by higher opportunity costs of poaching (as a result of increasing economic opportunity in Nepal), other factors staying constant. The high-growth scenario in

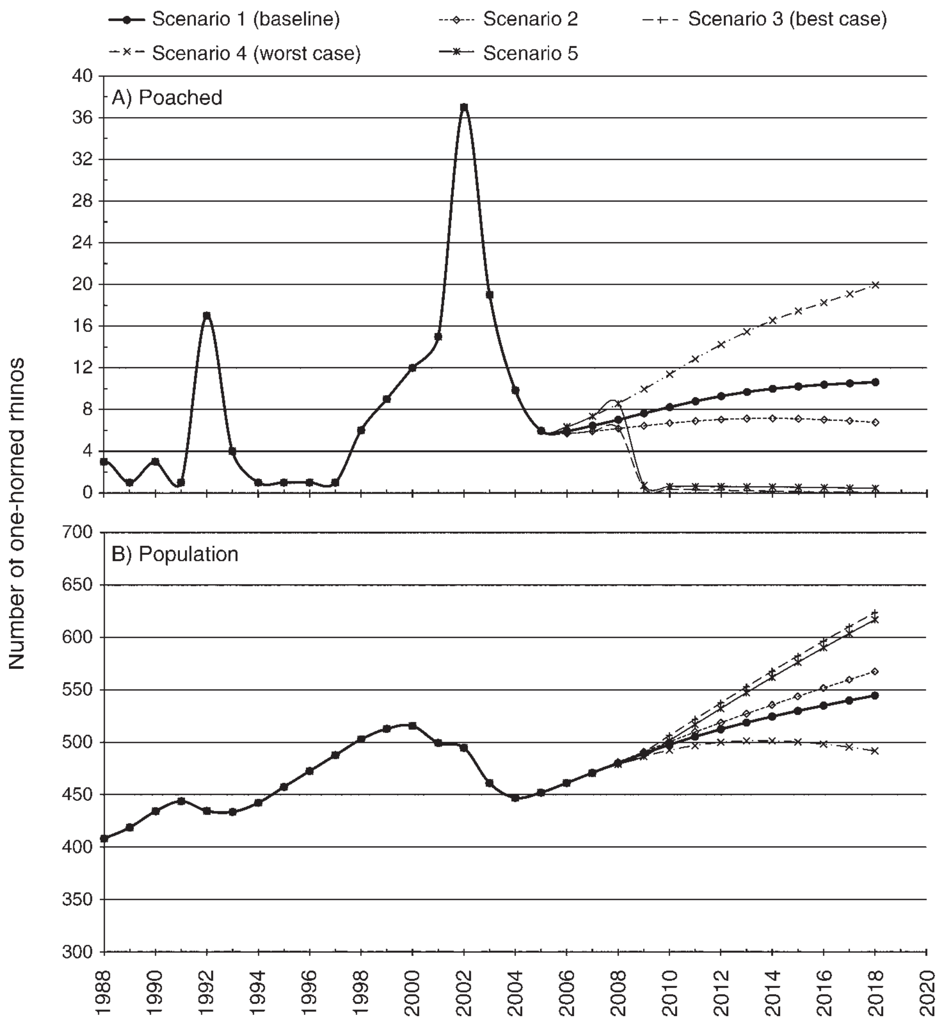

FIG. 2. Simulated poaching and population figures for the one-horned rhinoceros in RCNP under the policy of no Antipoaching Units (APUs) (i.e., APU = 0). (A) The portion of the figure until year 2003 shows the actual number of rhinos poached in RCNP, and the portion of the figure from year 2004 onward shows the predicted poaching figures from the simulation model for years 2004–2018 under the baseline and four alternative scenarios. (B) The portion of the figure until year 2003 shows the rhino population in RCNP obtained from the model in Rothley et al. (2004), and the portion of the figure from year 2004 onward shows the predicted rhino population from the simulation model for years 2004–2018 under the baseline and four alternative scenarios.

Nepal assumes solution of the Maoist insurgency and subsequent high growth in the economy starting in 2009, so that Scenario 3 is identical to Scenario 2 up to the year 2008 (which is reflected in the poaching and population figures in Table 6). However, Scenario 2 predicts higher and increasing poaching figures at each level of APUs compared to Scenario 3, which predicts significantly lower and decreasing poaching from year 2009 onward.

Although an increase in the number of APUs in Scenarios 2 and 3 is predicted to produce an immediate impact by reducing the level of poaching, under Scenario 2 poaching gradually increases, regardless of the number of APUs. However, as the rhino population rises over time, even under Scenario 2, the gradual increase in poaching could be due to this population effect (i.e., as population increases, it will require less poaching effort to kill a rhino). This means antipoaching efforts will have to increase (or be more effective) over time to control this additional poaching. This suggests that the solution of the Maoist problem, combined with better economic conditions, could reduce the level of poaching. But it also indicates that increasing the number of APUs alone would be insufficient to control poaching in the long run, as has been demonstrated historically in RCNP.

Scenario 4 reveals a much more alarming picture, arising from the negative impacts of the external factors taken together. First, poaching in the long run under this scenario is the highest among all the alternatives

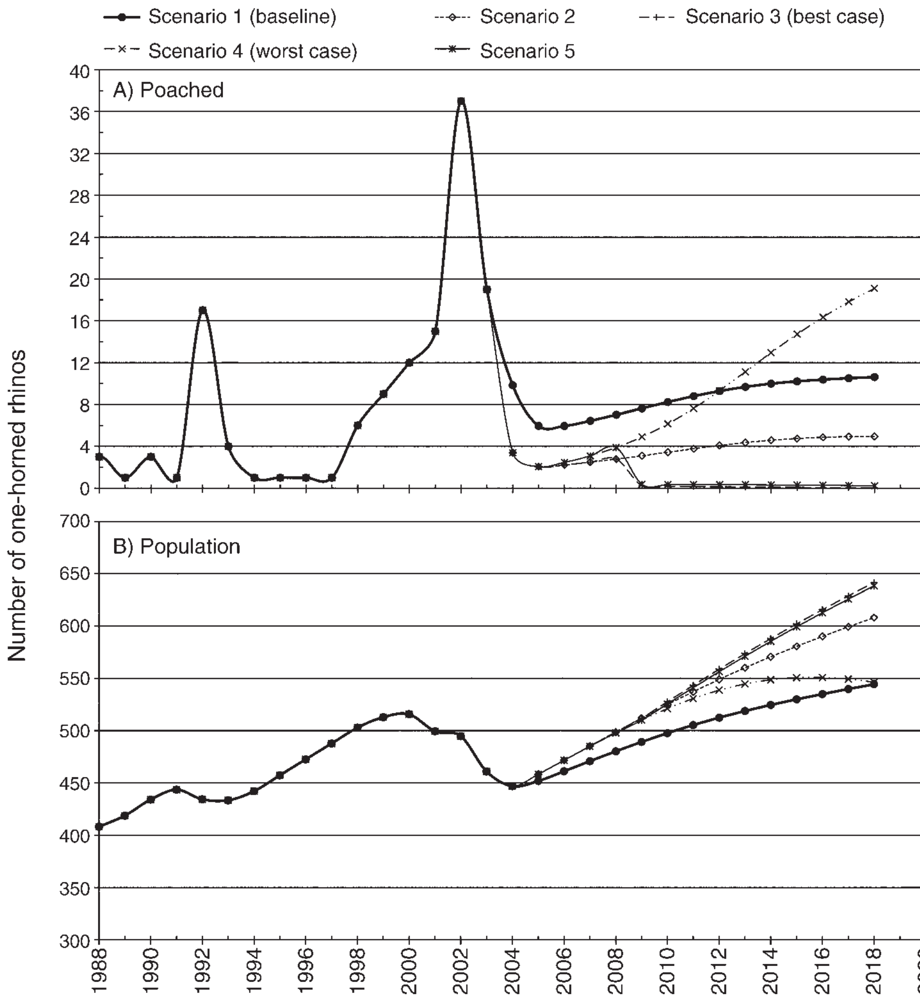

FIG. 3. Simulated poaching and population figures for the one-horned rhinoceros in RCNP under the policy of five APUs (i.e., APU = 5). (A) The portion of the figure until year 2003 shows the actual number of rhinos poached in RCNP, and the portion of the figure from year 2004 onward shows the predicted poaching figures from the simulation model for years 2004–2018 under the baseline and four alternative scenarios. (B) The portion of the figure until year 2003 shows the rhino population in RCNP obtained from the model in Rothley et al. (2004), and the portion of the figure from year 2004 onward shows the predicted rhino population from the simulation model for years 2004–2018 under the baseline and four alternative scenarios.

considered. Mean poaching under this scenario at 15 APUs is comparable to the average poaching under Scenario 3 (the "best case") with no APUs (Table 6). This shows that under extreme external conditions, APUs at a fixed level are predicted to be increasingly ineffective and a very large number of APUs will be required to control poaching completely or at a level comparable to the "best case." Secondly, the rhino population under Scenario 4 is the lowest among all the alternatives, and at all APU levels, with a predicted population decline from the year 2014 onward (Table 6, Figs. 2B, 3B, 4B, 5B).

In light of this discussion, recent policy changes have emphasized the strengthening of antipoaching capability through various additional measures. The Rhino Conservation Action Plan focuses on legislative measures to strengthen antipoaching capabilities, such as by bringing into force the CITES (Convention on International Trade in Endangered Species of fauna and flora) Bill nationally. In contrast, the Antipoaching Strategy for RCNP focuses strictly on increasing antipoaching capacity at the field level by providing better resources to APUs. For example, the antipoaching patrol strategy now includes a different form of patrol, termed the "sweeping operation." In a sweeping operation, all resources are consolidated to create a large patrolling unit, and antipoaching patrols are conducted in suspected areas, often using elephants (Martin 2004). This approach initially resulted in more successful control of poaching; however, more recent reports of renewed heavy poaching, and a declining rhino population in RCNP, raise significant questions regarding the effectiveness of this strategy.

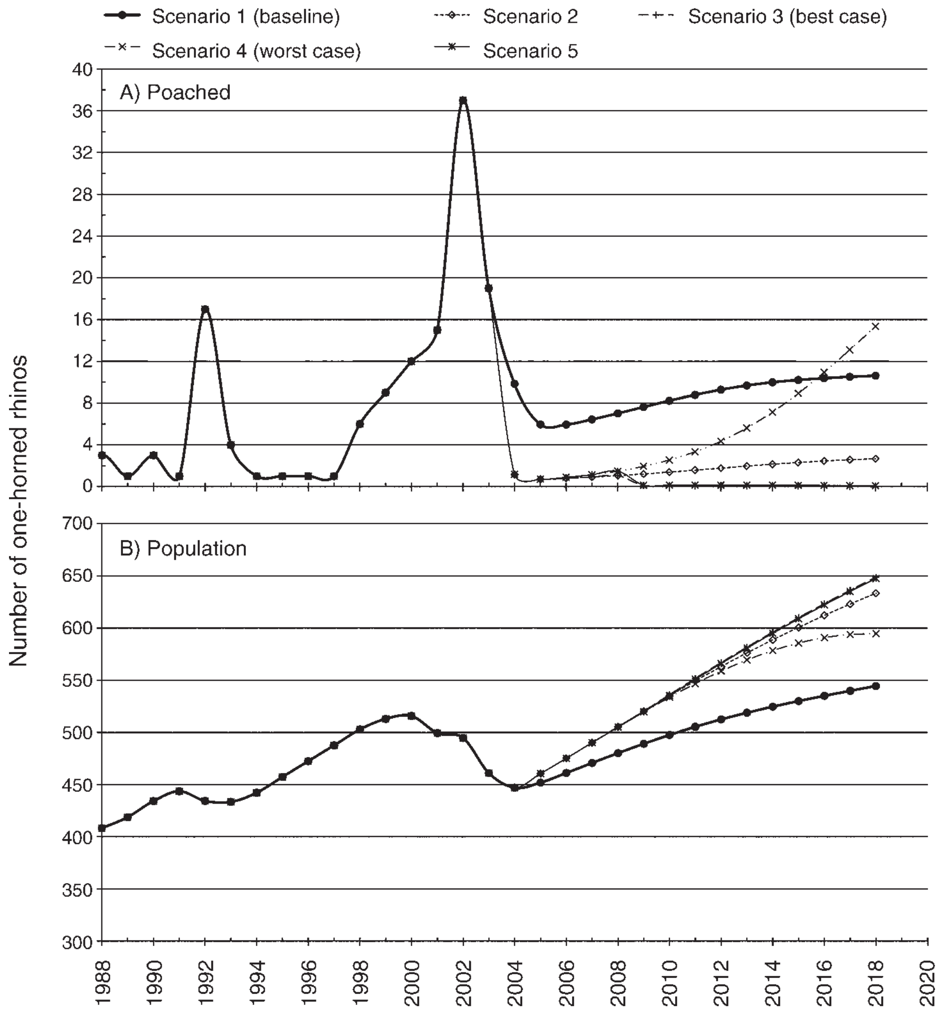

FIG. 4. Simulated poaching and population figures for the one-horned rhinoceros in RCNP under the policy of 10 APUs (i.e., APU = 10). (A) The portion of the figure until year 2003 shows the actual number of rhinos poached in RCNP, and the portion of the figure from year 2004 onward shows the predicted poaching figures from the simulation model for years 2004–2018 under the baseline and four alternative scenarios. (B) The portion of the figure until year 2003 shows the rhino population in RCNP obtained from the model in Rothley et al. (2004), and the portion of the figure from year 2004 onward shows the predicted rhino population from the simulation model for years 2004–2018 under the baseline and four alternative scenarios.

Several limitations of our study are worth noting. Regarding the estimation of the reduced-form poaching model, the major limitation was the availability of time-series data for the independent variables used in the model. As a result, proxy variables were used to capture the effects of a number of variables. For example, data on the international (black) market price of rhino horn were not available for the entire period of analysis. Hence, we used GDP per capita for East Asia (the main consumer of rhino horns from Nepal) as a proxy variable to capture the influence of demand for rhino horn. Although the variable was consistent in the nature of its impact on poaching, it was not highly significant, as could be expected for a proxy variable. Since we use a proxy variable, we cannot truly test the potential effectiveness of a policy measure such as reducing the price of rhino horn, as argued by Brown and Layton (1997) in their analysis of rhino poaching in Africa.

Two further points can be made regarding data availability for the poaching model. Firstly, data on antipoaching enforcement efforts were either not available (e.g., records on antipoaching patrols by the APUs) or not obtainable due to security concerns related to the political situation in Nepal (e.g., records on size and antipoaching patrol efforts by the Royal Nepalese Army [RNA]). Thus, we relied on the number of APUs in RCNP to capture the impact of antipoaching efforts on poaching. This could have led to an overestimation of the impacts of APUs on poaching; however, other studies have suggested that APUs were more effective in controlling poaching than the RNA alone (Martin 1996, 1998, Martin and Vigne 1996, Adhikari 2002). The other

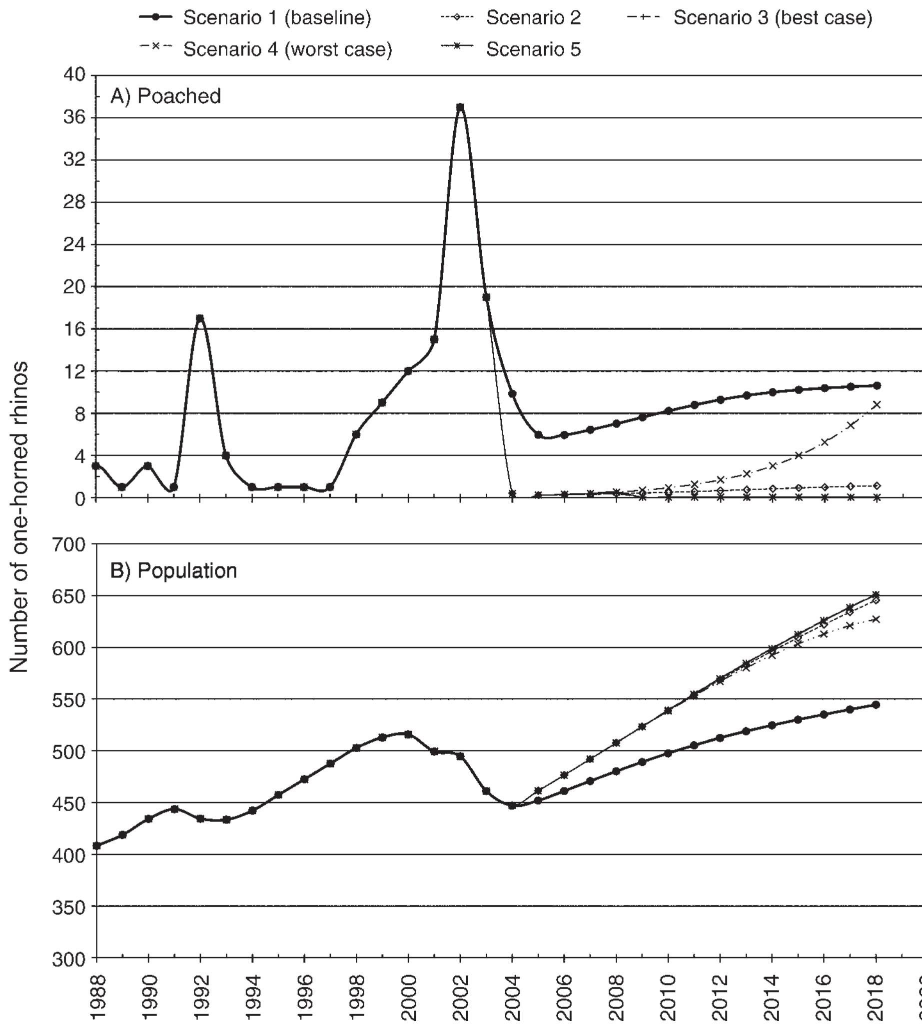

FIG. 5. Simulated poaching and population figures for the one-horned rhinoceros in RCNP under the policy of 15 APUs (i.e., APU = 15). (A) The portion of the figure until year 2003 shows the actual number of rhinos poached in RCNP, and the portion of the figure from year 2004 onward shows the predicted poaching figures from the simulation model for years 2004–2018 under the baseline and four alternative scenarios. (B) The portion of the figure until year 2003 shows the rhino population in RCNP obtained from the model in Rothley et al. (2004), and the portion of the figure from year 2004 onward shows the predicted rhino population from the simulation model for years 2004–2018 under the baseline and four alternative scenarios.

aspect of antipoaching enforcement—the maximum fine for convicted poachers—changed only once during the study period. This meant that the real level of fine (i.e., inflation adjusted) was continuously declining over much of the period of analysis. Thus, it is not surprising that the impact of the maximum fine level (adjusted for inflation) was insignificant, although the direction of impact was negative, as expected.

Secondly, we used per capita GDP for Nepal to capture the opportunity costs of poaching, instead of local economic indicators such as local/regional GDP, employment, or the local unskilled wage rate. Although information on the unskilled wage rate was available for some recent years, data on this and other local economic indicators for the entire 30-year period required for model estimation were not available. Nevertheless, the estimated coefficient on the national GDP was highly significant and consistent, and the nature and direction of its impact on poaching conformed with our expectation.

Conclusions and policy implications

In policy terms, current rhino conservation policies have not been effective in controlling poaching and protecting rhinos, even within the national park boundaries. Our findings highlight the influence of socioeconomic, political and legal factors on the level of poaching in Royal Chitwan National Park (RCNP), and consequently on the park's rhino population. A better understanding of these influences could be used for formulating policies to control poaching and to better protect rhinos in the long term. The analysis of alternative scenarios is especially important as it allows us to compare outcomes for different policy scenarios under a range of exogenous internal and external conditions.

The results indicate that these exogenous factors have had a significant impact on the level of poaching, and consequently, on the rhino population in RCNP and that these influences should be factored into future policy decisions. This is highlighted especially by the results under Scenarios 3 and 4 (the "best case" and the "worst case" scenarios, respectively). The difference in the level of APUs required to control poaching between these two extreme cases shows that the internal policy response (i.e., setting the number of APUs) should take into account the prevailing external conditions to form successful antipoaching policy. Furthermore, differences in the simulation outcomes between Scenarios 2 and 3, and between Scenarios 4 and 5, highlight the importance of alternative economic opportunities in Nepal for reducing the level of poaching.

However, the experience from the illegal bushmeat trade in Africa is of interest with respect to the alternative economic opportunities argument (Yamagiwa 2003). Where war and disturbance are involved (as is the case in Nepal as well), food supplies may be diminished, alternative income opportunities may be fewer, and poaching enforcement reduced, so that poaching increases. Only if the disturbance is profound enough to reduce access to wildlife will conservation conditions improve under these social upheaval conditions (Draulans and van Krunkelsven 2002). Nonetheless, there are differences when illegal poaching involves food vs. tradable commodities (e.g., elephant ivory, rhino horn). The motivation of local populations to poach for food for survival purposes, or to sell poached products locally, is not present with the trade in rhino horn. Instead, the presence of an international market for rhino horn means that poaching needs to be organized by sophisticated gangs rather than being undertaken by isolated local households seeking basic sustenance. In such circumstances, the presence of social disturbance may foster increased poaching because the probability of being caught declines and the local opportunities for alternative income are reduced, as we have captured in our paper.

How best to raise the opportunity costs of poaching? This can be achieved by creating better alternative economic opportunities. This is especially important in deterring poaching by local poachers in the Chitwan Valley in Nepal, as they come from the very poor and landless group. In addition to creating alternative economic opportunities, incentives could be provided at the community level through community-development programs that mobilize community support in reducing poaching. Indeed, a number of policy measures have been conceived and implemented in this respect in recent years. The revenue-sharing mechanism with the buffer-zone communities around RCNP is one such policy measure.

Stronger antipoaching efforts also will have an immediate impact in controlling poaching, but these measures are liable to decline in effectiveness over time as poachers respond by changing their behavior and if the rhino population increases in size. As a result, it may be necessary to increase antipoaching effort over time to compensate for these effects. Furthermore, relying only on antipoaching enforcement to control poaching (and to conserve rhinos) could prove financially costly, as a large number of APUs or other similar forms of antipoaching efforts will be necessary to completely control poaching, even in the short run. Moreover, given the recent political and security situation in the country, the control of poaching and conservation of rhinos within and outside the park would have to rely increasingly on APUs alone, and without military (Royal Nepalese Army [RNA]) help. The Maoist insurgency is a national issue, which the park authorities will not be able to control or mitigate. However, they can adapt to the changed circumstances and focus on making other parts of the antipoaching setup efficient to compensate for the reduced RNA activities.

Finally a concluding note on antipoaching policies in Nepal, in light of our research and recent policy changes. It is clear that the favored antipoaching policy in RCNP has been the fines-and-fences approach since the establishment of the park. This approach has had some success over the 30-year period, but this success has not been sustained. One of the major aims of this study was to demonstrate that there are other factors that could be important determinants of poaching and, if given attention along with antipoaching enforcement, could help rhino conservation. Indeed, our results demonstrate that antipoaching policies need to be supplemented with policies that provide greater economic incentives to the communities in the park buffer zone for rhino conservation to be successful in the long run.

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