# Conservation risks: When will rhinos be extinct?

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## Abstract

Development of driver-based scenarios of species extinction risks is in its infancy. For many species, the dynamics of anthropogenic impacts driven by economic as well as non-economic values of associated wildlife products along with their ecological stressors can help meaningfully predict extinction risks. Rhinos epitomize these challenges with a key question: When will rhinos be extinct? Extinction is complete conservation failure, collapse of traditional Asian medicinal use, loss of income to non-government organizations, and short-term profit for illegal traders. For rhinos, extinction is in the control of humans. We develop an agent-based economic-ecological model that captures these effects and apply it to the case of South African rhinos. Our model use observed rhino dynamics and poaching statistics. It seeks to predict rhino extinction under the present scenario. This scenario has no legal horn trade, but allows live African rhino trade and legal hunting. In addition, rhinos have high ecotourism value and stimulate a vibrant South African wildlife industry. Rising Asian demand for horn associates with economic well being of eastern countries. Rising demand also introduces lengthy demand reduction strategy lag effects. Present rhino populations are small and threatened by a rising onslaught of poaching. This present scenario and associated dynamics predicts continued decline in rhino population size with accelerated extinction risks of rhinos by 2036. Our model supports the computation of extinction risks at any future time point. This capability can be used to evaluate the effectiveness of proposed conservation strategies at reducing a species' extinction risk.

Acknowledgement: Travel for Timothy C. Haas was supported by a SANParks Visiting Scholar grant.

keywords: Wildlife trafficking, extinction risk, agent based economic models, ecological modeling, individual based models

# I. INTRODUCTION

The extinction of species carry several risks to society [1]. Biological diversity provides numerous services to humans [1], most of these non-tangible and hard to quantify [1]. Conservationists, thus seek to minimize extinction risks because biological diversity provides ecosystem resilience [2], and human quality of livelihoods associate with system resilience

[3]. The material value basis of most socio-economic-ecological systems [4], however, reduces conservation outcomes to a basic price judgment. If a species pays, it stays [5].

Wildlife trafficking of charismatic mammal products, fueled by Asian demand, poses significant threats to biodiversity persistence. International trade bans may result in poaching-fed illegal supply chains because high demand and low supply stimulate high commodity prices attractive to organized crime [6]. Some conservationists argue that the reliance of a species' persistence on its economic value is the basis of its recovery from near extinction (e.g. the Vicuña [7]).

Rhinos are facing extinction risks [8], largely because rhino horn is of high value to Asian societies for several cultural reasons [9], [10, ch. 14]. All rhino species' populations dramatically collapsed over the past century [11] with seven extant species and sub-species remaining [12]. Asian rhino species are holding on – barely [8], while some African species have recovered, most noticeable those with primary ranges in southern Africa [12]. Sustainable use proponents argue that recognition of most values of southern white rhinos (Ceratotherium simum simum) and to some extent southeastern (Diceros bicornis minor) and southwestern black rhino (D. b. bicornis) is the reason for recovery [13]. Unprecedented poaching [14] now places the continued recovery of these species at risk.

Reducing the demand for rhino horn [15], protecting rhinos better [16] and providing horn to consumers [17] offer strategic options to combat rhino poaching. Promoters of introducing trade in rhino horn [6], [18] relies strongly on the dependence of a species' existence relating to its economic value. The focal mechanism, however, is a form of central selling organization [6]. This is effectively a legal monopoly replacing or competing with an illegal one. Cost-benefit analyses illustrate strategies that stockpile horn, provides best financial return when the species go extinct [19], [17], [20].

Proponents of trade bans recognize non-tangible commodity values [21] and advocate demand reduction [22] along with intensified anti-poaching tactics [16]. Trade-banners accidentally and unknowingly trade in extinction anxiety, the key source of non-government organization (NGO) funding. Cost-benefit analyses predict that unintended extinction anxiety trade provides best financial return when a species remains highly endangered. The bankers (legal and illegal stockpile traders) and betters (inadvertent extinction anxiety traders) thus challenge the reliance of a species' persistence on its economic value.

When will rhinos be extinct? It is not a trivial question. For conservationists, extinction is complete failure. For Asian users, extinction collapses a medicinal tradition. For betters, extinction degrades income. For bankers, extinction is profitable. For rhinos, extinction is an option in the control of humans. It is within this context that we seek to predict rhino extinction risk and when that may realize.

At present, no legal trade in rhino horn is allowed [23], but trade in live African rhinos, part of which feed the hunting industry [5] is legal. Rhinos also contribute significantly to ecotourism revenue [24] and has stimulated a vibrant wildlife industry in South Africa [25]. Asian demand is rising [26] and associates with the ebb and flow of economic well being of eastern countries [27]. In the short to medium term it is expected that Asian demand for rhino horn may increase [27], [26], introducing lengthy lag effects of demand reduction strategies. Rhino populations are relatively small and it is debatable whether the present conservation asset can provide for the demand of rhino horn [27] even if horn is harvested from live rhinos [6]. The present status quo is characterized by a rising onslaught of poaching on rhinos [14].

We develop an economic-ecological model of the interaction of poachers, their middlemen, legal traders, consumers, and the South African rhino population. We integrate an agent-based economics submodel with an individual-based rhino population model impacted by the actions of the economics submodel. To the best of our knowledge, our model is one of the first to achieve such integration. The stochasticity of our model allows us to compute species extinction risk as the expected value of a loss function where "loss" is defined to be the non-use value of rhinos residing in a protected area [28].

This article is structured as follows. In Section II, we describe the current situation surrounding rhino horn trade and consequent rhino poaching. Then, in Section III, we describe our economic-ecological model of this trade and its impact on the South African white rhino population. In Section IV, we predict extinction risks over a 35 year horizon. In Section V, we compare our model's output to data-based estimates of white rhino abundance and generate predictions of the coupled dynamics of rhino horn trade and rhino abundance. We discuss the implications of our results in Section VI and reach conclusions in Section VII.

# II. SUPPLY AND DEMAND OF WILDLIFE PRODUCTS

Bulte and Damania [29] employ an economic model to find that multiple equilibrium states exist when a legal trade system operates in parallel to an illegal one. Some of these equilibrium states exhibit accelerated poaching leading to the extinction of the species being harvested. These are called *Bertrand equilibrium* states. The opposite of Bertrand equilibrium is *Cournot equilibrium* wherein the higher-priced trader's market share is reduced.

Ferrier [30] derives equilibrium models of the size of price differentials needed for illegal wildlife trafficking to take place. These models refer to the situation wherein a country has issued a trade ban that makes it illegal to harvest a wildlife product in that country. Ferrier [30] also models the effects of the smugglers' level of risk aversion, the probability that a smuggler will be caught and penalized, and the price elasticity of demand for the wildlife product.

The definition of price elasticity of demand is the percentage change in quantity demanded divided by the percentage change in price [31]. It has been observed that doubling rhino horn price has little to no effect on the demand for it [32]. In other words, the demand for rhino horn is inelastic. There is some evidence [27], [33] that the demand for rhino horn is about four times the amount that is actually sold. Hence, it is important to distinguish between what the total demand for rhino horn is versus the portion of that demand that is satisfied.

## A. Competition

We consider three products traded in three largely separate markets: (1) horn for Asian consumers, (2) live rhinos for the South African recreational hunter market, and (3) the international market for satisfying global anxiety about the future of biodiversity. We refer to the third market as the Species Extinction Anxiety Reduction (SEAR) market.

The last market is served by private firms and NGOs, hereafter referred to as simply SEAR traders. It is in the interest of SEAR traders to amplify and keep in the media the idea that the rhino is headed for extinction due to poaching. In other words, if rhinos cease to be endangered, the global feeling of anxiety towards the future of rhinos would be reduced thus reducing the demand for the service SEAR traders are selling (anxiety reduction).

There are three consumer groups: horn consumers in Asia, donors to SEAR traders, and recreational hunters of rhinos. There is little overlap between these groups. Legal and illegal traders would engage in direct competition if consumers of rhino horn were able to choose between illegal and legal horn.

## B. The Nature of the Illegal Rhino Horn Trade

Illegal traders have bribery costs but no supply maintenance costs, no taxes, no regulation costs, and no labor union costs. Generally then, their overhead costs can be lower than legal traders. And they do not reinvest any of their profits in growing or maintaining their supply so that their profit margins can be larger than that of a legal trader [34]. Crime syndicates pay a small sum [35] to poor, rural people who have limited economic opportunities [36] thus almost guaranteeing an illegal supply of rhino horn.

In a review and critique of the literature on the coexistence of legal and illegal rhino horn trade, Campbell [37] does not find compelling arguments or evidence pointing towards a legal rhino horn trading scheme driving illegal rhino horn traders out of business. There is some reason to believe that competition may actually increase poaching (e.g. [29]). Due to the potential complexity of side-by-side legal and illegal rhino horn markets, any economic model of a competing legal and illegal horn trade needs to account for several elements.

The first is recognizing imperfect competition - organized crime continues to have a near monopoly on tradable horn. Organized crime can thus manipulate supply in order to force higher prices. Second, demand is so great it is mostly inelastic to supply. In addition, poachers will conduct poaching raids for very small wages because there are almost no other competing labor sectors open to them. Therefore, as long as the criminal network can sell horn, they will most likely continue to sponsor poaching raids. The reality is that criminal networks have few rules. In contrast, legal traders have maintenance costs and transaction costs that are substantially higher than what illegal traders have.

# III. THE ECONOMIC-ECOLOGICAL MODEL

Source code for the economic-ecological model (available at [38]) captures a model that consists of two interacting, stochastic submodels: an agent-based model of competing traders modified from a model developed by Catullo [39], and an Individual Based Model (IBM) [40] of a wildlife population modified from a model developed by Kostova, Carlsen, and Kercher [41].

## A. Applying Agent-Based Economic Models to Wildlife Trade

An agent-based economic model represents individual firms as agents and individual consumers as agents. During one step or cycle, each trader makes decisions about product re-supply and product pricing that maximizes their individual utility. Also during this cycle, each consumer makes decisions about entering a market, and once entered, purchasing decisions that maximize their individual utility. Time is incremented, and another cycle is executed [42], [43], [44].

Building on Catullo [39], we construct an agent-based submodel of the international trade in rhino poaching goods across three markets. Our submodel contains a criminal network involved in illegal rhino horn trafficking, a firm involved in seeking to trade legally in horn, the effect of a meta-firm serving the international SEAR market, and the effect of a local, South African meta-firm serving the rhino hunting market.

Arthur [45] finds that an agent-based economics model is able to distinguish among multiple equilibria: a feat that is difficult for models formed from the equilibrium solutions of systems of differential equations. The suspected existence of multiple equilibria in the dynamics of wildlife products trade [29] is possibly the central reason for the reluctance that non-government organizations and international convention secretariats such as CITES have towards the legalization of trade in wildlife products from endangered species. In essence, these agencies suspect multiple equilibria and have no assurance that reality will not settle into an equilibrium point of a species' extinction.

Arthur [45] also notes that agent-based economics models can model the effect that trader expectations can have on future product supply. An example of this in the present application is where illegal rhino horn traders expect to be undersold once legal horn trading is enacted - leading them to accelerate their poaching activities to maximize their profits before being forced out of the market [19].

In another review and critique of the literature advocating legal trade in wildlife products [46], the authors find many articles reaching conclusions based on analyses of static models. The authors see this as inadequate as such models cannot shed light on how wildlife trade markets might unfold through time. Our agent-based submodel on the other hand, is a fully dynamic approach. The authors are also critical of the assumption of a downward sloping demand curve present in all pro-trade articles. Recent theoretical results, specifically the Sonnenschein-Mantel-Debreu theorems (see [46] have shown that a *market demand* curve need not share any characteristics of an individual's demand curve. Hence, any theory that assumes aggregate behavior is a simple scaled function of individual behavior is theoretically invalid. Again, our agent-based submodel allows aggregate behavior to emerge from the interacting actions of many individual consumers.

## B. Integrating Economic Behavior and Wildlife Dynamics

In our approach, an IBM [41] is employed to represent the South African rhino population as they are impacted through time by their birth process, natural death process, and the poaching process produced by the agent-based economics submodel of the legal and illegal traders (Fig 1).

In this model, the traders' submodel runs every 12 weeks and produces $m$, the number of rhinos to poach each week for the next 12 weeks. Then, the rhino IBM runs every week for 12 weeks. Each week, $m$ mature rhinos are randomly selected and set to the value *dead*.

## C. Traders as Agents

Note that in the present state of rules, legal rhino horn traders are only seeking to obtain permission to trade, but are not trading any rhino horn.

### i) Rhino horn traders

In our economic submodel, there are two traders, one legal and one illegal. There are several levels of middlemen involved in the illegal rhino horn trade [47]. We model these as one meta-firm, i.e., we model the middlemen that directly purchase rhino horns from poachers up through the exporters as working for a single firm: the illegal trader. We argue that a criminal middleman has a restricted number of potential customers: other criminal middlemen or criminal exporters. Hence, the collection of middlemen up through the exporter acts more like a cooperative than a set of competing firms. Implicit profit sharing occurs as a middleman at one level will only be willing to purchase rhino horn from a lower level middleman if that price allows the middleman to make a profit. Ultimately, the ability of this cooperative to make a profit depends on the price demanded by poachers and the black market price that consumers are willing to pay. As long as the unit cost to this cooperative is lower than consumers' reserve price, the illegal trader will stay in the business of rhino horn trafficking – lowering or raising their black market price in response solely to the purchase decision making of consumers. Therefore, some estimate of an illegal trader's unit cost is needed.

In our economic submodel, the unit cost for acquiring and selling one kilogram ($kg$) of rhino horn by either trader is \$5,000. For the illegal trader, this number is arrived at by considering that trader's costs as follows. First, the illegal trader needs to purchase a horn from a poacher. In Eloff and Lemieux [48, p. 21], the black market price for one kilogram of rhino horn is estimated to be between USD \$35,000 and \$60,000 with about 5% of that being used by the illegal trader to purchase the rhino horn from poachers. Using the lowest black market price, poachers are paid \$1,750 for one kilogram of rhino horn. Next, the illegal trader needs to purchase a courier's airfare from Maputo, Mozambique to some city in Asia for \$2,000. Finally, the illegal trader needs to pay the courier's fee of \$500 [49] per rhino horn or \$100 per kilogram of rhino horn assuming an average rhino horn weighs about $5kg$ [50]. Using these numbers, the trader has incurred a cost of \$3,850 to bring one kilogram of rhino horn to an Asian market. Thus, a conservative unit cost is \$5000.

As mentioned in Section II, the literature speculates that the legal and illegal traders may settle into an equilibrium state wherein the illegal trader pays to accelerate the poaching rate. Our model is constructed so that either trader will continue to produce their product (through poaching for the illegal trader) as long as their unit cost is less than what they can sell the product for. Therefore, our model's output (not shown) has the two competing traders settling into an equilibrium state wherein the traders both quickly drop their prices to their respective unit costs. If one trader's unit cost is higher than the other's, that trader is driven out of the market. This is not Bertrand equilibrium because neither trader is strategically over-producing. And it is not Cournot equilibrium either because neither trader is voluntarily offering fewer products for sale.

Traders are not allowed to engage in product "dumping," i.e., sell their rhino horns for less than their unit costs. Each week, traders always sell as many kilograms of rhino horns as there are consumers willing to purchase them. In other words, demand is insatiable [27].

Vietnamese rhino horn merchants usually have a number of rhino horns available for inspection [34]. This implies that (a) there is no direct order placed by a customer before a rhino is poached, and (b) illegal traders maintain a buffer stock (inventory) of rhino horn.

### ii) International SEAR Trade

Poaching frequency is a proxy for the amount of international anxiety about looming rhino extinction - a special case of species extinction anxiety defined in Section II.B. Consumers wish to reduce their amount of this anxiety. Conservation-focused NGOs solicit donations by promising to help curb rhino poaching. In effect, these NGOs are selling anxiety-reduction aids [51]. A SEAR trader's revenue is driven by the demand for their product which in turn is driven by the amount of rhino poaching perceived by the international community. If the perceived amount of poaching lessens, a SEAR trader's revenue tends to lessen and vice versa. SEAR traders do indeed fund a portion of anti- poaching measures.

Therefore, if rhino poaching is reduced, external funds for anti-poaching measures are reduced. This effect is modeled in the agent based submodel by weakly tying anti-poaching effectiveness to the number of rhinos poached per week (see Section III.C.iii, step 10). The economic submodel thus does not directly simulate SEAR trader transactions with their customers.

### iii) South African Trade in Rhino Tourism and Rhino Hunting

Tourism is tied to charismatic species [24], one of which is the rhino. Tourism experiences, however, are complex and rhino specific contributions may be minimal. It is more likely that militarization associated with anti-poaching activities [52] influence tourism experiences. Militarized anti-poaching degrades the sense of place of protected areas [53] a key societal value [54]. Rhinos, at best, thus have weak indirect effects on revenue generated to trade in rhino tourism. This possible effect is thus not explored in this article.

The effect of recreational hunting of rhinos on private ranches (hereafter ranches) is modeled in the rhino abundance submodel (see Section III.D.ii, step 8). Economic transactions between these ranch owners and recreational hunters are not modeled in the economic submodel.

### iv) Submodel Operation

1. Compute the expected new price. First compute the response to the price goal function:

$$\texttt{q-response}_{t-1} = \texttt{price}_{t-1}\left[1 - \frac{\texttt{capacity}_{t-1} - \texttt{nmsold}_{t-1}}{\texttt{capacity}_{t-1}}\right]. \tag{1}$$

   To see "what the market will bear" (see [55]),

$$\texttt{q-response}_{t-1} = 1.01\texttt{price}_{t-1} \text{ if } \texttt{capacity}_{t-1} = \texttt{nmsold}_{t-1}. \tag{2}$$

2. Compute the expected value of the new price:

$$\mu_t = (1 - \texttt{learn-rate})\mu_{t-1} + \texttt{learn-rate} \times \texttt{q-response}_{t-1}. \tag{3}$$

3. The new price is found by sampling once from a normal distribution with mean $\mu_t$ and a standard deviation of \$200.

4. The net revenue is:

$$\texttt{netrev}_t = \texttt{nmsold}_{t-1}(\texttt{price}_{t-1} - \texttt{unit-cost}). \tag{4}$$

5. The response to the production capacity goal function is:

$$\texttt{c-response}_t = \texttt{netrev}_t - \texttt{netrev}_{t-1}. \tag{5}$$

6. The production capacity decision constant is:

$$\texttt{q-prodcap}_t = (1 - \texttt{learn-rate})\texttt{q-prodcap}_{t-1} + \texttt{learn-rate} \times \texttt{c-response}_t. \tag{6}$$

7. The production capacity decision Binomial distribution probability is:

$$p_c = \frac{\texttt{q-prodcap}_t + \texttt{maxnetrev}}{2 \times \texttt{maxnetrev}}. \tag{7}$$

8. Production capacity is reduced, left unchanged, or increased according to the following rules. First, let $D$ be a binomially distributed random variable with $n = 2$, and probability of success equal to $p_c$. Sample once from this distribution. If $d = 0$, $\texttt{prodcap}_t = \texttt{prodcap}_{t-1} - 1$. If $d = 1$, $\texttt{prodcap}_t = \texttt{prodcap}_{t-1}$. If $d = 2$, $\texttt{prodcap}_t = \texttt{prodcap}_{t-1} + 1$ up to this trader's maximum production capacity. Both traders have a maximum production capacity of $150kg$ of rhino horns per week. Because the horns from an adult rhino weigh approximately $5kg$, this value represents 30 rhinos per week. In 2013, an average of 20 rhinos were poached per week across South Africa [56]. Hence, this maximum is ten rhinos above the 2013 weekly average.

9. Reduce the production capacity of the illegal trader in proportion to the effectiveness of anti-poaching operations as follows. Let $N_p$ be binomially distributed with $n = \texttt{prodcap}_t$ and probability of success equal to $p_a$. The probability $p_a$ is set to a number close to 0.0 if anti-poaching operations are very effective at curbing poaching. Sample once from this binomial distribution to find $n_p$, the actual number of rhinos poached this week in spite of anti-poaching operations.

10. Model the effect of additional anti-poaching funds donated by SEAR traders by reducing $n_p$ by 5% if $n_p$ is greater than 25.

11. Model the effect of population growth on the Asian continent on the number of potential rhino horn consumers. Most consumers of rhino horn live on the Asian continent [26]. Table 1 contains population projections found in [57].

The initial consumer population is created as follows. To represent the assumption of insatiable demand at current (illegal) production levels, consumers are created as necessary to purchase all rhino horn poached under the maximum poaching rate of 30 rhinos per week across South Africa (20 in KNP, and 10 on the ranches). Because each rhino horn weighs on average $5kg$, these numbers are multiplied by 5. Therefore, in the year 2014, the potential number of consumers is set to 300 ($5 \times 60$). This value is increased in proportion to the entries in Table 1 to a maximum of 325 in the year 2033.

For the case of a legal trading scheme operating in parallel to the illegal trade, this consumer pool is doubled. Because demand for rhino horn in the near future is predicted to be about four times current sales [27], doing so is well-within current demand forecasts. The supply of legally-traded rhino horn would be sourced from stockpiles and/or shavings from live rhinos.

By a "consumer" we mean a group composed of a number of real-life individuals. Guilford [58] reports an individual purchase for $2000 of rhino horn powder. At the per-kilogram prices mentioned above, this would be between 33 and $57g$ of rhino horn. Other individuals may purchase other amounts of rhino horn. In our model however, one of our "consumers" always buys exactly one kilogram of rhino horn at each purchase event. Hence, one of our "consumers" represents approximately 18 to 30 real-life individuals. By doing so, we ignore the variability in the amount of purchased rhino horn and in-effect, lump approximately 18 to 30 real-life purchase events into one purchase event. Hence, our purchase event time series shown below should be viewed as the aggregate behavior of groups of approximately 18 to 30 real-life individuals.

12. Consumer behaviors start with the decision to enter the rhino horn market or not. If there is a media campaign aimed at potential rhino horn consumers that delivers a message that rhino horn has no medicinal value, some of the potential consumers may decide to not try to purchase rhino horn. This media campaign effect is represented as follows. Let $n_{pc}$ be the number of potential consumers each week. Let $p_m$ be the effectiveness of a horn-is-not-medicine media campaign run in the country where the consumers live. If $p_m$ is close to 1.0, the chance that a randomly chosen potential consumer will decide to buy rhino horn is close to zero. Let $N_c$ be binomially distributed where there are $n_{pc}$ trials, and the success probability is $1-p_m$. Sample once from this distribution to find $n_c$, the number of consumers for that week who enter the market for rhino horn.

13. Simulate rhino horn purchases. Each consumer buys one kilogram of rhino horn from the trader offering it at the lowest price as long as this price is below the consumer's

reserve price of $60,000 [14]. Because the illegal trader maintains a buffer stock of rhino horn, the number of kilograms of rhino horns the illegal trader sells each week need not equal five times the number of rhino horns poached the previous week.

## D. An Individual-Based Model of the Rhino Meta-Population

An IBM of animal abundance is valid for any range size and number of animals when data structures and mapping functions are suitably developed [40, ch. 4]. But a differential equation model of the animal's population dynamics (see [59]) may, depending on the assumptions that underlay its derivation, need relatively larger range sizes and initial abundance values for it to be a faithful representation of actual population dynamics. Within ranches however, rhino abundance and range are often small.

An important characteristic of this habitat is that rhino are artificially restricted to anthropogenically-defined patches which in this case are those within the subregions of Kruger National Park (KNP) and ranches. An IBM can be developed to accurately represent the effects of these restrictions on the dynamics of the within-patch populations. The ability of IBMs to handle complex habitat-use conditions is one reason given by McLane, Semeniulk, McDermid, and Marceau [60] for why IBMs should be used to model managed wildlife populations.

A spatially-explicit submodel of the South African rhino meta-population is built as opposed to a non-spatial, aggregated single population submodel for the following reasons:

1. Different tolerances for risk across ranch owners can be modeled. For example, a ranch owner might offer the opinion: "I won't keep rhinos, too risky."
2. Ranch-specific financial returns for keeping rhinos can be modeled.
3. Spatial effects on the amount of available forage can be modeled.
4. Spatially-heterogeneous anti-poaching effectiveness can be modeled.
5. Spatially-heterogeneous poaching pressure can be modeled.
6. Rhinos are highly territorial [61]. A spatially-explicit IBM is flexible enough to realistically capture all aspects of this behavior.

An IBM for the South African rhino meta-population is developed along the lines of the prairie vole (Microtus ochrogaster) IBM of Kostova, Carlsen, and Kerche [41]. As with the prairie vole IBM of [41], the rhino IBM is stochastic in that one run over a time period will not necessarily produce the same history of abundance and dispersal as another run over the same time period. For this reason, many replications of the IBM over the same time period are needed so that at each time point, both the expected value of abundance and extinction probability can be computed.

### i) Rainfall Predictions and Available Vegetation

Rainfall predictions for KNP over the simulation interval (Figure 2) are found by evaluating a mathematical model that has been statistically fitted to rainfall observations. Rainfall data from the years 1903 through 2013 contained in the SANParks data repository [62] are used to estimate the parameters of a neural network time series model that includes the effect of the El-Niño-Southern Oscillation (ENSO) phenomenon [63]. A logistic transform [64] to a quasi-periodic function formed from the product of three cosine functions [65] is used to model the ENSO phenomenon.

The rainfall predictor at time $t$ is constructed as follows.

1. The quasi-periodic function is

$$q_t = \prod_{i=1}^{3} \cos\left[\frac{\pi t - \phi}{\gamma_i}\right],\ i = 1, \ldots, 3 \tag{8}$$

   where $\phi$ is the phase, $\gamma_i$ is the period of the $i^{\text{th}}$ component, and $\zeta$ is the amplitude.

2. The logistic function is:

$$m_t = \left[1 + \exp(-\beta(\zeta q_t - \alpha))\right]^{-1}. \tag{9}$$

3. A neural network nonlinearly transforms $m_t$ to produce a rainfall prediction:

$$h_{ti} = (1 + \exp(-\omega_{1i}\texttt{year}_t - \omega_{2i}\texttt{week}_t - \omega_{3i}\texttt{weeknm}_t - \omega_{4i} m_t - \omega_{5i}))^{-1}, i = 1, \ldots, n_h \tag{10}$$

   where $n_h$ is the number of "hidden rows" in the neural network, $\texttt{week}_t$ is the week counter starting at week 1 in the year 1903, and $\texttt{weeknm}_t$ is the number of the week within the year and takes on the values $1, \ldots, 52$. The neural network's output layer consists of a single variable:

$$o_t = \left[1 + \exp\left(\nu_{n_h+1} + \sum_{i=1}^{n_h} \nu_i h_{ti}\right)\right]^{-1}. \tag{11}$$

The parameters of this model are $\phi$, $\gamma_1$, $\gamma_2$, $\gamma_3$, $\beta$, $\zeta$, $\omega_{ij}$, $i = 1, \ldots, 5$, $j = 1, \ldots, n_h$, and $\nu_i$, $1, \ldots, n_h + 1$. Least squares parameter estimates are found with nonlinear optimization.

Rainfall either observed or predicted is used as a scaled proxy of available vegetation. A scaling constant is selected so that approximately 25% of the population experiences a food deficit during the dry season [66].

Specifically, a value of $c$ is found such that

$$0.75 = \frac{c \times a \times v}{12000 \times \boldsymbol{wfi}} \tag{12}$$

where $a$ is the area of KNP, $v$ is unscaled available vegetation for week $i$ set equal to the 0.91 quantile of the observed rainfall observations from 1910 to 2012, and $12,000$ is the desired (target) value of rhino abundance in KNP. Week $i$'s new vegetation per square kilometer is computed from the week $i$'s rainfall with $veg_i = c \times r_i$ where $i = 1, \ldots, m$ and $m$ is the number of weeks in the observation or prediction interval.

At any point in time, the available vegetation for a rhino's food supply is no more than 36 weeks old. To model this, the net vegetation in a week is set to the sum of the left-over vegetation from the past 36 weeks. This moving-window sum is initialized by setting the first week's net vegetation to four months of a representative value of weekly new vegetation. Specifically, the first week's available vegetation is set to 16 times the 0.99 quantile of the $veg_i$, $i = 1, \ldots, m$ values found from the observed rainfall series.

### ii) Submodel Operation

The IBM executes the following schedule of actions each week.

1. Delete all rhinos set to *dead* during the previous time step.
2. Find within-patch populations.
3. Increment each rhino's age.
4. Up to a rhino's mean energy budget (*meb*) or juvenile energy budge (*jeb*) value, a rhino's energy budget is updated in the following manner:

   (a) Compute the *vegetation ratio*:

   $$\texttt{vratio} = 0.01 \left[ \frac{\texttt{netveg}_t}{\boldsymbol{wfi} \times \texttt{nmindiv}_t + 1} - 1 \right] \tag{13}$$

   where $\boldsymbol{wfi}$ is a rhino's weekly food intake, $\texttt{nmindiv}_t$ is the number of patch residents at time $t$, and $\texttt{netveg}_t$ is the available vegetation within the patch at time $t$.

   (b) Compute the amount of energy change:

   $$\texttt{ec} = \frac{2}{1 + \exp(-\texttt{vratio})} - 1. \tag{14}$$

   (c) If $\texttt{netveg}_t < \boldsymbol{wfi} \times \texttt{nmindiv}_t$, do the following for each patch resident. First, sample once from $V$, a random variable uniformly distributed over the unit interval to obtain $v$. Then, if $v < 0.4$, $\texttt{energy}_t = \texttt{energy}_{t-1} + \texttt{ec}$.

   (d) If $\texttt{netveg}_t > \boldsymbol{wfi} \times \texttt{nmindiv}_t$ then for each patch resident, $\texttt{energy}_t = \texttt{energy}_{t-1} + \texttt{ec}$.

(e) For each rhino having $\texttt{energy}_t = 0$, draw a realization from $V$ to obtain $v$. Set this rhino to *dead* if $v < 0.1$.

5. Set to *dead*, any rhinos having an age greater than $le$.

6. Simulate food deficit and animal density effects on birth and mortality rates. Ferreira, Greaver, Knight, Smit, and Pienaar [66] finds that juvenile mortality rates may be higher if the previous year's rainfall was low. Rainfall is used within this model as a proxy for food availability. Food scarcity affects the population through three mechanisms.

   At moderate levels of scarcity, juveniles are affected when they die because their energy budget has fallen to zero. At high levels of scarcity, birthrates are affected by an increases in the intercalving interval and an increase in a female's maturation age. Specifically, within a patch, if 50% of the females experience food stress in a particular week, all females in that patch have their maturation age and intercalving interval increased by 50%. These values are interpretations of evidence reported in Ferreira, Greaver, Knight, Smit, and Pienaar [66]. At extreme levels of scarcity, all adult rhinos are affected through the increased chance of mortality due to their energy budgets going to zero.

   Rachlow and Berger [67] report that both the age at first calving and the intercalving interval are significantly increased under high spatial density of rhinos due to the consequent social stress. Within a patch whose density rises above 3.2 animals per square kilometer, this effect is modeled here by having both the intercalving interval and female maturation age increase 50%.

   In reality, food scarcity and spatial density take some unknown amount of time to affect the maturation age and intercalving interval of female rhinos. Our view is that ignoring these time delays should not significantly affect our submodel's long-term abundance trends which is the focus of this modeling exercise.

7. Process poaching actions. Read $m$, the weekly number of rhinos that are to be poached. Randomly select $m$ mature rhinos and set them to *dead*.

8. Legal hunting on ranches . The hunting off-take from the ranch population is 50% of the oldest males annually. An "old male" is defined to be a male older than the 90th percentile of male ages on the ranch. Find these individuals as follows.

   (a) Sort all male ages, and then locate the 90th percentile age.

   (b) Form a group of males older than this threshold age. Say there are `nm-old-males` individuals in this group.

(c) If `nm-old-males` is positive, compute `nmhunt`, the number of individuals to hunt (kill) each week with $\texttt{floor}(0.5 \times \texttt{nm-old-males}/52)$. Otherwise, set `nmhunt` to zero.

(d) Randomly select `nmhunt` individuals from the old male group and kill them.

9. Sell some ranch rhinos. This off-take is from all age classes and both genders. Each year, one-fourth of the exponential growth rate is removed from ranches. The exponential growth population model is $N_t = N_0 \exp(rt)$ where $N_t$ is abundance at the end of the time interval, $t$ (measured in years), $N_0$ is the initial population size, and $r$ is the exponential growth rate. Then, for a given $r$, the selling off-take each week is $0.25r/52$.

10. For each mature female rhino, create one new rhino if (a) its $\texttt{time-since-last-birth}_t$ is greater than *intercalv*, (b) some males are residents of the female's patch, and (c) the female's energy is greater than *meaneb*.

11. For each female not giving birth,

$$\texttt{time-since-last-birth}_t = \texttt{time-since-last-birth}_{t-1} + 1. \tag{15}$$

12. Update patch membership by randomly moving rhinos into different patches within subregions that possess nonzero net vegetation.

13. Update the net vegetation of each patch. First, find the amount of new vegetation at this time point from the above set of vegetation predictions. Second, find the amount of left over vegetation at this time point as

$$\texttt{vegleftover} = \texttt{netveg}_t - \mathit{wfi} \times \texttt{nmindiv}_t. \tag{16}$$

Finally, sum these values of left over vegetation across the previous 36 weeks. If this sum is negative, reset it to zero.

### iii) Submodel Parameter Values

Table 2 gives the population dynamics parameters and their values used in the simulations. There are two subregions (KNP and ranches) each with four patches. The initial age distribution is gaussian with a mean of 7.5 years, and a standard deviation of 3 years truncated between one week old and the life expectancy of a rhino which here, is 38 years [74] (see Table 2).

# IV. COMPUTING THE RISK OF EXTINCTION

What is society's loss function as a function of the time at which a species becomes extinct? Denote this function with $L(t_e)$ where $t_e$ is the time at which the species first becomes extinct. Note that $L(t) = 0$ for $t < t_e$. The loss due to the extinction of a species residing in a protected area can be approached through its *non-use value* [28]. Non-use value is the sum of the species' *bequest value* and *existence value*. Existence value is the value of knowing that a species exists, and bequest value is the value of conserving a species for future generations (see [69]). As non-use value is unitless, we choose to define it over the unit interval. For those members of a species living in a protected area, there is usually no *use value*, e.g. harvesting the species for its economic value. Let $V(t)$ be the non-use value of the species at time $t$. Note that $V(t) = 0$ for $t \geq t_e$. Let $L(t) = V(t - \epsilon)$ where $\epsilon$ is a small positive number.

Say that at $t = 0$, the non-use value of the non-extinct species is $V_0$. Under the assumption that this value is constant across future time, $L(t_e) = V(t_e - \epsilon) = V_0$ for $t_e \geq \epsilon$. If however, future value is discounted (*time discounting*), $L(t_e) = V_0 D(t_e)$ where $D(t)$ is a discounting function. A standard approach to discounting the cost of extinction in the future is with an exponential discounting function, $D(t) = (1 - d)^t$ (see [70]). Setting $d = 0.035$ is not unusual.

A typical definition of *risk* used in environmental protection is the expected value of loss [71]. Mathematically, $R(t) = E\left[L(t)\right]$. Because $L(t)$ equals zero if the species is not extinct, and takes on a positive value otherwise,

$$R(t) = L(t)P(\texttt{species first becomes extinct at time}\, t). \tag{17}$$

We use $V_0 = 1$, $d = 0.035$, and extinction probabilities computed from our economic-ecological model to compute local extinction risks over the period 2014 through 2045 (Figure 4).

# V. MODEL OUTPUT

## A. Submodel Output Compared to Survey Estimates

Ferreira [68] reports on estimates of KNP rhino abundance based on surveys conducted between 1998 and 2012. Table 3 indicates a good fit of the IBM submodel to these estimates. Figure 2 contains a prediction of rhino abundance over the period 2014 to 2033 under no poaching in either KNP or the ranches but with ranch-based recreational hunting, and ranch-sourced removals. The 2014-2033 time period is the same over which the status quo strategy responses will be computed in Section V.B. This plot indicates that with no poaching over this period, the South African rhino population is robust and increasing.

## B. Simulating the Effects of the Present Rhino Management Strategy

The economic-ecological model may be used to predict rhino abundance and the behavior of the rhino horn market under different management strategies. One such strategy is that of continuing current management practices (current levels of anti-poaching enforcement, no changes to the current set of laws controlling trade in wildlife products and continued increases in demand for horn). Call this the status quo strategy. To assess the effects on future rhino abundance under this strategy, the model is run over a 20 year period: from January 1, 2014 through January 1, 2034. The 20 year period allows rhino population dynamics to react to management actions as this interval is approximately three rhino generations. Time series output from this run is plotted in Figure 3. The present scenario predicts consistent decline of rhinos over the next 20 years in both KNP and the ranches. And, because there is no competition, rhino horn purchases are executed at prices that are just below the consumers' reserve price (not shown). The illegal trader quickly reaches a steady state production level that is usually not far from the maximum number of rhinos that can be poached per week.

### Extinction Risk

Under the status quo strategy, probabilities of local extinctions are zero until suddenly climbing around the year 2036 (4) for both the KNP and ranches rhino populations. Because of time discounting this delayed ramp-up of local extinction probabilities results in low, but increasing local extinction risks. Hence, with time discounting, sudden increases in local extinction probabilities that happen around 2036 results in extinction risks that are not alarming in the short to medium term. Because of this phenomenon, extinction risk with time discounting may not be the best information to present when attempting to motivate the public to support an increased focus on conservation. The trends in populations may serve as a better motivation in the short term.

# VI. DISCUSSION

The onslaught on the world's wildlife resources [76] is a central theme in the international arena at present. All extant rhino species are threatened by poaching for their horn [77], [78]. Our modeling of southern white rhinos, the most numerous of the remaining extant species, suggest continuous declines if the present status quo remains for the next twenty years. We also predicted rapid increase in local extinction risks by 2036.

We acknowledge, however, that our predictions may carry some constraints. For instance, our white rhino IBM sub-model derived parameters through comparison with observed trends in the southern white rhino population of Kruger National Park [66] as well as derived estimates for southern white rhinos living outside Kruger National Park in South Africa [77]. Our agent-based economic model uses proxies of poachers, middlemen and consumers to tract anticipated effects of changes in demand for rhino horn in eastern countries [79]. Our retroactive model predictions, however, tract southern white rhino population estimates in Kruger National Park well from 1998 to 2012. We thus argue that these proxies serve as good substitutes of tracking economic dynamics to help predict scenario outcomes.

Although our prediction of extinction suggests that risks only escalate dramatically by 2036, the continued decline of rhinos is a key concern. Although complete extinction is not as urgent, various values associate with rhinos. The trends in predicted populations suggest a gradual degradation of some of those values. The reduction of a conservation asset, such as the predicted decline in rhino populations, introduces vulnerability to environmental as well as stochastic risks associated with small populations [80]. Similarly, Asian consumers may face degradation of a resource highly sought after [81].

The trends in predicted populations, may also suggest a gradual increase of some of the values associated with rhinos. For instance, SEAR traders may gain significantly through increases in extinction risks – extinction anxiety may increase leading to more willingness of the public to fund initiatives that can disrupt the extinction predictions. Traders in rhino horn, at present these are only illegal, may substantially gain value in horns stockpiled in anticipation of extinction [20].

These brief implications suggest that both bankers (traders in rhino horn) [20] and betters (traders in extinction anxiety) [82] may benefit substantially financially if the present status quo with changing demand dynamics persist. Given that the dichotomy of international debate diverges into trade proponents [6], [17] and opponents [46], imposing a banking and betting [20] debate that creates response inertia, the present status quo may be maintained for some time. The world's most abundant rhino will continue to decline in the face of the banking and betting power struggle. Our model of the present status quo and associated dynamics thus seriously challenge the reliance of a species' persistence on its economic value.

Under the present scenario, rhinos have an increased extinction risk by 2036. Who will safe them? The dichotomy of trade versus no-trade has distracted conservationists from considering sensible solutions. Integrated approaches [83] identified parallel initiatives that manage the threat to rhinos as well as enhancing rhino populations through ecological management [66]. The reality is that central to these strategic initiatives [83] is the involvement of transnational organized crime. The disruption of organized crime syndicates poses a key challenge to authorities, and should be of the highest priority.

Organized crime, however, exploits rural communities abutting protected areas [84]. These areas seldom offer economic opportunities other than those based on trading natural resources [85]. Communities living next to protected areas also carry the biggest opportunity costs inflicted by western conservation philosophy [86]. Some of those costs recently escalated when several resourced-based economic opportunities degraded such as those imposed by western and global north bans of hunting trophy imports [87]. These complex drivers thus place rural communities specifically at risk of being exploited by transnational organized crime focusing on rhino poaching. Authorities seeking to disrupt transnational organized crime also need to create economic opportunities for rural communities abutting protected areas.

One particular class of economic opportunity is that associated with wildlife products. It could include rhino-based initiatives. This is particularly attractive as it provides opportunities for authorities to develop economic options that do not fall in banking and betting on extinction strategies. Such initiatives can use dumping strategies [20] that predicts lower economic return, but persistence of rhinos and thus also many values associated with rhinos.

Our agent-based economic model allows incorporation of such scenarios that include complimentary initiatives as proposed before [83]. Predicting the outcomes of such inclusive scenarios can help inform decision makers and remove the inertia imposed by the banking and betting on extinction power struggle [20].

# VII. CONCLUSIONS

We have developed an economic-ecological model of trade in a wildlife product and the effect of that trade on the harvested wildlife population. Our model is realized as an agent-based economic submodel interacting with an individual-based ecological submodel. Computations with this model delineate the difference between the chance of a species' extinction versus its risk of extinction. We have shown by example how the ecological submodel can be validated by comparing its output to data-based estimates of wildlife abundance.

As the rhino horn trafficking example shows, for model output to be reliable enough to inform policymakers charged with evaluating different management strategies for conserving biodiversity, the model needs to incorporate a complex mix of economic and ecological processes. This is typically referred to as *construct validity* [72, ch. 1]. A model enjoying some level of construct validity can then undergo a final test of its relevance to policymaking, that of its ability to reproduce real-world observations, typically referred to as *predictive validity* [72, ch. 1].

An additional requirement of such modeling is required when applied to biodiversity protection. Namely, to be effective, a policy needs to be implemented before extinction risk becomes large. Hence, extinction risk predictions need to be made available to decision makers many years prior to the potential extinction event. The model described in this article provides one way to compute these forecasts. Use of time discounting reduces extinction risk at distant future time points and hence makes risk predictions less powerful for mobilizing the general public to act against biodiversity threats.

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# Tables

| Year | Population Estimate/Prediction |
|---|---|
| 2010 | 4,165,440,162 |
| 2020 | 4,581,523,062 |
| 2030 | 4,886,846,140 |
| 2040 | 5,080,418,644 |

Table 1: Asian continent population projections taken from [57].

| Name | Notation | Units | Value | Source of Value |
|---|---|---|---|---|
| Average Weekly Food Intake | *wfi* | kg | 140 | [73] |
| Life Expectancy | *le* | years | 38 | [74] |
| Maturation Age | *ma* | years | 4 | [75] |
| Maximum Energetic Budget | *meb* | weeks | 5 | after [41] |
| Mean Energetic Budget | *meaneb* | weeks | 4 | after [41] |
| Juvenile Energetic Budget | *jeb* | weeks | 3 | after [41] |
| Intercalving Interval | *intercalv* | years | 2.5 | [75] |
| Available Vegetation | $av(t)$ | g/m2 | (as given in Figure 2) | see Sec. III.D.i |

Table 2: IBM parameters and their values.

| Time | Data-Based Abundance Estimate | Model-Based Expected Abundance |
|---|---|---|
| 1998 | 2674 | 2706 |
| 1999 | 2938 | 3090 |
| 2000 | 2683 | 3401 |
| 2001 | 4552 | 3764 |
| 2002 | 4223 | 4217 |
| 2003 | 4765 | 4841 |
| 2004 | 5308 | 5465 |
| 2005 | 6974 | 5990 |
| 2006 | 8893 | 6704 |
| 2007 | 9119 | 7677 |
| 2008 | 11498 | 8601 |
| 2010 | 10621 | 10929 |
| 2012 | 10495 | 8453 |

Table 3: Estimated and IBM-generated abundance.

# Figures

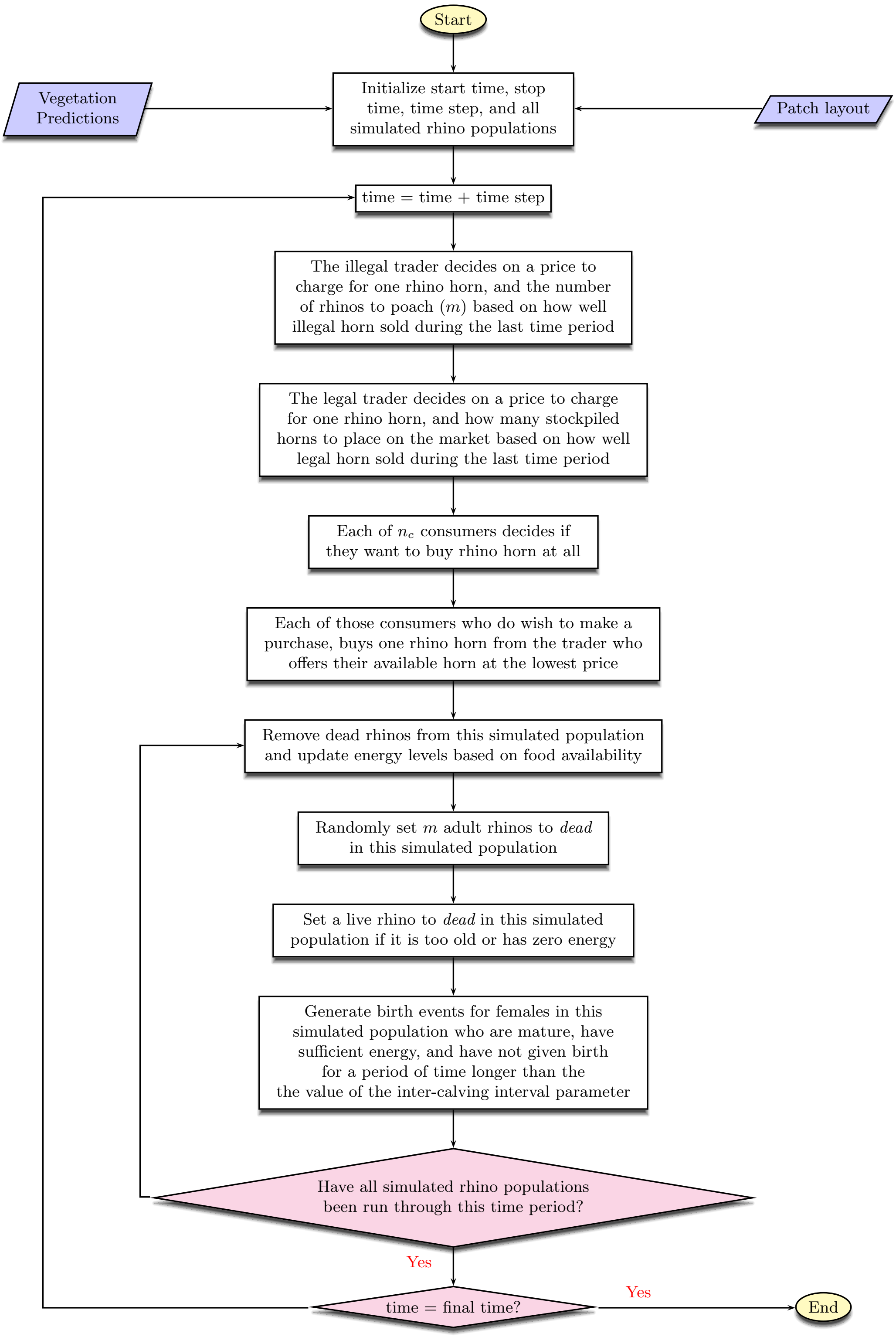



Figure 1: Flowchart of the economic-ecological model.

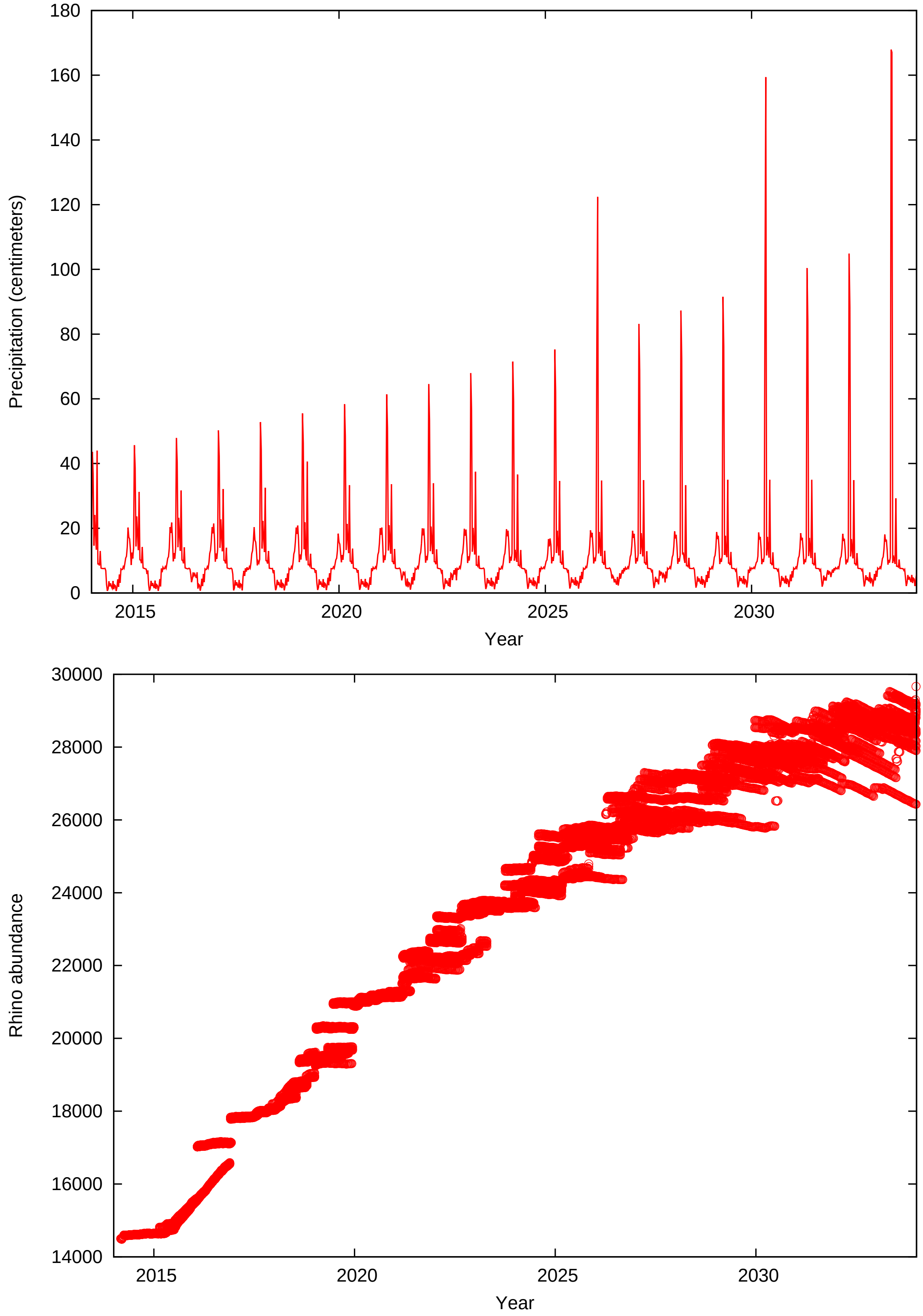



Figure 2: Top: predicted KNP precipitation (proxy for new vegetation) ($cm$); bottom: IBM predictions of rhino abundance under no poaching, ranch-based recreational hunting, and ranch-sourced removals.

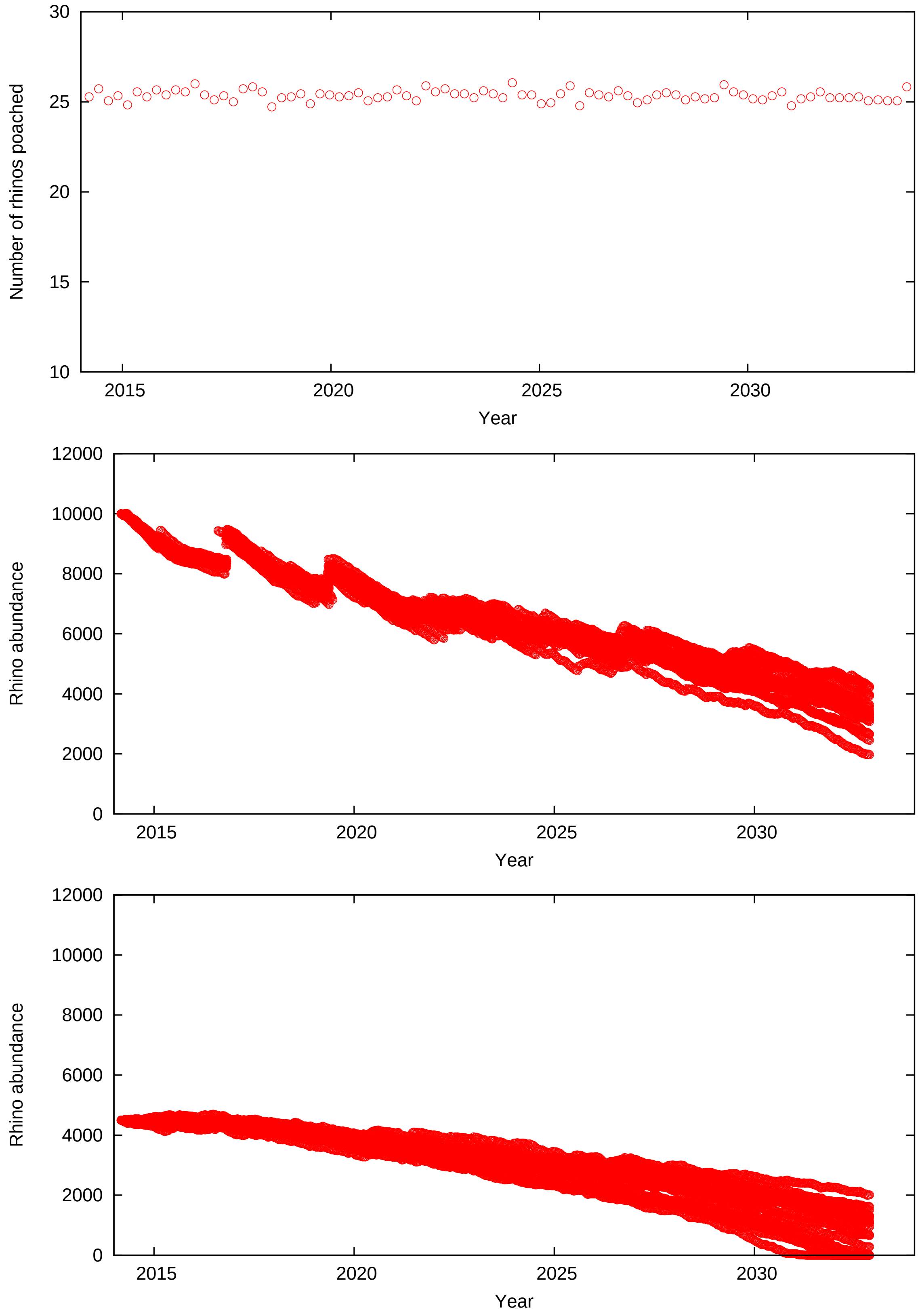
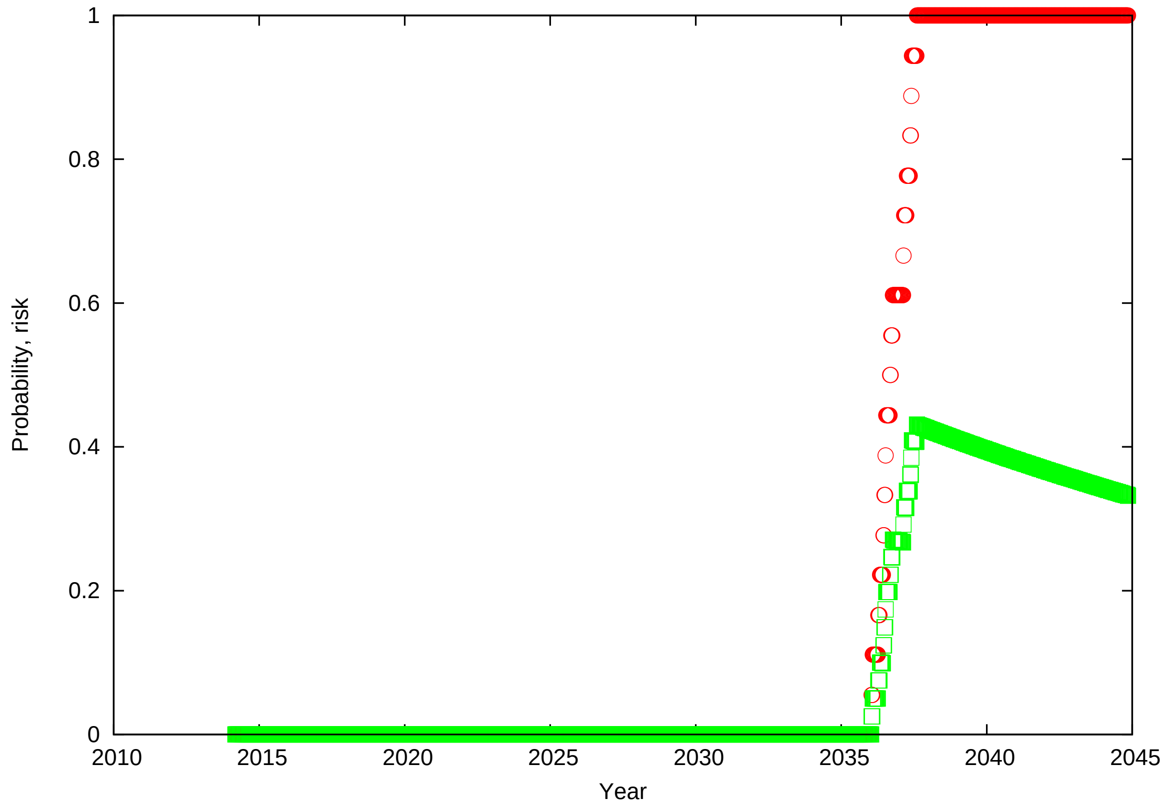



Figure 3: Economic-ecological model time series output under the status quo strategy. Top: number of rhinos poached per week. Second: KNP rhino abundance. Third: ranches rhino abundance.



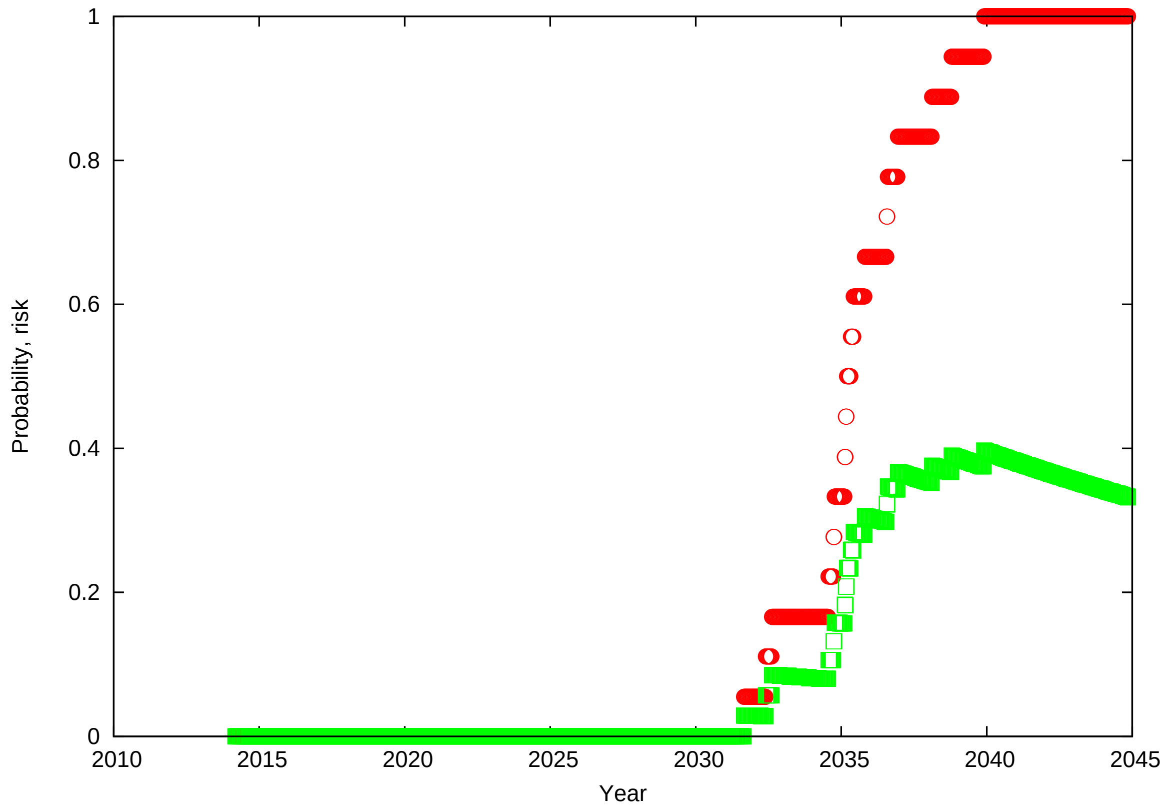




Figure 4: Local extinction probability (circles), and local extinction risk (squares) under the status quo strategy. Top: KNP, bottom: ranches.