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Using return on investment to maximize conservation effectiveness in Argentine grasslands

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The rapid global loss of natural habitats and biodiversity, and limited resources, place a premium on maximizing the expected benefits of conservation actions. The scarcity of information on the fine-grained distribution of species of conservation concern, on risks of loss, and on costs of conservation actions, especially in developing countries, makes efficient conservation difficult. The distribution of ecosystem types (unique ecological communities) is typically better known than species and arguably better represents the entirety of biodiversity than do well-known taxa, so we use conserving the diversity of ecosystem types as our conservation goal. We define conservation benefit to include risk of conversion, spatial effects that reward clumping of habitat, and diminishing returns to investment in any one ecosystem type. Using Argentine grasslands as an example, we compare three strategies: protecting the cheapest land ("minimize cost"), maximizing conservation benefit regardless of cost ("maximize benefit"), and maximizing conservation benefit per dollar ("return on investment"). We first show that the widely endorsed goal of saving some percentage (typically 10%) of a country or habitat type, although it may inspire conservation, is a poor operational goal. It either leads to the accumulation of areas with low conservation benefit or requires infeasibly large sums of money, and it distracts from the real problem: maximizing conservation benefit given limited resources. Second, given realistic budgets, return on investment is superior to the other conservation strategies. Surprisingly, however, over a wide range of budgets, minimizing cost provides more conservation benefit than does the maximize-benefit strategy.

economic cost | benefit:cost ratio | rarity

The precipitous decline in worldwide biodiversity places a premium on the immediate conservation of threatened biodiversity. In some environments, such as that discussed here, the problem is of staggering dimensions. Temperate grasslands are among the least-protected major habitat type on earth—only 5% have been protected globally (1). Less than 0.5% of grasslands in Argentina, which contains virtually all of South America's temperate grasslands, are protected (2). These percentages compare with the goal, widely embraced by government and nongovernment organizations, of protecting 10% (sometimes more) of the area of each country or major habitat type (3).

Conservationists in the field face two conflicting pressures in such situations. First, the 10% goal can best be met by protecting large areas of cheap land, a strategy we call "minimize cost" that is dear to the hearts of deal makers. In fact, although there has been no systematic study of the relative cost of protected vs. unprotected land, conservation historically in much of the world has probably used something close to the minimize-cost strategy. Protected land has tended to have low potential for agricultural and other use and therefore low cost (4, 5). The minimize cost approach may work reasonably well in cases with large variation in cost across space and relatively homogenous or randomly distributed species. It will tend not to protect biodiversity on lands that people find valuable (e.g., coastlines and fertile soils). Second, the conservation biologist typically wants to protect areas with the greatest conservation value, a strategy we call "maximize benefit" that is illustrated by the "conservation hotspots" approach (6). We explore these two strategies plus a return-on-investment (ROI) strategy that jointly considers both benefits and costs (7).

There is a mismatch between the scale of on-the-ground decisions and the scale at which biological data are available. This mismatch presents severe impediments to defining effective conservation, especially in developing countries. Land-use and land-management decisions are made at relatively small spatial scales, and it is the sum total of such local fine-scale decisions across broad landscapes that determines conservation outcomes. Yet there is typically very little information on biodiversity at the species level at relatively small spatial scales and little likelihood of getting such information in a reasonable time (8). Indeed, it has been argued that such effort would be misplaced (9). Our study system, the grasslands of Argentina, exemplifies this problem: The area is vast—110 million ha remain unconverted—but conservation decisions need to be made at a scale of a few hundred to a few thousand hectares, and there are virtually no data on species at this level of spatial resolution.

All conservation strategies that try to measure biological benefits must deal with the above impediments. So, in spite of ecological ignorance, the maximize-benefits and ROI strategies require us to define an explicit conservation objective—the conservation benefit to be maximized given available resources—and a clear way of measuring success in achieving that goal (7). The benefit used in most previous applications is the number of species, often the number of endemic species (7, 10). Endemic species—although a taxonomically narrow objective—have some advantages as a conservation goal (6), but the required data are typically not available at the appropriate spatial grain, as is the case here. Instead, we use plant community types—henceforth ecosystems—as our unit for defining conservation benefit; in Argentine grasslands there are 77 such unique ecosystems (11).

Ecosystem type, in fact, is arguably a better category than species for biodiversity protection, for several reasons. Most biodiversity exists in taxa for which little or no information is ever available, and particular (well-known) taxa are not typically good surrogates for other components of biodiversity (12). Conserving the spectrum of ecosystems is likely to provide the conditions needed to conserve the spectrum of species. Each habitat or ecosystem type has unique unknown species and unique properties other than species such as genetic diversity, uniquely evolved local interactions among species, or even special ecosystem processes (4, 13–18).

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Our definition of benefit also places higher priority, all else being equal, on rare ecosystems and on ecosystems facing high risk of conversion. Protecting areas within a given ecosystem type is likely subject to diminishing returns (10), and protecting habitat when there is little remaining is likely to be of greater conservation benefit per unit area than is protecting habitat when large amounts remain. Ecosystems facing high risk of conversion are likely to become rare if not protected. Also, if an area is not at risk for conversion, so that it would remain intact even without protection, there is no benefit to conserving it (19–21). We define conservation benefit as the conservation value gained by taking action in the planning period compared with taking no action.

Besides clearly defined benefits, the ROI strategy also requires information on costs and threats to maximize conservation benefit per dollar spent. Prior work on setting conservation strategies has demonstrated large conservation benefits from incorporating information on costs and threat along with information on benefits (7, 10, 22–28).

Previous ROI applications have focused mainly on a very coarse spatial grain and a large spatial scale, for example entire ecoregions, with low need for local spatial information (e.g., ref. 25), or on a relatively fine spatial grain, a small spatial scale, and a high need for spatial information (e.g., ref. 26). In contrast, we evaluate some 53,000 parcels covering the 110 million ha. We illustrate an approach that tries to solve these problems to produce insightful analysis at appropriate grain and scale.

Habitat, Risk, and Cost

Habitat. The grassland ecosystem data came from a recently published study of South American ecosystems (11). Argentine grasslands currently cover 110 million ha of the original 160 million ha. The grasslands have been classified into 77 ecosystem types (hereafter ecosystems). They are distributed across four ecoregions, of which two (Patagonian Steppe and Low Monte) contain 77% of total grassland cover (Fig. 1*A*). Ecosystems vary widely in total area—from 20 ha to >16 million ha. The vast majority are currently unprotected, with 64 ecosystems having <1% of their area protected. Some 515,132 ha are protected in International Union for Conservation of Nature categories I–IV, 96% of which are within the Patagonian Steppe (Table S1).

We subdivided the grassland into contiguous parcels that were homogeneous with respect to ecoregion, ecosystem, cost, and protected status. We set a maximum parcel size of 10,000 ha to ensure the largest unit did not exceed practical limits. The resulting map has 53,884 parcels.

Risk, Rarity, and Cost. The risk to a parcel is the probability it will be converted from grassland over the planning period. No direct measure of risk of conversion per unit time is available; we have measures of relative risk only. For purposes of this exercise, we used the human footprint dataset, an index of risk level based on population density, land transformation, access (road, rail, river), and electric power infrastructure (29). We interpreted this index as a probability of conversion in the planning horizon. To get values between 0 and 1, we normalized the human footprint values by dividing each parcel value by 100 (Fig. 1*B*). The average risk level is low (Fig. 1*B*), reflecting low human population density and development in most of the grasslands. The human footprint is less well adapted to measuring risk in grasslands (29); however, the results are essentially the same if we use a risk index developed for South America (11).

It is also useful to define ecosystem risk, which is the average risk across all parcels in the ecosystem weighted by parcel area. The average risk can be thought of as the fraction of the ecosystem that would be converted by the end of the planning period if we took no conservation action—higher risk leads to a lower fraction surviving.

Ecosystems vary greatly in size; rare ecosystems have little area. The 23 rarest ecosystems each contain <50,000 ha. Ecosystem risk and rarity are weakly positively correlated overall ($r = 0.197$, $P = 0.083$). This result is mainly because ecosystems in the Humid Pampas and, to a lesser extent, the Espinal, are small and they are at higher risk (Fig. 2). However, within an ecoregion, risk and rarity are not correlated.

We used estimated purchase price of land as our measure of the cost of protection. Conservationists are likely to use various conservation methods, for example easements, and real costs may include, for example, future management; but purchase price is a useful proxy. To estimate the per-hectare cost of land in each parcel, we used land-cost maps created by an Argentine real estate company (30) that showed maximum and minimum land prices for 79 zones in Argentina. Within each zone we assume that land price increased linearly with population density, which was estimated for each parcel from the Landscan 2004 database (31).

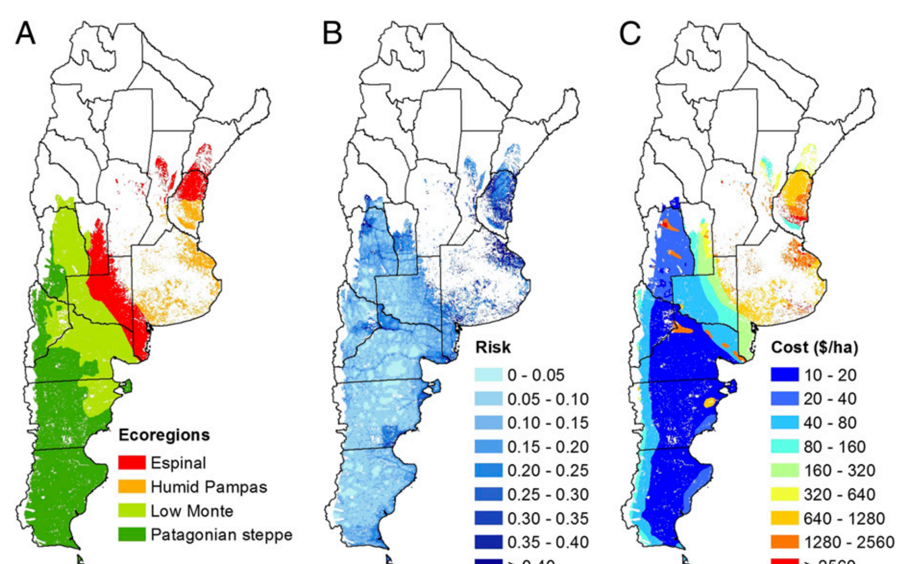

Fig. 1. (*A*) Ecoregions containing Argentinean grassland; (*B*) parcel risk values; (*C*) land price per hectare.

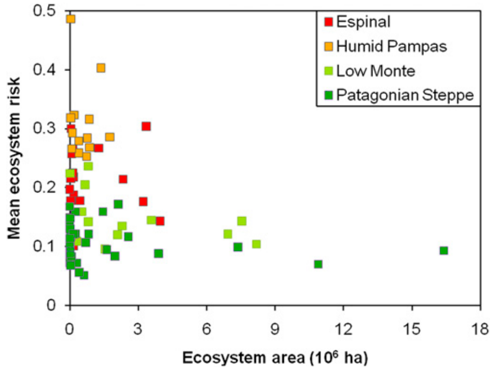

Fig. 2. Mean ecosystem risk as a function of ecosystem area for the four grassland ecoregions.

Estimated land cost ranged from \$10 to nearly \$40,000/ha and is higher in the Humid Pampas and Espinal than in the other two ecoregions (Fig. 1*C*). The average cost per hectare in an ecosystem is highly correlated with average ecosystem risk (Fig. 3).

Defining Conservation Benefit

Diminishing Returns. We use the basic functional relationship in a standard species-area curve to define conservation benefit as a function of area of an ecosystem that remains intact. The standard curve exhibits diminishing returns [$S = ca^z$, the number of species (S) present increases but for $z < 1$ does so at a decreasing rate as area (a) is accumulated]. In our case, the conservation benefit increases with hectares but does so with diminishing returns. Other forms could be used; this one is familiar, and the specific form is relatively unimportant (32).

To take risk into account, we define the benefit of protecting a given parcel as the conservation benefit that would be lost if no conservation action were taken. Thus, if a parcel is not at risk, no benefit is obtained by protecting it.

We considered two aspects of risk. First, for parcel i, risk is the probability that it will be converted, and will lose all conservation value, during the planning period. Thus the actual benefit of protecting parcel i is its intrinsic value multiplied by the probability the parcel will be converted if we do not protect it, namely p_i. The parameter p_i might also be thought of as the fraction of parcel i that will be converted if we do not protect the parcel.

Second, we calculate the degree of risk faced by the ecosystem to which parcel i belongs. We define ecosystem risk at time t as the fraction of the ecosystem that would be converted by the end of the planning period if we took no additional conservation action. If a_f is the area of parcel f in ecosystem type n, then p_f is the probability that parcel f will be converted if it is not protected. We then define $u_f = a_f(1 - p_f)$ to be the expected area of parcel f unconverted if it is not protected. This is an approximation if individual parcels are either wholly retained or wholly converted. Note that if parcel f is protected, then $p_f = 0$ so that $u_f = a_f$; i.e., its full area will remain unconverted at the end of the planning period. Finally, at any time t, $\sum_f u_{f,t}$ is the fraction of ecosystem type n that will remain at the end of the planning period.

We can now see how diminishing returns and risk affect the marginal conservation benefit derived by protecting a particular parcel. For notational clarity, we use lowercase b to indicate the conservation benefit derived by considering local, nonspatial properties of parcels. The marginal conservation benefit obtained by preserving parcel i in ecosystem type n at time t is defined as

$$\Delta b_{t,i} = \left[\left(\sum_{f\in n} u_{f,t} + a_i\right)^z - \left(\sum_{f\in n} u_{f,t}\right)^z\right] p_i, \qquad [1]$$

where n is the ecosystem type, f indexes parcels of ecosystem n, excluding parcel i, $z < 1$ is an adjustable parameter that controls how quickly b saturates as area increases, u_f is the expected area of parcel f that survives, and t is the time step in the calculation, which is done iteratively (see below).

We set $z = 0.3$. The set of parcels selected is quite insensitive to variation in z in a wider range (0.1–0.5) than is commonly seen in empirical species-area curves (*SI Text*). Eq. **1** is most easily understood by reference to Fig. 4. First, at time t, before we make a decision about parcel i, $\sum_f u_{f,t}$ is the area of ecosystem n that is expected to survive unconverted at the end of the planning period; this is therefore the value on the x axis in Fig. 4 where we evaluate the expected marginal benefit of saving parcel i. The larger and less threatened the ecosystem is, the further to the right will $\sum_f u_{f,t}$ lie, and the smaller will be the marginal benefit from protecting parcel i (Fig. 4). Second, the term inside the brackets in Eq. **1** is the expected conservation benefit that would derive from protecting parcel i, with area a_i; when parcel risk is ignored, this value is denoted by δb_i in Fig. 4. Third, to get the nonspatial marginal benefit of protecting parcel i, we multiply δb_i by risk of conversion, p_i.

Eq. **1** favors rare ecosystems because, all else being equal, they have higher marginal values by being closer to the origin of Fig. 4. Similarly, higher-risk ecosystems are more favored because they have lower values of $\sum_f u_{f,t}$ and are closer to the origin in Fig. 4. In addition, particular parcels that are at higher risk (large p_i) have a larger value. Parcels from less rare and less threatened ecosystems are given lower value. Finally, Eq. **1** tends to generate representativeness because, as protected area of any one ecosystem accumulates, it becomes more beneficial,

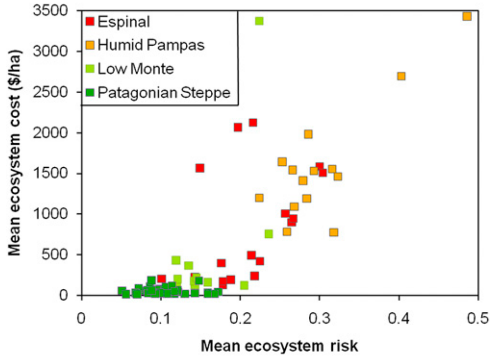

Fig. 3. Mean ecosystem cost increases with mean ecosystem risk.

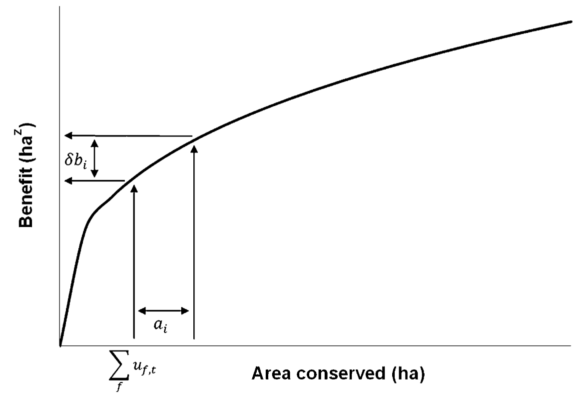

Fig. 4. Marginal conservation benefit, δb_i, of protecting a parcel of area a_i hectares at time t, when the effects of risk and spatial properties are ignored.

all else being equal, to protect a different ecosystem that is less protected.

Spatial Values. Conservation planning typically seeks to balance a general spreading of protection across the landscape vs. clumping or connecting protected areas. We clumped protected parcels by assigning higher conservation benefit to parcels that had more protected land in their neighborhood. For each target parcel i, a neighborhood was defined by a 10 km-wide buffer from all edges of parcel i, a distance over which many species can disperse. All parcels that overlapped even partially with this buffer were considered to be in the neighborhood of parcel i. The minimum distance (parcel edge to parcel edge) from parcel i to each parcel in its neighborhood was calculated, and closer parcels were given greater importance by letting their contribution decline exponentially with their distance from the target parcel.

We implement the neighborhood contribution at time t by multiplying the marginal benefit of parcel i (Eq. **1**) by a weight, $w_{i,t}$, that lies between $w_{\min}$ and 1. If no neighboring parcels are protected, $w_{i,t} = w_{\min}$; if all of the neighborhood is protected, $w_{i,t} = 1$. If, for example, $w_{\min} = 0.5$, the parcel receives only half of its nonspatial value when nothing in its neighborhood is protected. The neighborhood weight takes the form

$$w_{i,t} = \frac{\sum_{k\in K} e^{-qd_{i,k}} a_k \left(w_{\min} + w_{k,t}\right)}{\sum_{k\in K} a_k e^{-qd_{i,k}}}, \qquad [2]$$

where k are grassland polygons in the neighborhood of i, of which there are K, a_k is area of polygon k, $d_{i,k}$ is minimum edge-to-edge distance from polygon i to polygon k, q is an adjustable parameter that controls the rate of exponential decline with distance [variation in q between 0.03 and 0.5 has little effect on the polygons selected (*SI Text*) and is henceforth fixed at $q = 0.1$], $w_{\min}$ is the minimum weight contributed by any neighborhood polygon at any time [variation in $w_{\min}$ between 0.1 and 0.9 has little effect on the polygons selected (*SI Text*) and is henceforth fixed at 0.5], and $w_{k,t}$ is additional weight contributed by polygon k: $w_{k,t} = 0$ if the polygon is unprotected and $w_{k,t} = 1 - w_{\min}$ if it is protected.

We now combine the local and neighborhood properties (combination indicated by uppercase B) of parcel i by defining the total marginal conservation benefit derived from protecting it at time t as

$$\Delta B_{i,t} = w_{i,t}\Delta b_{i,t}. \qquad [3]$$

Saving 10%: The Ranking Process

At each time step in the calculation, all parcels are ranked, the highest-ranked parcel is chosen, the remaining parcels are reranked, the next parcel is chosen, and so on until 15.5 million ha (10% of the original grassland area) are accumulated. We rerank at each time step because protecting an additional parcel will change the neighborhood of some unprotected parcels and hence the benefit of conserving any parcel according to Eq. **1**. This procedure is a "myopic greedy algorithm" and such algorithms have been shown to find solutions close to the globally optimal solution in conservation problems involving complex decision spaces (e.g., refs. 19 and 33).

We calculated three different conservation strategies:

(*i*) *Minimize cost*: Choose the cheapest 15.5 million ha; that is, parcels were ranked by their area/cost ratio. Thus, we maximize the area conserved for the fixed budget.

(*ii*) *Maximize benefit*: Choose the areas with the greatest conservation benefit regardless of cost. In this case, we ranked parcels according to their marginal conservation benefit *per hectare*; that is, for parcel i, rank is determined by $\Delta B_{i,t}/a_i$. This procedure avoids us preferring larger over smaller parcels that are otherwise of equal benefit.

(*iii*) *Maximize benefit per dollar*—the ROI strategy: Here we rank parcels by their marginal conservation benefit *per dollar*; that is, rank is determined by $\Delta B_{i,t}/a_i c_i$, where c_i is the cost per hectare of parcel i.

We compute the total conservation benefit achieved in an ecosystem at any time by summing the benefits provided by all parcels in the ecosystem that are protected at that time. The benefit from each parcel is defined by Eq. **3**. The total conservation benefit achieved by any strategy at that time is obtained by summing benefits across ecosystems.

Results

Using the 10% Goal. The original grasslands covered 160 million ha, of which ~0.5 million are protected, so 10% of the original area would require protecting an additional 15.5 million ha. We

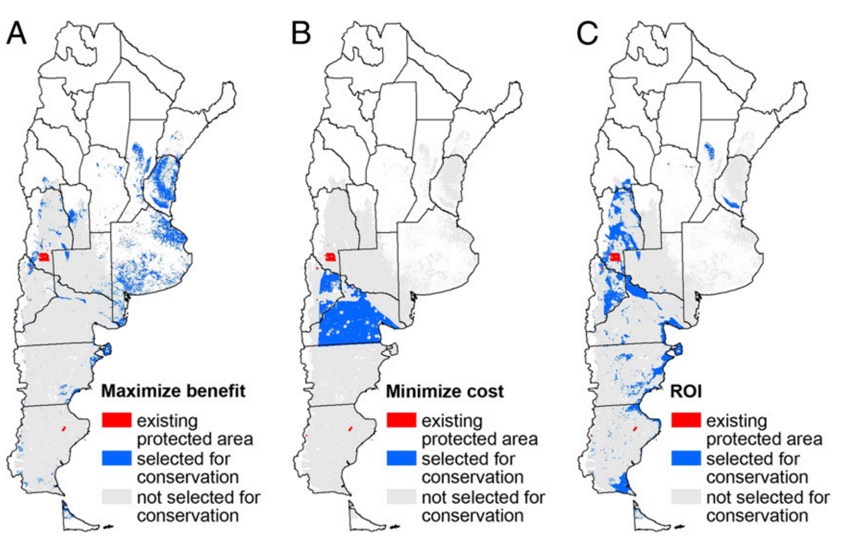

Fig. 5. Distribution of new protected areas resulting from applying each of three conservation strategies to achieve the 10% goal (15.5 million new hectares conserved). (*A*) Maximize benefit; (*B*) minimize cost; (*C*) ROI. Existing protected areas (0.5 million ha) are shown in red.

Table 1. **Allocation of protected areas among the four ecoregions by each of three conservation strategies**

	Strategy		Ecoregion	
	Espinal (%)	Humid Pampas (%)	Low Monte (%)	Patagonian Steppe (%)
Maximize benefit	3,981,079 (25.7)	7,499,488 (48.4)	2,016,842 (13)	1,992,841 (12.9)
Minimize cost	0 (0)	0 (0)	9,155,106 (59.1)	6,337,956 (40.9)
ROI	455,659 (2.9)	328,777 (2.1)	8,503,340 (54.9)	6,200,518 (40)

compare the results from the three strategies described above, where we select parcels totaling 15.5 million ha under each.

Minimize cost set. This strategy buys the cheapest land, ignoring conservation benefits and risk. It aggregates most protected area in a single vast clump in the cheapest province, Rio Negro, which lies where Low Monte and Patagonian Steppe meet (Fig. 5*B*).

Maximize benefit set. The 15.5 million ha selected by Eq. **3** represent a set that is based on ecological properties and risk—their conservation benefit—regardless of cost per unit area. The selection is biased toward the Humid Pampas and Espinal, because these regions have rarer ecosystems that are also subject to greater risk of conversion (Table 1, Fig. 5*A*).

Maximize benefit per dollar (ROI) set. Protected areas are now more broadly distributed than in the cheapest scenario, but Humid Pampas and Espinal still receive only a small fraction of the total protected area because land there is more expensive.

Table 2 shows that the maximize-benefit set, as expected, best meets standard conservation desiderata: Most ecosystems are represented, most have >5% protection, and the rarest and highest-risk ecosystems all receive some protection. The minimize-cost solution yields a very poor conservation outcome by these criteria: low representation and few rare or risky ecosystems protected (Fig. 5*B*). The ROI solution is closer to the maximize-benefit solution than to the cheapest solution in these three categories (Fig. 5*C*).

The differences in the cost of achieving these relative conservation gains, however, are staggering. The clearest comparison uses conservation benefit. The maximize-benefit solution achieves slightly more than twice (2.3) the ROI benefit, but it costs >50 times as much. The ROI benefit is >6 times higher than the minimize-cost benefit, and it is achieved at slightly more than twice the cost (Table 2).

Using Budget Constraints. In the real world, budgets typically constrain conservation action. The $18.7 billion required to purchase 10% of the grassland area using the maximize-benefit strategy is far beyond any imaginable conservation budget that might be available over a reasonable time such as 10–20 y. Even if all conservation were achieved via conservation easements, the cost would likely be at least $4–10 billion.

We therefore examine the performance of the three strategies under a more realistic range of budgets, namely up to $500 million to be spent on land purchase over 10 y (Fig. 6). If easements and other conservation actions were used instead of land purchase, and easements cost 20% of purchase price (a low estimate), the maximum expenditure would be $10 million/y.

The ROI strategy clearly uses resources most effectively (Fig. 6). For total budgets lying between $1 million and $500 million, ROI yields >3 times to ∼4.5 times as much conservation benefit as does the maximize-benefit strategy. The minimize-cost strategy is initially disastrously bad. With a budget of $60,000 the minimize-cost strategy buys only 0.3% as much conservation benefit as does the ROI strategy. At a budget of $10 million, the ROI strategy buys 17 times as much conservation benefit as does the minimize-cost strategy. But eventually the relative performance of the minimize-cost strategy improves. In fact, the minimize-cost strategy outperforms the maximize-benefit strategy over a large range of budgets. At a budget of $500 million, the ROI strategy does only 2.5 times better than the minimize-cost strategy.

We can get useful insight into how ROI achieves greater conservation benefit for a given budget, through balancing benefits and costs, by considering some components of conservation benefit. We chose two budget levels: $100 million and $500 million for illustration (Fig. 6). At both budget levels, with one exception, the ROI solution includes more, and typically many more, high-risk, rare, and total ecosystems than the two other strategies (Fig. 7). The exception is that the maximize-benefit strategy gives some protection to one additional rare ecosystem (25) than the ROI strategy (24) at the lower budget.

The reasons for the superiority of ROI are reflected in the amount and distribution of land protected (Figs. 8 and 9). The maximize-benefit strategy seeks out the riskiest areas at enormous cost per hectare with the result that only 0.4% of the grassland habitat is protected for $500 million. At the other extreme, the minimize-cost strategy buys an enormous area (almost 40% of the grassland for $500 million) of the cheapest and low-risk habitat clumped into a few ecosystems. ROI seeks out areas with intermediate risk but far lower (∼1/70–1/100) cost than the maximize-benefit strategy, protecting almost 20% of the grassland habitat for $500 million.

As in the "protect 10%" case, only the maximize-benefit strategy commits significant resources to the expensive Humid Pampas and Espinal. With a $500 million budget the minimize-cost strategy purchases no land in these ecoregions and the ROI strategy allocates only 4.3% of purchased land there. The maximize-benefit solution distributes almost 88% of the (small amount) of land purchased to these two ecoregions. One could, of course, constrain the ROI strategy to have a larger minimal representation in these highest-risk ecoregions, which would reduce ROI's relative advantage.

Robustness. The results are robust to changes in parameter values (z, w, and q). Results are also little changed when we give more

Table 2. **Number of ecosystems given protection by three different strategies**

Scenario	Represented	>5% protected	Rare	Highest risk	Conservation benefit	Cost ($ billions)
Maximize benefit	72	60	23	23	109.8	18.66
Minimize cost	20	12	3	2	7.8	0.15
ROI	70	54	22	17	47.1	0.35

There are 77 ecosystems; 23 are defined as rare (<50,000 ha) and 23 at highest risk.

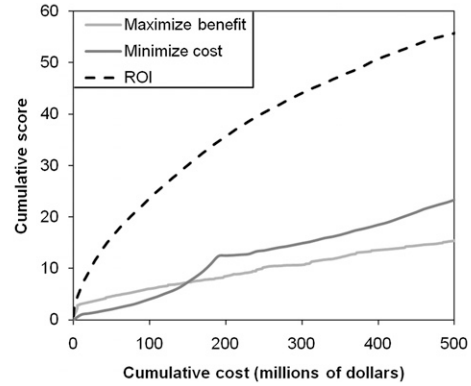

Fig. 6. Conservation benefit accumulated by three conservation strategies as expenditure increases.

weight to target parcels whose neighborhoods have more area with the same ecosystem type as the target parcel, rather than to those with a high fraction of the neighborhood protected. For example, when the goal is to protect 10% of the grassland, the solution is slightly cheaper with same-ecosystem weighting, but there is 89% overlap in the area protected by the two weightings in both the maximize-benefit and the ROI strategies (Table S2 and Fig. S1). Furthermore the relative cost advantage of the ROI strategy over the maximize-benefit strategy is almost exactly the same (Table S3): The maximize-benefit strategy is 53.9 times more expensive than the ROI strategy with protected-parcel weighting and 54.8 times more expensive with same-ecosystem weighting (*SI Text*).

Discussion

Conservation priorities have typically been established with the aim of protecting places yielding greatest conservation benefit, that is, following a maximize-benefits approach. Our analyses show that, except for very small budgets, this is an extremely poor approach. Even something as simple as conserving the cheapest possible land outperforms a maximize-benefits approach for many budget levels. This result is important because it is usually relatively easy to identify cheapest options, whereas it is often quite hard to collect data to support a maximize-benefits approach. Minimize cost is better than maximize benefit mainly because it protects so much more land, albeit land at low risk and lower conservation value.

If biological data are lacking, the message is clear: Find the cheapest options. But if biological data are available, then a formal application of an ROI approach can yield enormous gains in conservation efficiency, which is of great importance in a world with limited resources devoted to conservation (7, 10). Across a wide range of budgets, the ROI approach yielded conservation outcomes per dollar that range from two to eight times better than either minimize-cost or maximize-benefits algorithms (Fig. 6). At present, many conservation organizations use a maximize-benefits approach, which means resources are being inefficiently applied.

Conservation Goals and Strategies. Many governments, multinational agencies, international agreements, and nongovernmental organizations define conservation goals in terms of the fraction of a country or a major habitat type to be protected (3). Such a goal has the advantage of being inspirational and, at least at first sight, simple. But it has serious disadvantages, especially as an operational target (34). Most obviously, it leaves unanswered the central question: Which areas should make up the fraction to be protected?

We illustrate the hidden ambiguity in the goal, and its avoidance of key decisions, using two very different conservation strategies that correspond with historical practices, to estimate the cost of protecting 10% of Argentine grasslands. The hidden decisions have enormous financial implications. The cost of protecting the 10% of grassland with the highest conservation values (maximize benefit) costs >100 times as much as protecting the cheapest land (minimize cost). The first strategy requires impossibly large financial resources whether the land has to be purchased (almost \$19 billion) or is protected more cheaply, for example, by conservation easements (\$4–10 billion).

Cost issues aside, threshold-area goals are arbitrary and not based on an analysis of ecological and conservation needs. Most important, area-defined goals distract us from the true target, namely biodiversity (or some related ecological entity), and from the real problem, which is to use limited resources to protect biodiversity in the most cost-effective way.

Resources available for biodiversity conservation are limited and maximizing conservation benefit for a limited budget is the real problem to be solved. As has been shown elsewhere (e.g., refs. 7, 25, and 28), an ROI strategy is the best solution to this problem. Given a 10% goal, an ROI strategy achieves six times as much conservation benefit, for only twice the cost, as buying the cheapest land; it achieves about half of the conservation benefit for <1/50th of the cost of buying land with the highest conservation value regardless of cost.

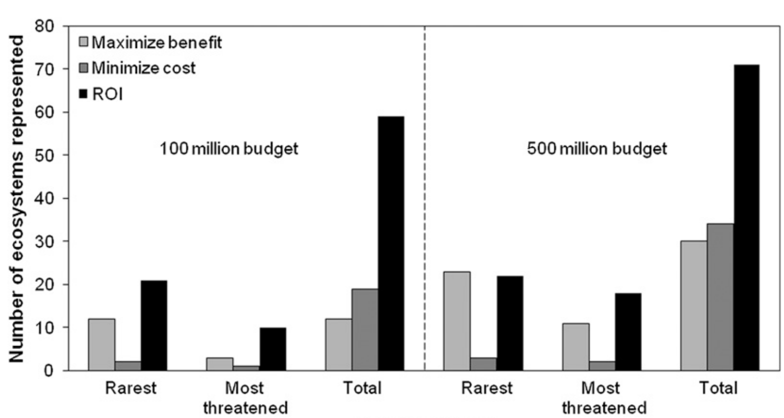

Fig. 7. Number of ecosystems in different categories gaining some protection under three different conservation strategies with an expenditure of \$100 million and \$500 million on land purchase.

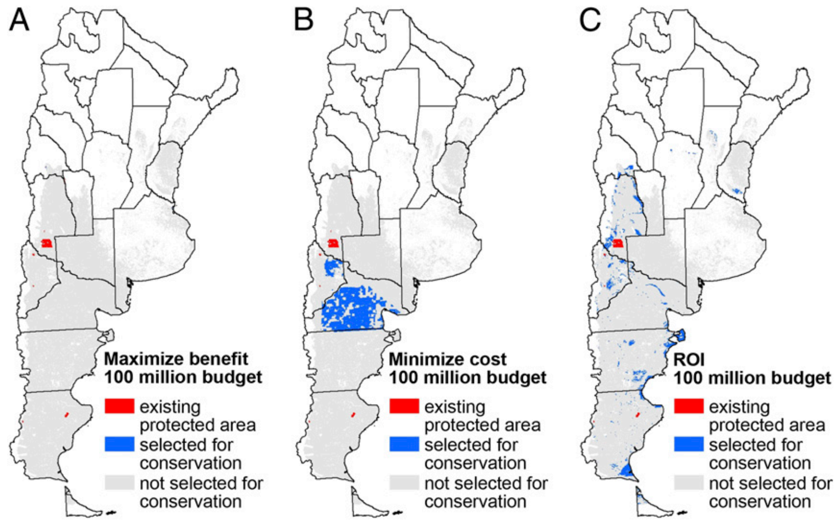

Fig. 8. Distribution of new protected areas resulting from applying each of three conservation strategies with a budget of $100 million. (*A*) Maximize benefit; (*B*) minimize cost; (*C*) ROI.

Value Judgments. Whereas science can inform conservation and make it more effective, conservation is ultimately an exercise in developing and implementing value judgments, that is, explicit preferences about conservation value. The ROI strategy, with its need to define the conservation benefit, reinforces this fact. Failure to define conservation benefit explicitly, which is standard in conservation, evades the central issue of values.

In Argentine grasslands we used hectares of each unique ecosystem as the basic unit, rather than, for example, populations of endemic species, because there is little information on the distribution of endemic or other species. But a good case can be made, especially in developing countries, that hectares of unique ecosystems, or of unique ecological communities, should in any case be the basic conservation unit. Almost nowhere is there sufficiently accurate or precise information, at the appropriate spatial scale, on the distribution of species of conservation interest. Furthermore, we typically have even approximate distributional information on only a few select taxa (e.g., plants, birds, and mammals), which also typically are poor surrogates for other taxa (12). So, if protection of biodiversity is our goal, we will more likely meet it by concentrating on ecological ecosystems (communities) rather than on species. Ecosystem type also has weaknesses, of course. It typically relies heavily on plant composition or type and is not likely to be a universal surrogate for other taxa.

Choosing ecosystems as the planning unit still allows us to exercise our judgment that some ecosystems may be more valuable than others. We may feel, all else being equal, that an ecological community with more vertebrate, or plant, or bird, or endemic species is more valuable. Communities, like wetlands, that are linked strongly to others may be considered intrinsically more valuable than, say, forests, or we might value more highly those believed to provide more ecosystem services. Such judgments can be incorporated simply and explicitly in the approach followed here. For example, we can assign different values to the coefficient, c, in the standard diminishing-return curve defined by $B = cA^z$ (7). In our case we had no basis for implementing such judgments and so we set the constant $c = 1$ for all ecosystem types.

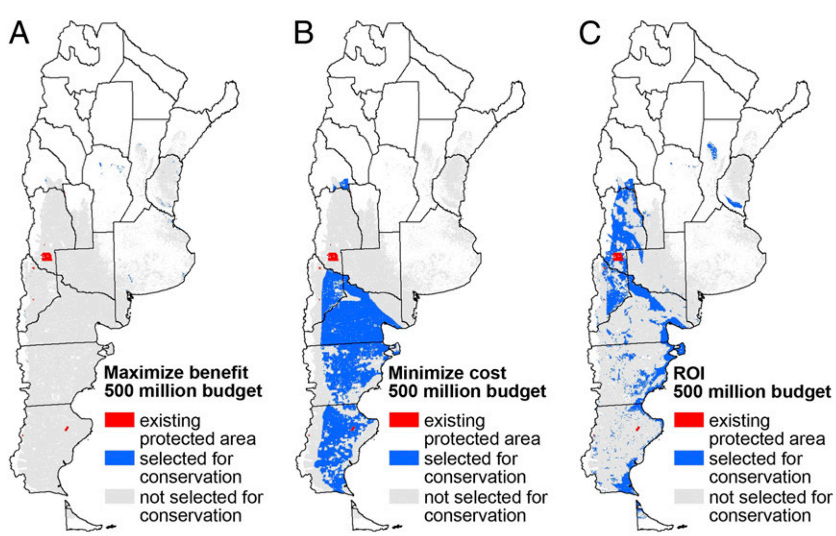

Fig. 9. Distribution of new protected areas resulting from applying each of three conservation strategies with a budget of $500 million. (*A*) Maximize benefit; (*B*) minimize cost; (*C*) ROI.

Our approach does place a premium on rare and at-risk ecosystems, on conserving a wide range of ecosystems, and on minimizing fragmentation of protected areas.

Finally, better data can lead to better conservation, and in the case of Argentine grasslands, perhaps the most needed data concern threats and their effect on grassland communities. Most of the grasslands are grazed, and have been grazed for over a century, but data on the original state of the grasslands, and the effects of grazing, are scarce or nonexistent. If grazing is the most widespread threat, the major conservation action may be the removal or reduction of grazing and restoration through stewardship arrangements. Unfortunately, there is little evidence about the vegetational effects of removing grazers in Argentine grasslands. Thus the expected increase in conservation benefit from such conservation actions is hard to predict.

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