**McBride et al 2007**

Limited resources mean that conservation investments must be chosen carefully and their performance evaluated critically. These approaches highlight areas in need of conservation action, but provide limited information as to how resources might best be allocated among regions over time. An alternative approach is to use information on biodiversity values, threats, and costs to determine a conservation investment schedule that maximizes our expected biodiversity returns for a fixed budget (Costello & Polasky 2004; Wilson et al. 2006).

This approach, however, does not address sources of uncertainty that have the potential to compromise conservation outcomes. Such sources include political instability and corruption; the impact of natural catastrophes; lack of budget continuity; weak governance; project failure; absence of stakeholder willingness to be involved; failure to effectively empower stakeholders or mainstream conservation priorities; and implementer “burnout” (Barrett et al. 2001; Byron et al. 2001; Smith et al. 2003; Laurance 2004;Winter et al. 2005; Knight et al. 2006).

Broadly, the impact of these socioeconomic uncertainties on conservation investments can be distilled into two types: (1) the ability to invest and the level of investment in a region are not constant over time (i.e., transaction uncertainty) and (2) investment in a region does not guarantee the long-term persistence of biodiversity (i.e., performance uncertainty). Investment uncertainty may result not only from country-level issues, such as weak governance, domestic conflicts, and corruption, but also from shifting priorities among donors and budget shortfalls.

We used stochastic dynamic programming (SDP) to determine an optimal resource allocation schedule, which is a statebased backwards-iteration algorithm. For each possible system state, the algorithm determines the optimal solution based on the current state and the expected return, given the likely transition probabilities (Bellman & Kalaba 1965; Clark & Mangel 2000).

There is however, a dearth of publicly available, quantitative, and empirical data that describes the relative successes of past conservation investments (Ervin 2003; Stem et al. 2005).

We explored two alternative expressions of transaction uncertainty to assess the effects of the possibility that resource allocation will either cease or increase. First, we introduced an annual probability that the ability to acquire more parcels might be lost and that all currently available parcels might become unavailable. Likewise, to examine the effects of potential increases in funding or capacity to mitigate parcel loss, we incorporated a probability that each year enough funding might become available to acquire all remaining available parcels. Second, we considered the effects of incorporating a less extreme expression of transaction uncertainty by relaxing the assumption that once funding ceased, it could never be reinstated. Instead we allowed for the possibility of variable lengths of funding unavailability, following which conservation investment in a region could recommence.

To assess the effects of performance uncertainty, we introduced a region-specific probability that each year any currently reserved parcel within a region had a fixed probability of failing.

Relative differences of between 1 and 10 were used for the predicted rates of habitat loss, which were obtained by predicting changes in human-footprint values on the basis of predicted rates of human population growth. The lowest predicted rate of population growth was 0.002 in the forest ecoregions of northeastern Spain and southern France, and the highest predicted rate of population growthwas 0.02 in the eastern Mediterranean conifer-sclerophyllous-broadleaf forests (E. Underwood et al., unpublished data).

Nevertheless, when differences in transaction uncertainty were less extreme, the relative vulnerability or the biodiversity value of each region remained more influential on the optimal approach. The minimize-loss heuristic again underperformed relative to the maximize-gain heuristic, delivering final outcomes that differed from the optimal solution by up to 20% (Fig. 2c). The impacts of transaction uncertainty on the performance of the heuristics were reduced significantly when the loss of funding was not final, and there was a chance that funding would resume at some time in the future (Fig. 2d).

When performance uncertainty was incorporated, the optimal solution became a complex trade-off between the immediate biodiversity benefits of acting in a region and the perceived longevity of the investment. In general, variation in the probability of reserve failure was more influential in determining the optimal solution than either the regional rate of habitat loss or biodiversity value, and the optimal SDP solution almost always favored the region with the greatest performance certainty, even if the alternative region was highly threatened or had higher biodiversity value (Fig. 3)

In societies that too often value short-term profits over long-term gains, it is important to look beyond the quick fix and focus conservation efforts on achieving lasting benefits for biodiversity. Current priority-setting approaches typically neglect the range of economic, political, and social factors that affect the likelihood of investment success.

Losing the ability to acquire new land in a region (i.e., cessation of funding) had considerable impact on the optimal resource allocation strategy. The optimal strategy changed if there was even a small chance that funding would cease. Under such uncertainty, strategies that maximize short-term gains are the most robust in the long term. In comparison there was little variation in the performance of the heuristics when there was a possibility that future funding levels would increase. These results, although intuitive, highlight the importance of considering the sensitivity of funding strategies over time to system dynamics and uncertainties. Regardless, our results highlight the pragmatic importance of a precautionary and opportunistic approach (i.e., Noss et al. 2002; Knight & Cowling 2007), which would likely be further emphasized under both sources of uncertainty.

When we included a measure of performance uncertainty— a possibility that investments fail to protect biodiversity assets—our resource-allocation strategies changed drastically. Resources were directed away from regions with high levels of performance uncertainty, particularly during the early years of an investment term. Because it affected the length of time reserves were able to contribute to biodiversity conservation, the level of performance uncertainty was the regional characteristic that had the greatest influence in determining where and when resources were allocated.

The underperformance of the myopic SDP algorithm further highlighted the importance of making decisions based on long-term expected outcomes, as opposed to decisions that optimize only in the short term.

Our results demonstrated the importance of accounting for the likely success and longevity of conservation investments when prioritizing the allocation of conservation resources at the interregional scale. Our results confirmed commonly held beliefs that, in systems such as the Mediterranean Basin, the exclusion of social and political factors may preclude the efficient allocation of conservation funds. These factors (and their potential influences on conservation outcomes) remain largely neglected within the field of systematic conservation assessment.

Lack of research into the effects of sociopolitical uncertainties on conservation priorities is likely to be driven by the paucity of empirical data on the relationships between the characteristics of conservation investments and their performance outcomes.

Our results show that the ability to invest and the performance of investments should significantly alter investment strategies and long-term conservation outcomes. It is clear from these results that the impact of investment uncertainty must be accounted for explicitly when prioritizing the allocation of conservation resources. By shifting the focus to achieving longer-term outcomes and by accounting for investment uncertainty when prioritizing the allocation of conservation funds, the ability to maximize conservation returns from every dollar invested improve substantially.

**Wilson et al 2006**

The relative cost of conservation in different regions is ignored in the identification of priority regions despite evidence that its inclusion improves the cost-effectiveness of conservation prioritization.

We have formulated the conservation resource allocation problem in a clear and transparent manner that involves defining an objective, identifying management actions, acknowledging constraints and incorporating uncertainty.

**Santana et al 2013**

The effectiveness of EU conservation investments in N2000 is poorly understood, because studies are scarce, and they tend to be geographically biased, short-term, and rarely consider interactions between various protection and funding schemes.

Our study showed mixed effects of long-term conservation investment in N2000 farmland. We found positive effects on flagship species, and on species associated with fallows, which were the main targets of conservation investment.,

Conservation investment appeared positive on populations of highly threatened flagship species (*O. tarda*, *T. tetrax*, and *F. naumanni*), supporting the view that targeted efforts combining legal regulations and adequate funding schemes may deliver major conservation benefits.

The contrasting effectiveness observed for flagship species and other steppe birds suggests that investment concentrating on charismatic species does not necessarily lead to the conservation of the overall steppe bird assemblage (Caro 2010).

First, we suggest that general biodiversity measures may be in some circumstances misleading indicators of conservation success. Parameters specifically tailored to reflect the outcome of conservation interventions may thus be needed, focusing for instance on the richness and abundance of groups of species of conservation concern that are specialized in specific habitat types.

Finally, long-term evaluations of conservation investment are required, in order to monitor and improve the effectiveness of billions of euros needed annually for managing N2000.

**Teshome 2014**

As population increases and land becomes scarce, land demand by the growing number of land claimants may be met by non-market mechanisms such as state land redistribution, informal land contracts and customary inheritance. The persistence of such mechanisms and absence of an established legal rights land system has resulted in increasing tenure insecurity and continued land fragmentation.

The absence of tenure security is highly linked to poor land use which in turn leads to environmental degradation (Otsuka and Place, 2001; Wannasai and Shrestha, 2008).

Investment in soil and water conservation practices are influenced and constrained by socio-economical and institutional factors (de Graaff, 1993; Shiferaw et al., 2009). Soil conservation investment may be undertaken when sufficient returns are expected for a considerable period of time in comparison with the situation when such investments are not made. This is possible with a secure land tenure system.

As expected, tenure insecurity has a significant negative influence on soil conservation investments. This suggests that tenure insecure households are less likely to invest in soil conservation technologies.

**Lennox & Armsworth 2011**

One drawback of early studies was that the solution was assumed to be static, in the sense that all desired sites could be secured instantaneously. However, the process through which land is protected is inherently dynamic. This temporal aspect of the acquisition process is caused by several factors. For instance, in any time period funding is rarely sufficient to allow the conservation agency to protect all sites of conservation concern (Costello and Polasky, 2004). Moreover, when and where sites will be available for conservation cannot be known in advance (Meir et al., 2004).

The dynamic nature of the acquisition process introduces several sources of uncertainty. Although the conservation agency may be able to determine current site availability, future availability cannot be known with certainty. Further, the conservation agency may be able to assess the current ecological value of sites but forces such as climate change or changing local land use can alter the future value of sites. Therefore, in order to make decisions that are robust to future change, conservation agencies must be aware of these sources of uncertainty and factor them into conservation planning.

In this paper, we examine the role that different kinds of uncertainty play in determining the relative advantages of short and long contracts. Specifically, we examine how uncertainty over future site availability and over future site ecological condition affects the choice of contract duration.

Empirical research has shown that the fate of land protected in temporary contracts is uncertain.

Firstly, the most important factor in setting the choice of conservation contract was the likely availability of sites in the next time period. Secondly, contract choice was relatively insensitive to uncertainty over future ecological conditions. Finally, when the probability that sites would be unavailable was moderate to high, a portfolio through time of short and long contracts was more advantageous than relying solely on either type.

**Murdoch et al 2010**

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).

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.

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.

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.

**Jansen et al 2006**

While forestry has considerable potential to contribute to livelihoods in some of the poorest hillside areas, forest area has been reduced to due land tenure inequities (recent household survey data in [Jansen et al. (2003a)](https://www.sciencedirect.com/science/article/pii/S0308521X05000910?casa_token=RFxxEaAHYqAAAAAA:nc1ZX24-1aTCf7lmPZB4D4CveBkltOdPdlYk3G-P-LtBWWxxgEf3xtldCaK2CT5dq7jnCKbvuQ49" \l "bib7) show that only 25% of all parcels in the hillsides have secure tenure), high population growth rates, and land reform programs in the 1960s and 1970s

Regarding human capital, population density in most communities increased around 50% between 1988 and 2000.

Population density shows a consistently negative and significant coefficient for five of the six income-earning strategies included in the model. For vegetable communities this result appears to be consistent with [Pender et al. (2001)](https://www.sciencedirect.com/science/article/pii/S0308521X05000910?casa_token=RFxxEaAHYqAAAAAA:nc1ZX24-1aTCf7lmPZB4D4CveBkltOdPdlYk3G-P-LtBWWxxgEf3xtldCaK2CT5dq7jnCKbvuQ49" \l "bib11) who reported relatively low population densities in such communities, despite a relatively high population growth. For the other strategies one would be inclined to argue that income-earning activities in densely populated communities are limited to basic grains and livestock, which carry relatively low economic returns. Within this logic high population densities are closely associated with poverty. On the other hand, 75% of the communities that focus on basic grains and livestock production are located in the north of the country (provinces of Atlántida and Colón) where average incomes are relatively high. However, since the logit model already controls for most of these factors, the negative and significant population coefficients indeed seem to suggest that a high population density (and the resulting land fragmentation) tends to lock people into producing basic grains and livestock (mainly for own consumption) and as such impedes the transition to other (possibly more profitable) income-earning strategies.

The results regarding the influence of population density on conservation practices suggest a U-type relationship. That is, up to a certain population density the four conservation practices considered here are less common in communities with higher population densities. However, after a certain point population density has a positive influence on the adoption of conservation practices. A possible explanation for this result may be that only after population density and degradation of natural resources reach certain critical levels, people start becoming aware of the need for conservation practices. To the extent that the latter are labor-intensive, their costs may also be lower in densely populated environments.

**Fryxell et al 2010**

Here we show that weak compensatory response by harvesters or resource managers can itself generate cyclic variation in resources, exacerbating the risk of collapse. Weak harvest regulation contributes to the problem rather than providing an acceptable management solution to resource fluctuation.

Weak harvest regulation should be particularly worrisome in populations with high levels of demographic or environmental stochasticity or with pronounced Allee effects due to predation, disease transmission, or reduced probability of breeding (28–30). Our simulations suggest that the risk of population collapse could be dramatically higher in systems with dynamic effort and quota levels (Fig. 3), simply because of extreme population excursions caused by quasiperiodic dynamics resulting from even mild levels of environmental stochasticity. When resource abundance is regulated by dynamic management responses via quotas, unstable systems are often vulnerable to extinction, even in the absence of Allee effects or stochastic variation in growth rates. These findings suggest that it is unwise to neglect dynamic patterns of change in both harvest effort and quotas in assessing longterm strategies for sustainable resource use.

**Wittemyer et al 2008**

Our results show average annual growth rates were higher in PA buffers than in rural areas of the same country for 245 of the 306 PAs and in 38 of 45 countries (Figs. 1 and 2).

Our comparison of population growth around the borders of PAs with average rural rates for the same country (11) may present a false picture of human settlement if parks are preferentially placed in areas of high ecological productivity. In such a scenario, humansmay settle in the same general region as PAs simply because the land there is better for agriculture or natural resource extraction rather than for reasons related to the PA itself (12). To account for this possibility, we refined our analysis by restricting our comparison of population growth rates in the buffers of Pas to those areas with the same ecological characteristics, defined using the Global Ecoregions Database (13). Results of this comparison show that, similar to our countrywide comparisons, human population growth around PAs is significantly higher than that observed in matched ecoregions (Wilcoxon test: Z = –291.5, n = 69, P = 0.04).

It also is conceivable that the observed high rates of human population growth in PA buffers are caused by the displacement of people living within PAs to their edges (3). In such a scenario, population growth within parks should decline over time as people move outwards toward PA edges. However, contrary to this expectation, population growth rates were positive, not negative, inside 85% of the PAs we surveyed with the remaining 15% showing no change. This finding makes clear that “leakage” from within parks does not explain our result, as population growth was positive not only at PA borders but also within PAs.

Although PAs may be positive for localized rural development in Africa and Latin America, human populations around PAs frequently have significant, negative impacts on biodiversity (22). The scale of human settlement around PAs is a strong predictor of illegal timber and mineral extraction (23), bushmeat hunting (24), fire frequency (25), and, more generally, species extinction (24) within PAs.

We examined such impacts directly by comparing population growth rates in PA buffers with published rates of deforestation in the area surrounding 55 forest PAs included in our study (26). Rates of deforestation were highest around PAs where human population growth was greatest (Fig. 3). This finding links population growth around PAs to habitat loss and suggests settlement around PAs may create a ring of disturbance that isolates PAs from surrounding habitats.

**Utami et al 2020**

For example, as economies and human populations grow across much of South East Asia, protected areas provide important combinations of values that are increasingly rare, including endangered species and spiritual connections to ancient traditions and essential natural resources for human subsistence (Chape et al., 2008; Juffe-Bignoli et al., 2014). However, without effective management of protected areas, biodiversity and its associated values have been shown to decline (Leverington, Costa, Pavese, Lisle, & Hockings, 2010). The effective management of protected areas in South East Asia is therefore essential for conserving, restoring and utilizing protected area values from local to global scales (Juffe-Bignoli et al., 2014; Sodhi, Koh, Brook, & Ng, 2004).

Making management decisions for multi-value protected areas is complex and challenging. Managers of protected areas typically have inadequate resources and information for managing all important values and threats (Bruner, Gullison, & Balmford, 2004; McCarthy et al., 2012; Watson et al., 2016). Values in protected areas can conflict, such as where wildlife threatens agri- culture (Karanth, Gopalaswamy, Prasad, & Dasgupta, 2013), or complement, where management actions can simultaneously improve biodiversity and reduce poverty.

Good management outcomes therefore rely upon understanding which management strategies will generate the greate st improvements across multiple values, using the limited resources available, while taking account of different objectives, trade-offs, synergies and opportunities.

Emerging conservation action prioritization approaches use structured decision making and expert elicitation (Hemming, Burgman, Hanea, McBride, & Wintle, 2018; Martin et al., 2012) to identify the management strategies that are likely to provide the greatest return on investment for improving biodiversity values (Carwardine et al., 2012; Cullen, 2013; Joseph, Maloney, & Possingham, 2009). These approaches assess the cost-effectiveness (calculated by the nonfinancial benefits divided by cost) (Levin & McEwan, 2001) for improving species outcomes through implementing a range of strategies or actions, in contrast to spatial conservation prioritization tools, like Marxan or Zonation, which require spatial data to provide optimal locations for protection and management (Ball, Possingham, & Watts, 2009; Moi lanen et al., 2005).

Investors and protected area directors are required to make decisions about how to use limited funding to improve multiple values. However, in many cases they do not have a consolidated set of knowledge to estimate how much resources are needed to achieve a good out- come, or which management strategies will collectively create the greatest improvements to all values.

**Cullen 2012**

**Meir et al 2004**

Here we explicitly consider the implications for biodiversity conservation of several key assumptions underlying systematic conservation planning methods. We explore both simple and more complex conservation problems in which (1) entire reserve networks cannot be implemented instantaneously, (2) there is uncertainty about when and where opportunities for conservation investment may arise, (3) budget constraints vary, and (4) there is degradation or loss of biodiversity over time in sites that remain unprotected.

Historically, the question of investing now or in the future has been answered haphazardly, but recently, many organizations have initiated planning processes to identify comprehensive or near optimal networks of reserves in the context of explicit biodiversity conservation objectives.

We have shown that comprehensive conservation plans may be worthwhile when the resulting reserve network can be fully implemented immediately after it is designed (e.g. when the lands or waters involved are entirely in government ownership). However, such comprehensive planning may not be necessary, and may even be counter-productive, when implementation is carried out over years. Our results suggest that relatively simple rules for deciding which areas to protect outperform both ad hoc investment strategies and comprehensive conservation plans (Figs 1 and 2). This is especially true when degradation rates and uncertainty are high (Fig. 3).