A General Framework for Prioritizing Land Units for Ecological Protection and Restoration

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ABSTRACT / Past methods of prioritizing sites for protection and restoration have focused on lists of criteria or algorithms. These methods lack a common underlying framework, such that the process is explicit and repeatable, assumptions are highlighted, and commonalities and differences among prioritizations can be readily assessed. Our objective in this paper is to provide such a framework for cases where the goal of setting priorities is to maximize the ecological benefit gained from limited resources. We provide simple and general models that can be used to prioritize sites based on the projected ecological benefit per unit restoration or protection effort and to estimate the total projected benefit of restoring or protecting a set of sites. These models, which are based on an expression of the functional relationship between an end point and effort, hold up under a variety of situations and provide a common language for prioritization. We then discuss procedures for estimating model terms—calculations from regression curves when data are available, and use of judgement indicators when data are relatively limited. Finally, we present two case studies that apply the models and examine selected past prioritizations in the context of our framework.

Ecological restoration and protection are two main applications of ecological knowledge. A primary question in these applications is: given a list of sites deserving of restoration or protection, where is it most important to target resources? In situations where resources such as time or money are not limited, there is no need to prioritize sites; all deserving areas can be restored or protected at the same time. Setting priorities is necessary, however, whenever resources are not sufficient to protect or restore all sites simultaneously (Leibowitz and others 1992). Prioritization can assure that we maximize the benefit gained from limited resources.

Every investment decision to restore or protect an area is implicitly a statement of priorities (Johnson 1995). However, methods that provide an explicit statement of priorities in the planning stages are needed to guide investment decisions. Past methods of prioritizing areas for protection and restoration have focused on lists of criteria or data layers (e.g., Margules and Usher 1981, Hamlett and others 1992, Johnson 1995) or on algorithms for selecting sites (e.g., Kirkpatrick 1983, Margules and others 1988, Pressey and Nicholls 1989, Kiester and others 1996, Pressey and others 1996). A shortcoming of past prioritizations is the lack of a common and rigorous framework for ranking sites, such that the process is explicit and repeatable, necessary assumptions are highlighted, and commonalities and significant differences among prioritizations can be readily assessed. Our objective in this paper is to provide such a framework for cases where the goal of setting priorities is to maximize the ecological benefit gained from resources that are both limited (in short supply) and limiting (their short supply limits the amount of benefit that can be achieved).

In our framework, there are two primary criteria for setting priorities: the projected marginal change in an ecological end point per unit of restoration effort, and the projected marginal change in an end point avoided per unit protection effort. These criteria are analogous to a cost–benefit ratio where benefit is expressed in terms of an ecological end point and cost is expressed in terms of effort. The application of cost–benefit analysis to environmental decision making is controversial (Freeman 1979, Kelman 1981, Lothrup 1986, Gillroy 1992, Boardman and others 1996). A key criticism of the approach is that benefits and costs are usually both expressed in dollars, yet many environmental benefits

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are not adequately valued in economic terms (Schramm 1973, Dixon and Sherman 1991, Randall 1991, Turner 1991). Benefits may be expressed in units other than economic (Macmillan and others 1998), however, and our framework is philosophically akin to such approaches.

We present our framework for prioritization in five main sections: we define key terms; model the projected benefit of restoration and protection; discuss the estimation of model terms; present two applications of the framework; and examine representative examples of past prioritizations in the context of our framework.

Definitions

We define restoration and protection in terms of end points, which are the goods or services that we ultimately want to restore and protect. Ecological end points are most commonly expressions of valued ecological functions such as flood control, pollution removal, and maintenance of biodiversity (Margules and Usher 1981, Suter 1990, Gentile and Slimak 1992, Keddy and Sharp 1994, Brinson 1995, Johnson 1995).

Several definitions of restoration are possible (Wyant and others 1995). We define ecological restoration as a change in an ecological end point from an altered state toward a reference state, resulting from some action. It is important to distinguish restoration of an end point from the replacement of structural characteristics that support the end point. For example, although restoration of wetland functions requires reestablishment of wetland hydrology (Delphey and Dinsmore 1993), focusing on hydrology can lead to erroneous conclusions about the success of restoration; cases exist where hydrology is replaced but wetland functions are not (e.g., Galatowitsch and van der Valk 1996).

Similarly, there are several ways to define protection. We define ecological protection as the avoidance of all or part of a change in an ecological end point that would inevitably take place in the absence of protection. This definition requires consideration of the potential level of an end point if an area is not protected.

General Models of Benefit

We present general models of the ecological benefit of restoration and protection as a function of management effort. These models can contribute to a common language for prioritizing sites and for estimating the total projected benefit of specific management scenarios once sites are prioritized. Because our models are intended to be used before actions are taken, the benefits analyzed are potential rather than actual.

In keeping with our above definitions of restoration and protection, we define the benefit of restoration as the magnitude of change in an end point produced by restoration actions, and the benefit of protection as the magnitude of change in an end point avoided by protection actions. Whereas schemes to prioritize restoration and protection may also consider economic, aesthetic, social, and institutional benefits, in this paper we focus on ecological benefits.

We present our models in two parts. We begin by modeling the ecological benefit of restoration and protection for a single site. Then, we discuss the ranking of multiple sites based on model terms.

Modeling Ecological Benefit Gained from Effort Expended on a Site

Two key variables are necessary in our models. First, let Y denote the level of an ecological end point over an area. The choice of Y and its scale depends on the management objective. For example, Y could be the level of river flooding, the abundance of oak trees, or extinction risk, and these could be considered over a wetland, watershed, or region. Use of our framework requires that the meaning and scale of Y be carefully and explicitly defined. For instance, modeling the benefit of restoration or protection to "all birds" versus "all waterfowl" versus "a threatened species of waterfowl" will likely require different assumptions and relationships between Y and effort. Moreover, modeling the benefit to birds over a watershed versus a region may involve these same differences.

Second, let E denote the level of restoration or protection effort necessary to produce or maintain a given level of the end point Y. By increasing E over a reference level, we apply restoration or protection actions. Effort is measured at the scale of a site, which is the spatial unit to be prioritized (note that the scale of Y may be larger than the site). Effort is often expressed in terms of the most limited and limiting management resource, usually dollars or hours of labor spent on activities such as land purchases, subsidies, enforcement, and tree planting, but effort can also include the opportunity cost to society of not allowing alternative land uses to proceed on a site.

In this paper we consider Y and E as scalars only. Modeling multiple end points Y introduces the problem of expressing benefit in terms of potentially conflicting factors, but our models provide no guidance on how to aggregate or trade off multiple end points. Multiple end points would need to be expressed in a common currency, such as units of energy or dollars (e.g., Ton and others 1998), or otherwise combined according to some rule (e.g., Sankovskii 1992, Tecle 1992). Discussion of multiattribute methods is beyond the scope of this paper.

Our models express the relative ecological benefit, in terms of a change or avoided change in *Y*, produced by restoration or protection effort *E* expended on a site. To model this benefit, we assume the form of the functional relationship between *Y* and *E*. We consider two functional forms of increasing complexity: a linear relationship (equation 1); and a piecewise linear relationship reflecting possible nonlinearity between *Y* and *E* (equation 2). Because the relationship between *Y* and *E* is likely to be nonlinear when considered over their entire range of values, our linear models are intended to represent linear transformations of, or approximations to, nonlinear forms.

We use the same general models for restoration and protection, which are modeled here as different sides of the same coin. However, the meaning of model terms may differ and will be contrasted for these two cases as the models are presented.

Linear benefit model. Let DY_{DE_j} be a change in ecological end point *Y* given a change in effort (DE) on the *j*th site. We use equation 1 to model the ecological benefit derived from this additional effort, assuming *Y* and *E* are linearly related.

$$DY_{DE_j} = Y_{final} - Y_{initial} = DE_j \times \frac{dY}{dE_j} \qquad (1)$$

The term dY/dE_j is the slope of the *Y* versus *E* line and reflects the ability of the end point to respond to a unit change in effort on the *j*th site. Although DE_j is always nonnegative, the benefit derived from DE_j must be interpreted with the sign of dY/dE_j in mind, which may be positive or negative depending on the definition of *Y*. For example, dY/dE_j is positive if *Y* is native species richness and is negative if *Y* is extinction risk; an increase in native species richness and a decrease in extinction risk are both considered possible benefits of restoration.

The model terms are illustrated in Figure 1, which shows hypothetical linear relationships between *Y* and *E*, where $Y_{final} > Y_{initial}$ and dY/dE_j is positive. We assume that *Y* is in temporal equilibrium with *E*; when *E* is held at a particular level, a corresponding *Y* is reached after a period of adjustment and transition. We also assume that all other factors besides *E* and *Y* are constant throughout, since changes in a related variable could also affect the relationships. Although the same model terms are used for restoration and protection, the trajectories and thus the magnitudes of model terms will likely differ for each case.

For restoration, equation 1 describes the change in *Y*, relative to the current level of *Y* (current *Y*), produced by a given increase in restoration effort from E_0 (Figure 1A). E_0 may be zero or some other reference level. Y_{final} is the projected restored level of *Y* (restored *Y*) and $Y_{initial}$ equals current *Y*. The slope of the line (dY/dE_j) reflects the ability of the system to respond to an increase in restoration effort in site *j*. DE_j is the additional effort required to establish and maintain restored *Y*.

For protection, equation 1 describes the change in *Y* avoided by a given increase in protection effort (Figure 1B). Y_{final} and $Y_{initial}$ are the projected levels of *Y* with protection (protected *Y*) and without protection (unprotected *Y*), respectively. The level of unprotected *Y* reflects the degree of threat to *Y*, which is related to the sensitivity and vulnerability of the site to stressors and the intensity of stress (Keddy and Sharp 1994, Johnson

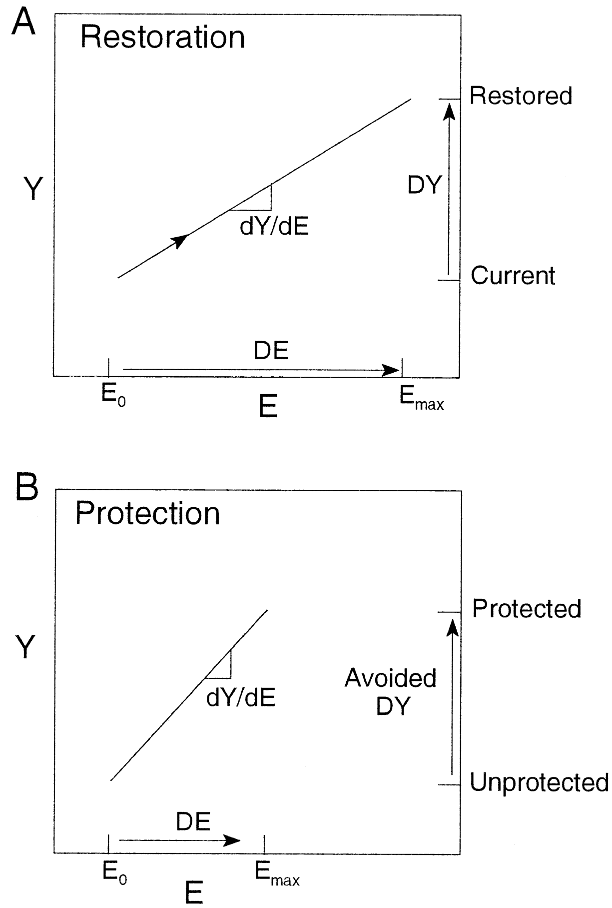

Figure 1. Conceptualization of linear changes in the ecological end point *Y* with changes in effort *E* from E_0 to E_{max}. In **A,** restoration increases *Y* from the current level (current *Y*) to a projected restored level (restored *Y*). The slope *dY/dE* is represented by a discrete approximation. In **B,** protection avoids part or all of the loss of *Y* assumed to occur without protection (unprotected *Y*). If protection is partial, the level of *Y* with protection (protected *Y*) is less than current *Y*.

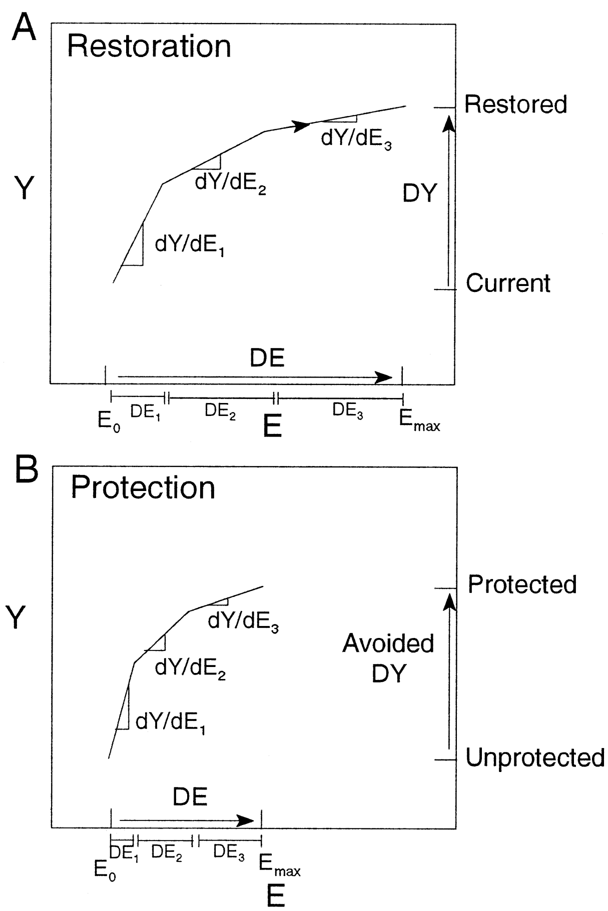

Figure 2. Conceptualization of piecewise linear changes in ecological end point Y with changes in effort E from E_0 to $E_{\max}$. The relationships for restoration (**A**) and protection (**B**) differ from those in Figure 1 in that now a nonlinear relationship between Y and E is approximated by a series of straight-line segments defined over the intervals DE_l.

1995, Rossi and Kuitunen 1996). As with restoration, dY/dE_j reflects the ability of the system to respond to an increase in effort; specifically, the change in Y avoided per unit protection effort. Equivalently, dY/dE_j can be viewed as the marginal change in Y inevitable in the absence of protection. DE_j for protection is the change in effort required to maintain protected Y, assuming that Y would change in the absence of protection.

There are three things to note about our treatment of protection. First, there is no actual change in Y with an expenditure of protection effort; there is only an avoided change. Second, the avoided change in Y is considered with reference to a given degraded state represented by unprotected Y, which is assumed to occur if Y is not protected. Third, protected Y may not be equal to current Y. This may occur if 100% protection of current conditions is not possible, or if protected Y is intentionally taken to be a reference state different than current Y.

Piecewise linear benefit model. The relationship between Y and E may have no linear transformation or may be poorly approximated by a straight line. To allow for nonlinearity while retaining the above model form, we consider a piecewise linear curve over the range from E_0 to E_{max}. The curve comprises a series of segments DE_{jl}, such that the relationship between Y and E is linear for the lth segment (equation 2, Figure 2). The linear model in equation 1 is a special case of equation 2 with $l = 1$.

$$DY_{DE_j} = \sum_l \left[DE_{jl} \times \frac{dY}{dE_{jl}} \right] \tag{2}$$

DY for the entire range of E_j is the sum of the DYs over all segments from E_0 to E_{max}. As the lengths of the segments DE_{jl} become very small, the sum approaches an integral.

Benefit model with intermediate variable. In a particular study we may have a poor understanding of the relationship between Y and E. In these cases, we can expand dY/dE_j into multiple terms by introducing into the model a variable, X, intermediate in the causal link between E and Y. The hope is that developing relationships between Y and X and between X and E may be easier than grappling with Y and E directly.

Let X_j denote the level of ecological structure at the jth site, and assume that E_j influences X_j, which in turn influences Y. A different X_j is not needed for every aspect of structure; e.g., if wetland functions are restored by plugging drainage ditches and planting vegetation, X_j can be taken as the area (square meters) of wetland with hydrology and vegetation replaced. Equations 3–5 expand dY/dE_j in equation 1 to include X_j (equation 2 can be similarly expanded).

$$DX_{DE_j} = DE_j \times \frac{dX_j}{dE_j} \tag{3}$$

$$DY_{DX_j} = DX_j \times \frac{dY}{dX_j} \tag{4}$$

Substituting equation 3 into equation 4:

$$DY_{DE_j} = DE_j \times \frac{dX_j}{dE_j} \times \frac{dY}{dX_j} \tag{5}$$

DY_{DX_j} is the change or avoided change in Y given a specific change in structure X on the jth site, and DX_{DE_j} is the change in X_j given a change in effort DE_j. Although DE_j is nonnegative, dY/dX_j and dX_j/dE_j may be positive or negative depending on the definition of Y,

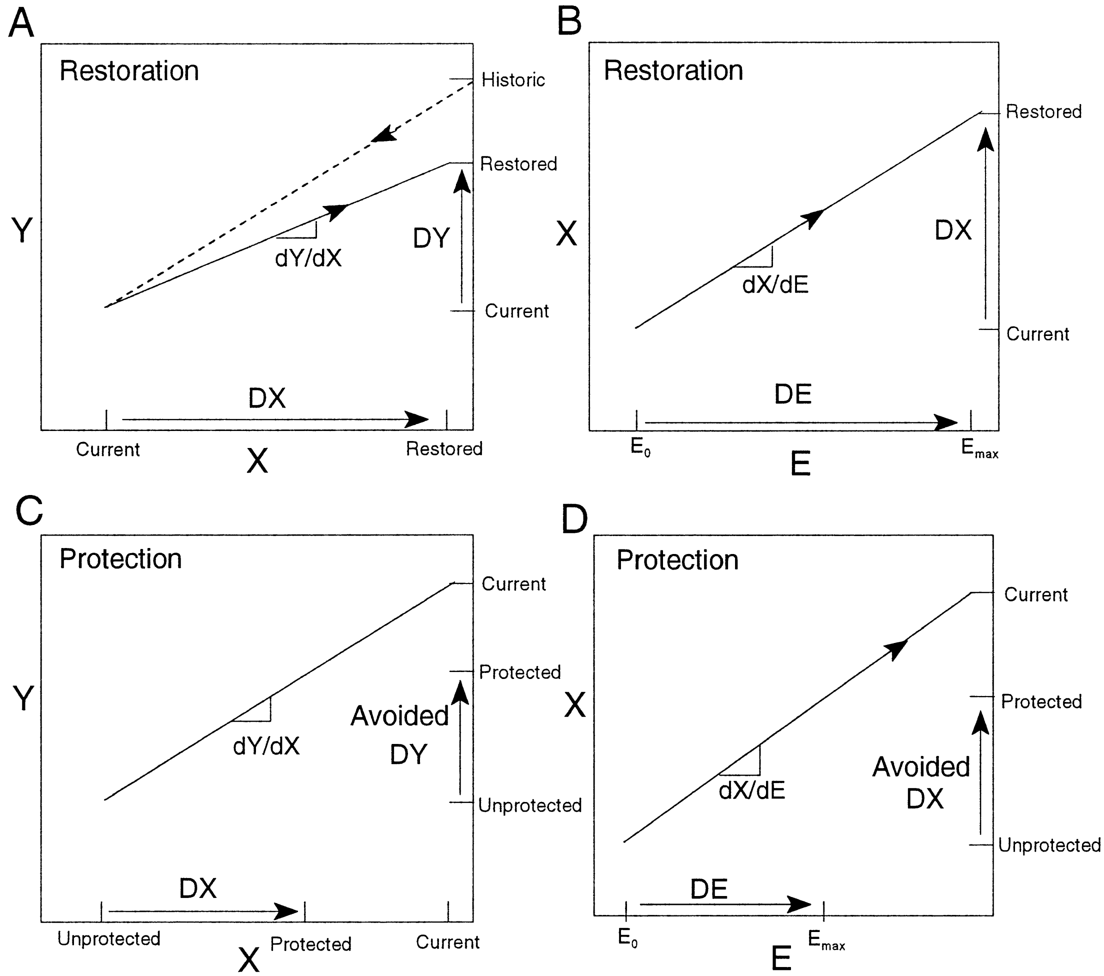

Figure 3. Conceptualization of linear changes in ecological end point *Y* with changes in structure *X*, and of linear changes in *X* with changes in effort *E*. In **A,** the top dashed line shows *Y* decreasing from historic *Y* to current *Y* due to the loss of *X*, whereas the bottom solid line shows the restoration of *Y* from current to restored levels with replacement of *X*. In **B,** replacement of *X* is produced with an expenditure of restoration effort. For protection, **C** shows the avoided loss of *Y* if *X* is maintained at a protected level (protected *X*), relative to the unprotected level (unprotected *X*). The line in **D** shows the avoided loss of *X* as a function of protection effort.

whether X_j has a positive or negative effect on Y, and whether actions increase or decrease the level of X_j. For example, if Y is aquatic species diversity, X_j is the level of nitrogen in a polluted lake, and E_j is restoration effort, then dY/dX_j and dX_j/dE_j would both be negative.

Figure 3 illustrates hypothetical relationships between Y, X, and E. As in Figures 1 and 2, we assume that all other factors are constant throughout and that the dependent variable is in equilibrium with a given level of the independent variable. We also assume that additional effort increases X_j, which promotes restoration of Y; therefore, dY/dX_j and dX_j/dE_j are both positive.

The solid restoration line in Figure 3A shows the increase in Y, relative to current levels, for a given increase in structure X_j. The slope of this line (dY/dX_j) reflects the resilience or ability of the system to respond to an increase in structure; e.g., an increase in biodiversity per unit area of wetland replaced on a site. DX_j is the difference between the structure required for restored Y and the current structure. Contrasted with the restoration line is the dashed degradation line, which shows the loss of Y with a decrease in X_j. Possible causes of a decrease in X_j include land conversion, pollution, and climate change. Because not all causes of a decline in X_j will be addressed by restoration, the trajectory for the loss of Y may differ from the trajectory for the restoration of Y. Figure 3B shows the gain in X_j with an increase in restoration effort. The slope dX_j/dE_j reflects the purchasing power of each additional unit of effort, e.g., hectares of wetland replaced per dollar expended on a site.

For protection, there is only a single line describing both loss and avoided loss of Y with a change in X_j (Figure 3C). This is because protection does not represent a separate process from degradation, but rather prevents a projected loss from fully occurring. The unprotected level of X_j reflects the threat to the system. The slope dY/dX_j reflects the loss of Y avoided by protecting a unit of X_j, or equivalently, the sensitivity of Y to additional loss of X_j. DX_j is the loss of structure avoided by protection and is the structure with protection minus the structure without protection. Figure 3D shows the avoided loss of X_j with an increase in protection effort. As with the restoration line, dX_j/dE_j reflects the purchasing power of each unit of effort.

Multiple Xs may be linked together in a causal sequence (X_{j1} influences X_{j2} which influences X_{j3}, and so on), and the model can be expanded to include as many intermediate X_js as necessary to characterize the link between Y and E_j. This is shown by expanding equation 1:

$$DY_{DE_j} = DE_j \times \frac{dX_{j1}}{dE_j} \times \frac{dX_{j2}}{dX_{j1}} \times \cdots \times \frac{dX_{jn}}{dX_{j(n-1)}} \times \frac{dY}{dX_{jn}} \quad (6)$$

For example, a model to prioritize wetland restoration might include four variables: E_j = dollars, X_{j1} = drainage ditches plugged, X_{j2} = wetland connectivity, and Y = waterfowl species richness. The introduction of X may or may not be helpful, depending on the particular study.

Ranking Multiple Sites and Calculating Benefits for Restoration or Protection

The models in equations 1–6 describe the potential ecological benefit of restoration or protection on a single site. To determine the overall benefit from multiple sites, the list of sites must first be prioritized, since it is assumed that the available effort is not sufficient to restore or protect all sites. Prioritizing sites by the marginal effect of a unit of effort, which is the dY/dE_j term or the terms in its expansion (equation 6), will result in the greatest overall benefit. After sites are ranked by dY/dE_j, these terms can be inserted into extensions of the above equations to give the total projected benefit over any subset of sites chosen from the ranked list.

There are two general approaches to determining dY/dE_j for multiple sites: iteration and scoring (Pressey and Nicholls 1989). Iteration is used where the magnitude of dY/dE_j for each site is dependent on the benefit derived from protecting or restoring the sites previously selected for the ranked list (Margules and others 1988, Pressey and Nicholls 1989, Kiester and others 1996). A list of sites ranked from highest to lowest priority can be built by recursively calculating dY/dE_k conditional on the $k - 1$ sites previously selected for the list. Scoring is used where the benefit derived from each site is independent of the order in which sites are selected. In this case, the ranked list of sites is built by determining dY/dE_j independently and simultaneously for all sites.

If sites are to be compared, an important assumption of our framework is that a unit change in Y is comparable over all areas; that is, we need a common basis (end point) for comparison. For example, if Y is waterfowl abundance, we assume that a gain of 10 waterfowl has the same meaning and constitutes the same benefit regardless of which areas and specific species are restored. If waterfowl abundance is truly the end point of interest, then the fact that different sites may have a different mix of species should not concern us. If this does concern us, then the end point needs to be redefined or different models should be used. It may be that a common end point can not be stated for all sites. Our framework would not be used to rank sites where the end point of concern is different across sites, such as when each site is considered a unique resource and the specific benefits accrued by protecting or restoring a particular site can not be produced on any other site.

Geographic prioritization. Sites can be ranked by dY/dE_j only if this term varies across sites. It is therefore important to determine what attributes may cause intersite differences in dY/dE_j. Such differences may occur if the ecological context differs (Bell and others 1997) or if socioeconomic factors such as land prices and landowner cooperation make restoration or protection more difficult at some sites.

It is also important to examine the spatial variation in the attributes expected to cause differences in dY/dE_j. If there is no spatial trend in these attributes, then the geographic location of sites is not relevant to prioritization, even if sites are ranked over a region. For example, landowner cooperation may vary without regard to geography. In contrast, land prices and landscape condition are likely to vary geographically (McAllister and others 1999). In this case there is a basis for geographic prioritization, where geographic areas containing groups of sites can be ranked for protection or restoration. The advantage of this is that it allows efforts to be targeted to a limited range of sites in locations where restoration or protection may be most beneficial.

Calculating total benefit. After sites are ranked by the absolute-value magnitude of dY/dE_j, the total benefit realized from protecting or restoring the n highest priority sites can be projected. A model of total benefit, corresponding to the piecewise linear model (equation 2), is shown in equation 7. The linear model can be viewed as a special case of the piecewise linear model.

$$DY_{DE} = \sum_{k=m_1}^{m_n} DY_{DE_k} = \sum_{k=m_1}^{m_n} \sum_{l} \left[DE_{kl} \times \frac{dY}{dE_{kl}} \right] \qquad (7)$$

The total benefit DY_{DE} is summed over all sites selected for restoration or protection. The index $k = m_1, m_2, \ldots, m_n$ represents an ordered list of sites ranked from highest (m_1) to lower (m_n) priority, where n is less than or equal to the total number of sites considered for ranking. Thus, the summation index k tallies site benefits beginning at the highest priority site and continues to add the benefits from subsequently ranked sites until the total restoration or protection effort (DE_{TOT}) is used up and n sites have been selected. The inner summation index l tallies over one or more linear segments in the piecewise linear model (equation 2). Note that while we already have the dY/dE_j terms from the ranking, we need to also determine DE_j to calculate total benefit. Restoring or protecting the selected sites produces a larger projected benefit for the total available effort than any other set of sites.

Estimating Model Terms

The model term dY/dE_j, which provides the theoretical basis for ranking sites, was developed without considering how difficult it would be to estimate. Estimates represent a theoretical variable inexactly and have associated uncertainties. The distinction between theoretical variables and estimates has a strong foundation in structural equation modeling (Bollen 1989), where a conceptual variable is represented by an indicator variable measured with error. This distinction is useful for several reasons: new and more accurate estimates can be incorporated without changing the theoretical basis for prioritization; differences among prioritizations can be understood in terms of differences in estimates of a common model, thus facilitating comparisons; and estimation error can be differentiated from the error introduced by using an inadequate model.

We identify three general types of estimates: calculated, confirmed indicator, and judgement indicator estimates (Leibowitz and Hyman 1999). A calculated estimate of dY/dE_j is a direct calculation from curves relating Y to E, such as those in Figures 1–3. The estimate of dY/dE_j is thus in the proper units. Another example of a calculated estimate of dY/dE_j is the product $dX_j/dE_j \times dY/dX_j$ (equation 5).

A confirmed indicator is an attribute that is statistically confirmed to be related to dY/dE_j. For example, if the functional relationship between a variable Y' and the end point Y is known, then dY'/dE_j would be a confirmed indicator of dY/dE_j. Although this parameterized functional relationship can be used to convert a confirmed indicator to the proper units, unlike a calculated estimate a confirmed indicator usually is expressed in units different than for Y.

Both calculated and confirmed indicator estimates of dY/dE_j would typically be derived by quantitatively modeling restoration or degradation processes for representative sites where data are available. These models would relate Y to E (Figure 1), or Y to X and X to E (Figure 3). If representative temporal data are not available for individual sites, a space-for-time substitution (Pickett 1989) may be required to develop replacement and avoided-loss curves. This substitution replaces the curve describing how Y changes with E or X on a single site with a curve describing how Y changes with E or X across sites. For example, Freemark and Collins (1992) used multiple sites to regress the size of forest fragments (X) against the abundance of interior-forest birds (Y). They found that the slope of the relation depends on the amount and connectivity of forest cover in the surrounding landscape. In this case, dY/dX could be taken directly from the regressions, or the amount and connectivity of forest cover could be used as a confirmed indicator of dY/dX.

A judgement indicator estimate of dY/dE_j is a variable assumed to be related to this slope, but unlike confirmed indicators, there is no statistical confirmation of this relationship. To use a judgement indicator, such a relationship must be assumed based on best professional judgement (Leibowitz and Hyman 1999). In spite of the fact that judgement indicators are sometimes called into question (e.g., Conroy and Noon 1996, Schumaker 1996, Flather and others 1997), their use is often unavoidable in environmental management (Abbruzzese and Leibowitz 1997). The quality of inferences based on judgement indicators depends on the strength of the relationship between the indicator and indicated variables, and the error of indicator measurement (Johnson 1995, Flather and others 1997). Although by definition these factors cannot be measured for judgement indicators, they can be evaluated by best professional judgement.

Applications to Prioritization for Wetland Protection and Restoration

The following two case studies illustrate the application of the above framework to prioritizing sites for wetland protection and restoration, respectively. Scoring (as opposed to iteration) was used in both studies

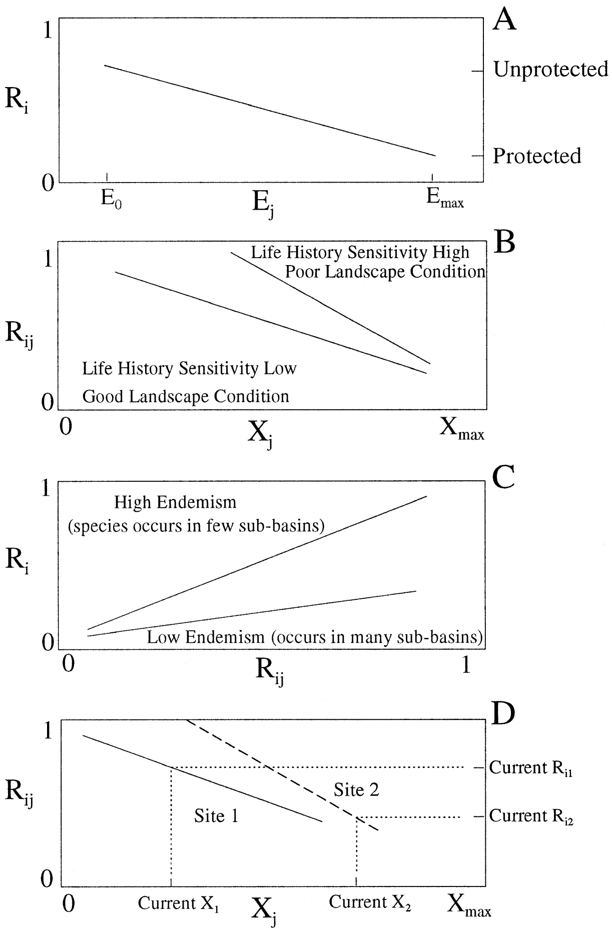

Figure 4. Hypothetical relationships used for prioritization study in EPA Region 7. (**A**) Regional extinction risk for the *i*th species (R_i) is assumed to decrease linearly with protection effort E in the jth subbasin. (**B**) We hypothesize that the slope of a straight-line relationship between local extinction risk (R_{ij}) and wetland area (X_j) depends on landscape quality and species life-history sensitivity. (**C**) We hypothesize that the slope of a straight-line relationship between regional extinction risk (R_i) and local extinction risk (R_{ij}) depends on the degree of endemism for that species. (**D**) Hypothetical example of how site 2 can have a lower current risk yet a higher marginal change in risk as wetlands are lost compared to site 1.

because benefit was considered independent across sites.

Protection Case Study

We helped apply the above framework to prioritize wetland protection efforts within the Environmental Protection Agency's Region 7 (Missouri, Iowa, Kansas, Nebraska). The goal of this prioritization is to maximize the avoided loss of wetland species diversity across the region for a given wetland protection effort. Here we outline the benefit model and briefly discuss the estimation of model terms; details of the study can be found in Schweiger and others (in preparation).

Sites to be ranked for protection are 225 USGS hydrologic cataloging unit subbasins (Seaber and others 1984), with an average size of 3281 km^2. The end point is defined as the risk (denoted by R_i) that a specified wetland species (i) will be extirpated from Region 7 ($0 \leq R_i \leq 1$). An increase in R_i means that regional extinction of species i becomes more likely in less time. The benefit of protecting wetlands in a subbasin is the avoidance of an increase in R_i. Although risk is initially modeled for each species individually, the final ranking of subbasins is based on an index of extinction risk aggregated over all species in a subbasin. Protection effort in the jth subbasin (E_j) is defined as the effort to review Clean Water Act Section 404 permit applications for wetland dredge and fill projects. Note that R_i is defined at the regional scale whereas E_j is defined at the subbasin scale.

The question to be answered by this study is this: in what subbasins would permit review efforts be most profitably allocated to avoid an increased risk of losing wetland species from the region, which is assumed inevitable in the absence of this effort? In general, available resources do not allow all permits to be reviewed intensively, so reviewers must choose a subset of all permits for intensive review. If reviewers knew where each unit of their effort was likely to produce the most benefit, they could focus more attention on permits that come from highly ranked subunits and make sure minimally reviewed permits come from low priority subbasins. Importantly, this prioritization supports, but does not replace, ranking at finer scales. For example, it is entirely possible that a low ranked subbasin contains ecologically important wetlands worthy of protection. Additional screening procedures would be necessary to alert reviewers to these wetlands.

We assumed the unknown relationship between R_i and E_j is reasonably modeled with a straight line (Figure 4A). Equation 1 was used to state the benefit of wetland protection to species i as the increase in regional extinction risk avoided by protecting wetlands in subbasin j:

$$DR_{i_{DE_j}} = DE_j \times \frac{dR_i}{dE_j} \qquad (8)$$

As previously discussed, the goal of maximizing benefits given limited resources is satisfied if we allocate effort to subbasins with the highest dR_i/dE_j. Conditions within each subbasin potentially affect the relationship between R_i and E_j and may create differences in dR_i/dE_j

among subbasins. Moreover, these conditions may vary geographically.

Whether dR_i / dE_j is directly estimated or is expanded using intermediate variables depends on the data available. Species presence / absence data for each subbasin were available from The Nature Conservancy's (TNC) Natural Heritage database (Noss 1987). Six hundred twelve species in the database were classified as wetland species, and this list was used as a surrogate for wetland biodiversity. Also available was a TNC designation of species conservation status (Noss 1987), simple life history information for some species, and remotely sensed land-use data for all sites. There was no information on species population abundances or trajectories or on detailed life-history parameters. Because dR_i / dE_j is difficult to estimate with the available data, this term was expanded by introducing two variables linking R_i to E_j.

The first intermediate variable is wetland area (square meters) in the jth subbasin (denoted by X_j), the ecological structure directly affected by management effort E_j. The second intermediate variable is the extinction risk of species i in the jth subbasin, denoted by R_{ij}. Note that R_{ij} is extinction risk at the subbasin level (local risk), whereas R_i is extinction risk at the regional level (regional risk). These two intermediate variables allowed us to estimate dR_i / dE_j in smaller steps. From equations 3–5, dR_i / dE_j was expanded as follows:

$$\frac{dR_i}{dE_j} = \frac{dX_j}{dE_j} \times \frac{dR_i}{dX_j} = \frac{dX_j}{dE_j} \times \frac{dR_{ij}}{dX_j} \times \frac{dR_i}{dR_{ij}} \qquad (9)$$

The term dX_j / dE_j is the avoided loss of wetland area per unit effort. Because permit review effort is assumed to include desk time only, this term is assumed to be constant across subbasins and thus does not affect the prioritization. If field visits to wetlands were an important part of permit review, however, then dX_j / dE_j may be larger in subbasins where more wetlands could be visited in a shorter time. The next term dR_{ij} / dX_j is the gain in local extinction risk for species i per unit loss of wetland area in the subbasin. Finally, dR_i / dR_{ij} is the gain in regional extinction risk for species i per unit gain in its local extinction risk. It was judged that at this level of expansion of dR_i / dE_j, each model term could be estimated using available data and indicators.

Next, estimates of dR_{ij} / dX_j and dR_i / dR_{ij} were derived. We identified two factors that could influence dR_{ij} / dX_j: landscape condition and life-history sensitivity. We hypothesized that a wetland species in a highly altered landscape is more sensitive to loss of wetland area (i.e., higher dR_{ij} / dX_{ij}) than the same species in a less altered subbasin, due to reduced population sizes, organism health, and dispersal to alternative habitats in the highly altered subbasin. We also hypothesized that dR_{ij} / dX_j is influenced by the extent to which species life history traits confer sensitivity to habitat loss. The expected influence of these factors on dR_{ij} / dX_j are illustrated in Figure 4B. These hypotheses suggest that permit review efforts would be most profitably spent in subbasins where landscape condition is poor and most species have high life-history sensitivity.

We then hypothesized that the second term, dR_i / dR_{ij}, will be greatest for species that occur in only a single subbasin, i.e., narrow endemics, and least for species that occur in every subbasin (Koopowitz and others 1994) (Figure 4C). This suggests that permit review efforts would be most profitably spent in subbasins where many species are highly endemic. Species endemism is commonly used as a criterion in prioritizations for species conservation (e.g., Turpie 1995).

Indicators of landscape condition and life-history sensitivity and direct estimates of endemism were developed from the available data for each species i in subbasin j (Schweiger and others in preparation). The measures were combined to form an index estimate of $dR_i / dX_j = dR_{ij} / dX_j \times dR_i / dR_{ij}$ (see Leibowitz and Hyman 1999 for necessary assumptions). The index estimates allow ranking of species for a specified subbasin and of subbasins for a specified species. Because our objective is to prioritize subbasins rather than species, we summed the estimates of dR_i / dX_j over all species listed for each subbasin to obtain an index of the marginal increase in the regional extinction risk per unit wetland loss in each subbasin. Subbasins were then ranked based on the magnitude of this index.

It is important to distinguish our approach, which prioritizes subbasins based on the increase in risk avoided per unit protection effort, from those that prioritize subbasins based on current risk. Subbasins with the highest current risk are not necessarily those with the highest marginal change in risk. In Figure 4D, site 1 has a higher current risk (R_{ij}), but site 2 would exhibit a greater increase in risk per wetland area lost (dR_{ij} / dX_j).

Restoration Case Study

We also helped apply our framework to the prioritization of wetland restoration efforts across the Prairie Pothole Region (PPR) of the US. The goal of this prioritization is to maximize the amelioration of downstream flooding across the region for a given wetland restoration effort. Flooding in the PPR is an important concern, and this has given rise to an interest in the role of wetlands in flood attenuation. Because of their ability to store and gradually release water, wetlands can function as sinks and reduce flooding by intercepting surface runoff. This study is concerned with downstream flooding only, as opposed to local flooding of farm fields or lakes. A detailed presentation of the study can be found in McAllister and others (1999).

Sites to be ranked for wetland restoration are 119 USGS hydrologic cataloging unit subbasins, with an average area of 2734 km^2. The end point for the PPR study, denoted by Y, is defined as the downstream flood volume (cubic meters) resulting from the drainage of wetlands on the landscape. Restoration effort E_j is defined as the dollars spent to replace currently drained wetlands to predrainage hydrology. We assume that replacement of wetland hydrology leads to restoration of the flood control function of wetlands and that without such effort Y would remain at its current level. The question to be answered is: in what subbasins would restoration of drained wetlands contribute the most to reducing downstream flooding region-wide?

The model used follows equation 1 and a subsequent expansion as in equations 3–5:

$$DY_{DE_j} = DE_j \times \frac{dY}{dE_j} = DE_j \times \frac{dX_j}{dE_j} \times \frac{dY}{dX_j} \qquad (10)$$

The term dX_j/dE_j is the area of wetland (square meters) restored in the jth subbasin per unit effort. The term dY/dX_j is the downstream flood volume (cubic meters), resulting from wetland drainage, attenuated per unit area of wetland restored in the jth subbasin.

As in the Region 7 study, indicators of these model terms were developed by identifying likely influences. The wetland area restored per dollar spent (dX_j/dE_j) is assumed to depend inversely on the landowner subsidy necessary for wetland restoration on private land, which comprises much of the wetland area in the PPR. A reduction in downstream flooding per unit area of wetland restored (dY/dX_j) is realized when runoff normally passing through drained wetlands is held in restored wetlands and prevented from entering a stream. This reduction is assumed to depend on the runoff volume (more runoff means more attenuation possible per unit wetland) and the likelihood and rate at which this runoff is delivered downstream.

Indicators of the above factors—landowner subsidy, runoff volume, and downstream conveyance of runoff—were developed for each subbasin j and used to indicate the ranking criteria dX_j/dE_j and dY/dX_j (McAllister and others 1999). Average land cost was used to indicate landowner subsidy; soil, land-use, and precipitation data were used to indicate runoff volume; and the density of streams and artificial channels on the landscape indicated downstream conveyance of runoff. Indicators were multiplied together to form an index estimate of dY/dE_j for each subbasin. Subbasins were then ranked relative to one another based on the magnitude of this index score. This ranking could be used by regional managers to help target wetland restoration efforts intended to ameliorate the flooding effects of wetland drainage.

Past Prioritizations

The framework can also help to clarify similarities and differences between the variety of prioritizations appearing in the literature. By focusing on the underlying model, differences among prioritizations can be seen as stemming from the use of different models, different assumptions associated with the same model, or different estimators of the same model terms.

The benefit models in equations 1–6 are appropriate if at least two conditions hold: the goal of the prioritization is to maximize benefits produced from limited resources; and a change in the end point, which is defined as a benefit, has the same meaning in all sites. If these conditions do not hold, then other models are more appropriate. We do not suggest that the cost-benefit perspective is appropriate for all situations. If our models are appropriate for a particular prioritization study, however, we can evaluate the prioritization with respect to the ranking criteria, assumptions, and estimates used. We now present a few indicative examples from the prioritization literature to illustrate how past studies may be evaluated in the context of our framework.

Prioritization studies may calculate the quantity DY directly as the difference between Y_{final} and $Y_{initial}$ (equation 1). For example, Fulmer and Cooke (1991) ranked reservoirs for restoration of water quality by their restoration potential, which was defined as the difference between the attainable phosphorus concentration (restored Y in Figure 1A) and the actual summer phosphorus concentration (current Y in Figure 1A). Restored Y was calculated from stream data that represent the best water quality in each ecoregion and a loading model. The use of DY as a ranking criterion, assuming our models are appropriate, requires the assumption that effort is not constrained, which is likely unrealistic, or is constant across sites (equation 1). If, in fact, effort DE is inversely related to dY/dE, then restoring a large number of reservoirs with small DY and DE may produce a larger total benefit than restoring a few reservoirs with high DY and DE.

Studies may alternatively use a measure of DX to estimate DY. In prioritizations for protection, DX often represents the threat of loss of one taxonomic group, such as plants or butterflies, as an indicator of the loss of all taxonomic groups (DY) (Dinerstein and Wikramana-

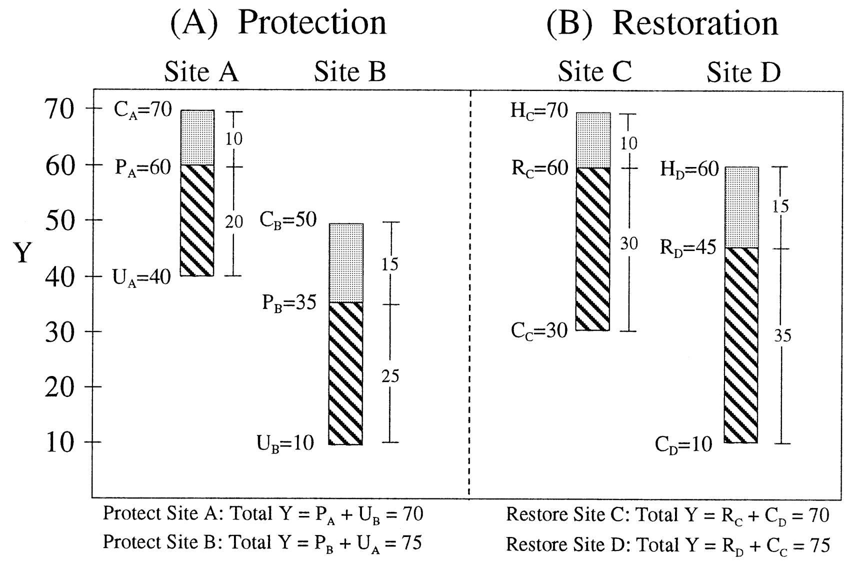

Figure 5. Hypothetical changes in endpoint *Y* for a pair of sites subject to protection (A) or restoration (B). C is current level of *Y*, P is level of *Y* with protection, U is level of *Y* if site remains unprotected, H is historic level of *Y*, and R is level of *Y* after restoration. Equations at bottom show total level of *Y* for both sites together after one is selected for proteciton or restoration. Sites are assumed to have the same *DE*.

yake 1993, Johnson 1995, Turpie 1995, Beissinger and others 1996). *DX* may also represent the threat of habitat loss; for example, Dinerstein and Wikramanayake (1993) used the forested area likely to be lost after 10 years (*DX*) within Indo-Pacific countries as an indicator of biodiversity loss (*DY*). The use of *DX* as a ranking criterion with our models assumes not only that dY/dX is constant across sites (equation 4), but also that *X* is a valid indicator of *Y*. Because taxonomic substitutions and deforestation rates are rough and uncertain proxies for biodiversity threats (Kremen 1992, Weaver 1994, Johnson 1995, Flather and others 1997), in these studies *DX* should be considered a judgement indicator of *DY*.

Still other studies may estimate the dY/dX term of our models (equation 5). Potential indicators of dY/dX for restoration or protection of biodiversity include the degree of habitat fragmentation and connectivity between targeted and supporting habitats (Llewellyn and others 1996, Olson and Harris 1997); incursion of irreversibly degraded land uses (Olson and Harris 1997); and geographic rarity of species (Myers 1988, Master and others 1998). Although dY/dX is a preferred ranking criterion given our models, the associated dX/dE term (equation 5) is usually omitted in such studies, possibly unintentionally. In our protection case study above, we intentionally omitted dX/dE because we considered it constant across sites.

Discussion

When developing new prioritizations, we can ask: How would the prioritization be improved by viewing it in the context of our framework? The use of our framework to prioritize sites can offer five advantages. First, if the cost–benefit perspective is desired, and benefits and costs differ among sites, the models identify fundamental terms to be included in the prioritization. Without a model as a reference, critical terms may be omitted. Second, our framework clarifies the proper prioritization criteria. For example, it can be shown that protecting the site with the highest current *Y* or the smallest difference between current *Y* and protected *Y* (Figure 5A, site A), or restoring the site with the smallest difference between restored *Y* and historic *Y* or between current *Y* and historic *Y* (Figure 5B, site C), may not maximize either total *DY* or total *Y* over both sites for a given effort. The model posits the possibility that the highest ranked site for protection is the one potentially exhibiting the greatest gain or avoided loss rather than the one with the most pristine state or nearest to a reference level. Third, the models provide a clear link between data typically available to prioritize sites and the cost–benefit objective function. New and more accurate estimates can be incorporated without changing the common theoretical basis for prioritization. For example, as data become available, calculated estimates may replace judgement indicator estimates. Fourth, our framework highlights assumptions necessary for a valid ranking of sites when data for estimation are limited. For instance, if the unprotected and protected levels of *X* cannot be estimated, the estimation of *DX* can be simplified by assuming that the unprotected level of *X* is zero or the same across all sites and that the protected level of *X* is equal to the current level (i.e., DX = current $X - 0$; Figure 3C). Without reference to a model, the assumptions behind the use of current *X* to estimate *DX* may not be clear. By varying the assumptions made, the model is very flexible in its ability to encompass a variety of protection and restoration perspectives (e.g., full versus partial protection or restoration). Fifth, the framework allows us to evaluate and compare prioritizations. We can examine whether a particular study appears to be based on the benefit models presented here, and if so, determine whether the criteria used (e.g., DY, DX, or dY/dX) and assumptions required are realistic and consistent with the prioritization goal of maximizing the benefit gained from limited resources.

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