WETLANDS, WILDLIFE, AND WATER QUALITY: TARGETING AND TRADE OFFS

STEPHEN C. NEWBOLD
US EPA, National Center for Environmental Economics
newbold.steve@epa.gov

MARCA WEINBERG
USDA, Economic Research Service
weinberg@ers.usda.gov

Paper prepared for presentation at the American Agricultural Economics Association Annual Meeting, Montreal, Canada, July 27-30, 2003

This research was funded in part by funds from the California Water Resources Center, and by an EPA Star grant, research category: Decision making and valuation for environmental policy, grant number R827932. The views expressed in this paper are those of the authors and do not necessarily represent those of the U.S. Environmental Protection Agency, the Economic Research Service, or the U.S. Department of Agriculture. In addition, although the research described in this paper was funded in part by the U.S. Environmental Protection Agency, it has not been subjected to the Agency's required peer and policy review. No official Agency endorsement should be inferred.

Abstract: Cost-effective targeting of conservation activities has only recently been addressed by economists. Most work to date has focused on finding the best locations to set aside land for the protection of biodiversity. An economic approach to the problem, where biodiversity reserve networks are delineated to maximize the number of species protected subject to a budget constraint, has been shown to be much more cost-effective than the standard approach, where reserve networks are delineated subject to an area constraint, ignoring differences in costs across sites. This paper is among the first to use spatially explicit models of production functions for ecosystem services in an optimization framework for prioritizing sites for wetlands restoration. Tradeoffs between two classes of environmental benefits from wetlands restoration, habitat, and water quality were assessed in the Central Valley of California. Habitat benefits were estimated by a count regression model that relates breeding mallard abundances to the configuration of land use types in the study area, and water quality benefits were estimated by a spatially distributed model of nonpoint source pollution and nutrient attenuation in wetlands. Two decision scenarios were analyzed. In the first scenario the optimal configuration of restoration activity was determined for a small watershed, and in the second scenario sites were selected from those offered for enrollment in an easement program throughout the valley. The results reveal the potential for gains in effectiveness from spatial targeting, and they suggest that there will be substantial tradeoffs between environmental benefits. Maximizing habitat quality in the small watershed yielded a 34% increase in mallard abundance and a 3% decrease in nitrogen loads to the river. In contrast, maximizing water quality resulted in a 25% decrease in nitrogen loads and a 2% increase in mallard abundance. Qualitatively similar results were obtained when sites were selected from a set of offered sites throughout the valley, but the tradeoffs were not as severe. The results also suggest that at traditional funding levels the Wetlands Reserve Program in California could reduce nitrogen loads to rivers by approximately 29,000 kg and increase total mallard abundance in the breeding season by approximately 150 individuals throughout the Central Valley in a given year.

I. Introduction

Land use changes can have profound effects on the quality of the environment. The conversion of natural lands to agriculture and urban uses can increase species extinction rates (Boulinier et al. 2001), affect landscape hydrological processes (Knox 2001), and even exacerbate climate change (Dale 1997). To address the potential effects of land use changes on biodiversity, conservation biologists (e.g. Williams et al. 1996, Stokland 1997, Howard et al. 2000, Margules and Pressey 2000), and increasingly economists (e.g. Ando et al. 1998, Polasky et al. 2001), have begun to develop optimization techniques for selecting areas to set aside as nature reserves.

The standard reserve site selection problem is to maximize the number of species protected in a reserve network that can contain only a limited number of sites (Underhill 1994, Camm et al. 1996). Effectiveness is treated in the standard problem by the constraint on the number of sites (or area) that can be included in the network, so the implicit assumption is that all sites are of equal cost. Ando et al. (1998) pointed out that a more effective method would account for differences in costs across sites and maximize species protection subject to a budget constraint. In their analysis of choosing reserves for endangered species in the continental United States, Ando et al. (1998) estimated that the cost of protecting half of the species in their dataset in a budget-constrained reserve network would be approximately one third the cost of a site-constrained network. Similarly, Polasky et al. (2001) estimated that the cost of including up to 85% of the terrestrial vertebrates in Oregon in a budget-constrained reserve network would be less than 10% of the cost of a site-constrained network.

This recent literature demonstrates that in a world of limited conservation resources systematic approaches to site selection problems can have important implications for environmental policies. However, to date the reserve site selection literature has ignored a number of important features of the general problem of targeting conservation activities cost-effectively. First, reserve site selection applications have not considered the effects of land use changes on species persistence. Land use changes are apparently presumed imminent outside of the reserve network, but these changes are left unspecified. The only management option considered is the establishment of a "reserve" on the selected sites, which would entail protecting the sites in their natural condition and prohibiting any incompatible uses on them. Because the species are already assumed present, no changes on the sites are envisaged aside from their legal status – thus the "reserve" site selection problem, not the "management" site selection problem or the "restoration" site selection problem. A more general specification would allow for enhancement or restoration, where the selected sites could be modified to provide habitat for species that do not currently occur there.

Perhaps because land use changes are not considered in the standard reserve site selection problem, the spatial interactions that affect species viability are generally ignored as well. In lieu of models of species-habitat relationships, most reserve site selection applications have relied on large-scale species range maps, and a species is then considered protected if it occurs on at least one of the sites included in the network. As Polasky et al. (2001) point out, species persistence is generally a complex function of the amount and type of land set aside, as well as its spatial configuration. In the ecology literature, spatially sophisticated treatments of species behavior, population dynamics, and habitat preferences are common (see Tilman and Kereiva eds 1997 for a survey), and Sanchirico and Wilen (1999) used a spatial bioeconomic model to investigate patterns of biomass and effort in a fishery, but spatially explicit models have yet to find their way into terrestrial reserve site selection applications. If the spatial interactions that affect species abundances are modeled explicitly in the site selection problem, then the benefits of including a site in the network will be a function of the size and location of other included sites, because these will affect the configuration of the surrounding landscape. Therefore, the benefits of management on each site, for each species considered, will be endogenous with respect to the decisions to manage other sites. Accounting for this endogeneity requires spatial models of species-habitat relationships, in addition to an explicit treatment of land use changes.

Another feature shared by most reserve site selection applications is an exclusive focus on the biological benefits of preserved areas. Beyond merely defining benefits in terms of the number of species included in the reserve network, Polasky et al. (2001) discuss the possibility of attaching weights to each species to represent their relative values to society, and the possibility of using a measure of taxonomic diversity as the objective. However, there has been no consideration in the literature of other types of ecosystem services from protected areas, such as opportunities for recreation, amenity values provided by proximity to natural areas, water quality maintenance, or flood control benefits, to name but a few possibilities. Finally, because only the maintenance of biodiversity has been considered, tradeoffs between different ecosystem services have been left unexplored as well.

In this paper we addressed a site selection problem where (1) sites were restored instead of protected, so land use changes from management decisions were modeled directly, (2) spatially explicit models of the production functions for ecosystem services were used in the objective function, and (3) two classes of environmental benefits were considered and the tradeoffs between them assessed. The case considered was that of wetlands restoration in the Central Valley of California, and the focus was on habitat and water quality benefits.

II. Methods

II.A. The optimization framework

The modeling approach for this analysis presumes a manager whose problem is to choose sites for wetlands restoration to maximize environmental benefits with a limited budget. The manager is concerned about both habitat and water quality benefits, but does not know the relative values of each and therefore wants to consider the set of solutions that maximize all possible combinations of the two benefits.

Conceptually, tradeoffs between the two benefits can be assessed by delineating a production possibilities frontier (PPF) for the two ecosystem services subject to a limited budget. The PPF summarizes much information about the potential environmental benefits of wetlands restoration and the tradeoffs involved. For example, for the hypothetical PPF in Figure 1 the top left endpoint represents the maximum water quality achievable from wetlands restoration given the budget and the associated improvement in habitat quality $\left(W^{\max}, H \mid W^{\max}\right)$, and the bottom right endpoint represents the maximum habitat improvement achievable given the budget and the associated improvement in water quality $\left(H^{\max}, W \mid H^{\max}\right)$. Each point along the PPF represents a unique configuration of restoration activities, and for each some weighted combination of the two benefits is at a maximum.

To consider these tradeoffs empirically, an optimization model was used where the objective was to maximize a combination of the habitat and water quality benefits of wetlands restoration activities subject to a budget constraint:

$$\underset{\mathbf{x}}{Max}\left[W_H f_H(\mathbf{x}) + W_W f_W(\mathbf{x})\right] \tag{1a}$$

$$s.t. \quad f_C(\mathbf{x}) \leq Budget \tag{1b}$$

In expression 1a, **x** is a vector of binary choice variables where x_i is 1 if cell i is restored and 0 otherwise, $f_H(\mathbf{x})$ is the expected habitat improvement if wetlands are restored in locations represented by **x**, and $f_W(\mathbf{x})$ is the expected water quality improvement. W_H and W_W are the weights applied to the environmental benefits. By varying the weights over the range from 0 to 1, such that in every instance they sum to 1, the production possibilities frontier can be delineated. In expression 1b, $f_C(\mathbf{x})$ is the total cost, which cannot exceed the *Budget*.

As described below, habitat benefits were defined as the increase in abundance of mallards in the breeding season. Mallards support a large recreational hunting industry in the region and are sometimes used as an indicator for the management of other waterbird species. Mallard abundances were estimated by a regression model of the relationship between land use configuration and observed breeding mallard counts. Water quality benefits were defined as the reduction of nitrogen loading rates to surface waters and were estimated by a spatially distributed model of water and nutrient fluxes across the landscape. The models were applied to a grid representation of the Central Valley, where each square cell was 200 meters on a side (4 hectares).[1]

Two decision scenarios were considered. In the first scenario a small watershed was selected and in it the optimal configuration of wetlands restoration was determined. In the second scenario the optimal subset of sites was chosen from those offered for inclusion in an easement program throughout the valley. Solving the optimization problems (expressions 1a and 1b) for each scenario reveals the optimal locations for wetlands restoration and the levels of each ecosystem service expected from the modified landscapes.

II.B. Mallard abundances

Predictions of mallard abundances were based on a regression model developed in Newbold (in prep). This section provides a brief description of the mallard model and describes how it was applied in the optimization scenarios. The mallard model was estimated using count regression techniques based on four years of abundance data at approximately 300 survey sites throughout the Central Valley. The data was collected as part of the North American Breeding Bird Survey, which is a USGS-sponsored program that coordinates counts of all native breeding birds along hundreds of routes throughout North America (Flather and Sauer 1996). Landscape characteristics in the vicinity of each survey site were extracted from a GIS data set of land use throughout the valley, and were related to mallard abundances using a negative binomial regression model. The independent variables used in the model included the percent of each of nine land use types (field crops, pasture, orchards, rice, vineyards, dairy

---

[1] The spatial resolution is much finer here than in previous site selection applications. For example, Ando et al. (1998) took counties as their unit of analysis, and Polasky et al. (2001) used hexagonal cells each of area 63,500 hectares.

operations, urban lands, wetlands, and deep water) within a 400-meter radius circle centered on each survey location (corresponding to the maximum distance surveyors were instructed to count individuals). Squared terms for each land use type were also included, to allow for variable returns to scale from each land use, as were interactions between wetlands with urban lands, rice, agriculture (all types combined), and deep water. The rationale for the inclusion of the interaction terms was that while wetlands were known to be preferred habitat for mallards, it was not known a priori whether the proximity of other land use types would affect the desirability of wetlands. Dummy variables were also included in the models to capture any fixed effects due to unobserved differences across years and routes; the survey locations consisted of 50 sites along each of many routes, and the entirety of each route in each year was surveyed by a different individual.

The model explained a substantial amount of the variation in the data. The pseudo-R^2 value was 0.670 and the squared correlation between the predicted and observed values was 0.311. Based on F-tests, the set of land use characteristics was highly significant ($p < 0.0001$), as were the variables related to spatial effects – the squared and interactive terms ($p < 0.001$), and the route-specific fixed effects ($p < 0.0001$). The variables that were significant individually were: the first and second order wetlands and rice terms, the first order orchard term, and the interaction between wetlands and rice. Mallard abundances exhibited initially increasing, then decreasing, then negative returns to the amount of wetlands and rice nearby. Orchards had a generally negative effect on mallard abundances, as did the presence of high amounts of both wetlands and rice.

The general conclusion from the regression analysis was that habitat selection by breeding mallards is a function of landscape configuration, above and beyond the amount of each land use type present. Specifically, mallards prefer breeding sites near a mix of wet lands (either wetlands or rice) and uplands. The relevance of this for management is that maximum conservation effectiveness can only be achieved if these spatial preferences are taken into account when choosing locations for wetlands restoration activities. Newbold (in prep) estimated that increasing the extent of wetlands by 50% in spatially targeted manner could achieve an increase in mallard abundances on the order of 85%, while non-targeted (evenly dispersed or randomly clumped) restoration could achieve increases on the order of 20-25%.

A parsimonious version of the full model, one that includes only wetlands, rice, and orchard parameters, was used in the following scenarios to estimate the habitat benefits of wetlands restoration. The model is:

$$\hat{\mu} = \exp\left[-1.91 + \left(125.0x_w - 1.01x_w^2 + 46.17x_r - 0.33x_r^2 - 54.67x_o + 0.46x_o^2 - 1.22x_w x_r\right)/1000\right] \quad (2)$$

where $\hat{\mu}$ is the predicted mean mallard abundance, and x_w, x_r, and x_o are the percent of wetlands, rice, and orchards within a 400 meter radius circle of the location of interest.

Equation 2 was estimated based on counts at point locations and associated landscape variables extracted from a polygon representation of the landscape in the vicinity of those locations, as in panel A of Figure 2. In the applications to follow, the landscape was represented as a grid of square cells, each with a single land use type assigned to it. Equation 2 was applied to the grid representation of the landscape by computing $\hat{\mu}$ for the centroid of each cell in the study area, with x_w, x_r, and x_o calculated in the neighborhood of the cell, shown in panel B of

Figure 2. The abundance on the cell was then estimated as $\frac{\hat{\mu}\times 200^2}{\pi\times 400^2}$, and $f_H(\mathbf{x})$ was calculated as the sum of the predicted abundances over all cells. Given the structure of the mallard model, the habitat benefits of restoring a particular cell are endogenous with respect to the decisions to restore nearby cells – the abundance on any given cell can be affected only if it or another cell within 400 meters is restored to wetlands.

II.C. Nutrient loads

The water quality benefits of wetlands restoration were estimated using a spatially distributed model of water and nutrient fluxes applied to the grid representation of the study area. This section contains an abbreviated description of the water quality model; more details can be found in the Appendix.

A water and mass balance approach was used to estimate fluxes of water and nutrients onto, across, and off of the landscape. Each cell was assumed to be homogeneous with respect to land use type, soil type, average precipitation, and population density. The water quality model distinguishes between 15 types of land use: ten types of agriculture, three types of urban land uses, wetlands, and natural uplands. Nitrogen and phosphorus fluxes are largely driven by water flows, but both can be taken up by vegetation, and phosphorus can also be bound to soil particles and effectively immobilized.[2] The water and mass balances for each cell, and for the entire landscape, are calculated for each month in a representative year, based on average monthly rainfall, and average irrigation and fertilizer applications for each crop type represented.

Each cell is treated as a control volume, where inputs of water (precipitation, applied irrigation water, runoff from up-slope cells) must balance outputs of water (evapotranspiration, groundwater infiltration, runoff to down-slope cells). Also on each cell, inputs of nutrients (fertilizer applications on agriculture cells and combined sewer-stormwater overflows from urban cells) must balance outputs of nutrients (plant uptake, infiltration to groundwater, transport with surface runoff). See Figure 3. Estimated irrigation applications are based on average monthly crop water demands (Goldhamer and Snyder 1989), and farmers are assumed to apply enough irrigation water to satisfy crop water demands after average rainfall for the month is accounted for, with some over-application to account for inefficiencies in irrigation technologies. Monthly fertilizer applications are determined by the average yearly applications for each crop (Owens et al. 1998, Padgitt et al. 2000), distributed across months in proportion to irrigation applications.

Overland flows are routed from upland cells along shortest-distance paths to receiving waters. Surface runoff is assumed to travel via drainage ditches on its way to receiving waters, unless it encounters either native upland or wetland cells, in which cases it is assumed to empty onto the cell and travel as surface flow across it. Nutrients are conserved in runoff water as it travels through drainage ditches, but may be taken up by vegetation if it flows over a natural upland cell, or attenuated according to a first order removal process as it flows over a wetland cell. The first order removal function is:

$$r = Q_{in} C_{in} \left(1 - \exp\left(\frac{-kD}{h} \right) \right) \tag{3}$$

---

[2] Phosphorus is much less soluble in water than nitrogen, so phosphorus transport is determined largely by sediment

where *r* is the removal rate [kg/yr], Q_{in} is the concentration of the pollutant in the inflowing water [kg/m^3], *D* is the detention time [yr], *h* is the average depth of the wetland [m], and *k* is the removal rate constant [m/yr] (Kadlec and Knight 1996). Equation 3 embodies all of the physical, chemical, and biological processes in wetlands that serve to attenuate nutrients (Johnson 1991). It implies that the mass of pollutant remaining in the water flowing out of the wetland is an exponentially decreasing function of the amount of time the water spends in the wetland, and so depends on the flow rate of water through the wetland, the wetland's size, and the concentration of pollutant in the inflowing waters.

The first order removal function, along with the spatial arrangement of wetlands with respect to other land use types (which affects the mass of nutrients that each wetland will have the opportunity to attenuate), and with respect to receiving waterways (which affects how much of those nutrients would end up there in the absence of the wetlands), will determine the total water quality benefits of wetlands in the landscape. The model was applied by solving the water balance for the cells highest in the landscape first, those that receive no runoff from other cells, and then working down the landscape towards the rivers to ensure that all contributing water and nutrient inputs were computed for each cell before its outflows were computed.

Table 1 shows baseline estimates for the entire Central Valley, and Figure 4 shows observed vs. predicted loads for a handful of river reaches with sufficient comparison data from USGS monitoring stations. The baseline results suggest that wetlands currently attenuate approximately 10% of nitrogen from surface runoff and 7.5% of phosphorus. Only seven observations for nitrogen loads and 20 for phosphorus loads were available for comparison, but

---

transport (Carpenter et al. 1998).

the model generates predictions of the right order of magnitude, and appears to explain some of the variation across monitoring stations. For the sake of simplicity, only the nitrogen loading component of the model was used in the scenarios considered later.

II.D. Restoration costs

The economic costs of restoration were estimated as the sum of assessed land values and expected wetlands construction costs in the region. Assessor data was obtained for most counties in the Central Valley, and for each the average per hectare total assessed value was computed for all land use types in the data set. In those counties for which data was not available, average values for the surrounding counties were used. Wetland construction costs were estimated by a regression model based on the expected costs of projects being considered for inclusion in the Wetlands Reserve Program (WRP) in California for the year 2000.[3] A model that posits fixed costs plus variable costs proportional to the size of the site (i.e. a linear regression of estimated construction costs on site hectares) fit the data well. The model is:

$$\textit{Estimated construction costs}\ [\$] = 14{,}057 + 83 \times \textit{Area} \quad N = 83 \quad R^2 = 0.96 \tag{4}$$

Where the area of the sites are measured in hectares. As with the benefits of restoration, the costs of restoring each cell are endogenous with respect to the decisions to restore other cells. Because there are fixed costs associated with each restoration project, the total cost depends on the arrangement of restoration sites, not just the total area restored and type of land uses purchased. The model assumes that contiguous sets of cells can be restored as a single project,

---

[3] The Wetlands Reserve Program, passed as part of the 1990 Farm Bill, supports the acquisition of easements and wetlands restoration activities on farmland throughout the United States. In California, approximately 62,000 acres have been enrolled since 1992, and in an average year $11 million is appropriated for the program. Thanks to Alan Forkey (NRCS, Davis, California) for this and other WRP information that we appeal to throughout the paper.

so the manager can take advantage of economies of scale with respect to the consolidation of the restoration projects to be undertaken.

II.E. Solving the optimization problem

Because of the binary nature of the decision variables – each cell can be either chosen and fully restored or left intact – a combinatorial algorithm was required to find the optimal solution to the manager's problem. A conceptually simple (but practically infeasible) algorithm would be to enumerate all possible combinations of cells for restoration, apply the habitat and water quality models for each combination, and compare the predicted mallard abundances and nitrogen loads for each combination to the baseline values. This algorithm is intractable for all but the smallest of problems; it requires 2^N comparisons, where N = the number of cells to be considered. A number of heuristics for combinatorial problems have been developed by operations researchers, and the performance of each generally depends on the nature of the problem to which it is applied (Reeves 1993, Michalewicz and Fogel 2000). In this paper we applied a simple heuristic, but one that takes advantage of the structure of the production functions for ecosystem services to make the problems tractable. The heuristic used to solve the optimization problems can be summarized as follows:

Step 1: Apply the habitat and water quality models assuming no cells are restored, to establish baseline conditions.
Step 2: Calculate the benefit/cost ratio for each cell.
Step 3: Select the cell with the largest benefit/cost ratio for restoration.
Step 4: Update the benefit/cost ratio for all interacting cells.
Step 5: Repeat Steps 3 and 4 until the budget is exhausted.

One way to think about this algorithm is as a walk down the marginal benefits curve. Under certain simplifying conditions an algorithm that excludes Step 4 will guarantee the globally optimal solution (Martello and Toth 1990, Hyman and Leibowitz 2000). One of those conditions, however, is that the benefits and costs of restoring each cell must be independent, which is not the case here. The reason that Step 4 is necessary is that the marginal benefits and costs of restoring any particular cell are endogenous with respect to the decisions to restore other cells. The marginal benefits curve cannot be delineated by merely calculating the benefits and costs of restoring each cell alone – all relevant combinations must be considered. The key is to consider *only* the relevant combinations. By taking advantage of the spatial structure of the production functions for ecosystem services, the algorithm limits its focus in each iteration to only those cells that (can possibly) affect each other's marginal benefits.

Consider first the production function for water quality. It bears repeating here how the spatial arrangement of restoration activities affects the water quality benefits of wetlands. Wetlands attenuate nitrogen from surface water runoff as it flows over them from cells higher in the watershed on its way to the river, so the amount of nitrogen that a wetland attenuates will depend on whether or not another wetland is restored uphill from it. The wetland higher in the watershed would intercept and attenuate some of the nitrogen in the runoff before it reached the wetland lower in the watershed. Two wetlands in a row, along a single overland flow path, do not do twice the work of a single wetland. In Step 2, benefits are computed as if no other cells will be restored, which means that after the first cell is selected the marginal benefits for (some) other cells must be re-computed to account for this change in the landscape. To make the algorithm more efficient, the watershed was divided into its constituent drainsheds, those sets of upland cells that drain to the river through a single cell. In Step 4, only the cells that share the same drainshed as the cell selected for restoration in the previous round need to be updated.

An even more efficient shortcut was available for the habitat quality production function. Mallard abundances were assumed to be a function of the amount and the arrangement of land use types in their immediate vicinity – corresponding to the regression model used to explain the variation in abundances. In Step 4, only those cells within 400 meters of the cell selected in the previous round need updating (refer back to panel B in Figure 2). Because both ecosystem services were considered simultaneously, the cells requiring updating in each round included those in the drainshed, and any extra cells not in the drainshed but still within 400 meters of the last cell selected.

In addition to taking advantage of the spatial structure of the production functions, there is another feature of this algorithm that makes it tractable: the fact that it is a "greedy" algorithm. Once a cell is selected for restoration it cannot be dropped from the set. Greedy algorithms have been shown to be sub-optimal for some problems, including the standard reserve site selection problem (Underhill 1994, Camm et al. 1996), but will nonetheless often yield nearly optimal solutions (Pressey et al. 1996, Csuti et al. 1997). The general nature of the tradeoffs between water quality and habitat benefits of wetlands restoration should not be much affected by slight inefficiencies in the optimization algorithm itself, and the potential gains in effectiveness from a spatially targeted approach to selecting sites can only be measured by the best feasible algorithms in any case.

II.F. Targeting in a small watershed

In the first scenario, a small watershed was selected and within it all cells were considered available for purchase and restoration to wetlands (see Figure 5). The watershed was defined by those upland cells that drain to a reach on the upper Yuba River between two USGS monitoring stations. The upper Yuba River has been designated as an area of interest for water quality improvement by CALFED, a large scale restoration program for the San Francisco Bay-Delta region of California supported by numerous state and federal agencies. The watershed is dominated by agriculture, especially orchards and rice, but also contains urban areas, natural uplands, and wetlands.

The watershed contains 1,047 cells, which represents 4,148 hectares. The drainsheds are shown in Figure 6. The points on the PPF, and the associated sets of sites, were found by solving the optimization problem in (1a) and (1b) where $i = 1,2,\ldots,1047$ – i.e., all cells were considered for restoration independently. The budget was set at $200,000. The average value of agricultural land in the county is approximately $6,572 per hectare, and there are currently 96 hectares of wetlands in the watershed, therefore the budget was sufficient to increase the extent of wetlands by approximately 30%.

In this case the hypothetical manager tries to identify landowners to solicit for participation in an easement program and wants to first approach those landowners whose properties, if restored to wetlands, would yield the greatest increase in habitat or water quality possible given the budget. The manager presumes that landowners would be willing to sell easements to portions of their property in accordance with average assessed land values in the county.

The watershed scenario is relevant because of what it can tell us about the general nature of the optimal spatial distribution of wetlands restoration activities, but it is illustrative of a case that is unlikely to occur in the real world. Here the hypothetical manager can choose from all cells in the watershed, whereas the reality of land use policies is that managers rely heavily on the availability of willing sellers for land acquisition programs. When willing sellers are required for policy implementation, managers will often have to choose from a small subset of all possible sites, which will limit the gains in effectiveness possible from spatial targeting. The second scenario was designed to approximate this more realistic situation.

II.G. Choosing from a set of offered sites

In the second scenario, the entire Central Valley was taken as the study area, and (simulated) sets of sites offered for inclusion in the Wetlands Reserve Program were considered available for purchase and restoration.[4] In the year 2000, WRP sites were selected from 87 offerings throughout the state, 83 of which were in counties that intersect the Central Valley. Because no information was available on the locations of the properties offered, 83 sites (contiguous agriculture cells) were chosen at random according to the county-level distribution of the properties offered in 2000. The number and the size range of potential restoration sites were replicated for each county. For example, in Tulare County ten sites were offered, the smallest of which was 20 hectares and the largest 453 hectares. Therefore in this scenario, 10 sets of contiguous agriculture cells totaling between 20 and 453 hectares were selected at random from Tulare County and treated as offerings to be considered by a hypothetical WRP manager. Sites were selected in a similar manner from other counties, and the entire set served as the basis of the optimization problem. The problem in (1a) and (1b) was then solved for a range of W_H and W_W to delineate the production possibilities frontier. In this case x_i, where i = 1,2,…83, referred to whether or not site i (each of which consisted of multiple contiguous cells) was chosen for inclusion in the WRP. This process was repeated multiple times to generate a distribution of possible outcomes, all loosely analogous to the situation in California in the year 2000.

The same optimization heuristic was used for this scenario. In this case, however, finding the globally optimal solution was virtually guaranteed because the offered sites generally only interact with each other weakly. The water quality benefits of restoring a particular site would only be affected by restoration of other sites that happen to be in the same drainshed, an unlikely occurrence given the small number of sites considered and the large study area. Similarly, the habitat benefits of restoring a particular site will only be affected by restoration of other sites that happen to be within 400 meters. As a result, Step 4 of the optimization algorithm was greatly simplified in this scenario. After Step 2, sites were checked for interactions with other sites and their benefits were adjusted accordingly.

III. Results

III.A. The Watershed Scenario

In the watershed draining to the Yuba River reach, there were seven sets of restored cells that maximize some combination of water quality and habitat benefits for less than $200,000.

---

[4] This case study focuses on wetlands throughout, but these methods can be used to inform other policies that rely on land acquisition or modification, such as the Conservation Reserve Program, the Farmland Protection Program, or the

That is, there are seven points on the production possibilities frontier, which is shown in Figure 7. The maximum increase in mallard abundance is 6.4 individuals, which is 34% of the baseline abundance. The arrangement of restoration activities that yields that solution, shown in panel 7 of Figure 8, leads to a decrease in nitrogen load to the river of 491 kg/year, which is 2.8% of the baseline load. The maximum reduction in nitrogen load possible is 4,365 kg/year, which is 25% of the baseline load. The arrangement of restoration activities that yields that solution, shown in panel 1 of Figure 8, leads to an increase in mallard abundance of 0.28 individuals, which is 1.6% of the baseline abundance.

The spatial arrangement of the restored cells at the endpoints of the PPF follow intuitively from the nature of the production functions. The optimal solution for water quality (panel 1 in Figure 8) consists of restored cells near the river, but disproportionately in those drainsheds with a large amount of contributing land area (refer back to Figure 6). The optimal solution for habitat quality (panel 7 in Figure 8) takes advantage of the inexpensive pasture land (refer back to panel C in Figure 5) to achieve a greater increase in the area of wetlands restored than is possible closer to the river. What is not obvious in panel 7 in Figure 8 is that the restored cells are mostly in the center of the pasture, leaving a ring of uplands around the new wetland and between the wetland and the large patch of rice nearby. Recall that the production function for habitat quality implies that a mix of wet and dry land is optimal for mallards. The intermediate points on the PPF consist of arrangements that strike compromises between these two extremes.

Flood Risk Reduction Program, to name three.

III.B. The WRP Scenario

Table 2 presents summary output for 25 repetitions of the WRP scenario, for simplicity focusing only on the endpoints of the PPFs: the maximum habitat and water quality benefits attainable and the water quality and habitat benefits associated with those solutions. The results were quite consistent across the repetitions, in spite of the wide latitude inherent in the randomization algorithm for defining the locations of sites offered. The most variable outcome was the nitrogen load reduction when mallard abundance was maximized. This is due to the strong spatial effects embodied in the production function for water quality, which also explains why in all repetitions the area of wetlands restored to maximize water quality was less than the area restored to maximize habitat quality (this was also the case in the watershed scenario). The structure of the production function for water quality is such that the benefits of restoring wetlands in very specific locations – close to the river and in those drainsheds with a large amount of contributing area – are much higher than other locations. This is due to the heterogeneity in the spatial distribution of surface runoff and associated pollutant loads across the landscape. In this model nonpoint source pollution is diffuse, but far from uniform.[5] The production function for habitat quality, on the other hand, implies that mallard abundances are less influenced by the spatial arrangement of wetlands. Because only cells within 400 meters of each other interact, there are many arrangements of restored cells that can lead to similar levels of habitat benefits. This can be understood more easily by imagining the production functions for habitat and water quality as general functions of the amount (A) and configuration (C) of

[5] Also note that we have ignored the uncertainty associated with predictions from the water quality model, thereby casting off one of the main difficulties in controlling nonpoint source pollution. Partly because it is difficult to predict

wetlands: $H = h(A,C)$ and $W = w(A,C)$. The marginal rate of substitution between the two "inputs," $-\frac{dA}{dC}$, is larger for the water quality function than the habitat function. This general feature of the production functions can also explain why the foregone water quality benefits of the solution that maximizes restored wetland area are greater, in percentage terms, than the foregone habitat benefits (compare $H|Area^{max}$ with H^{max} and $W|Area^{max}$ with W^{max} in Table 2). In this case, maximizing wetland area is a more effective heuristic for increasing mallard abundance than it is for reducing nitrogen loads.

The average nitrogen attenuation rate on restored cells also points to the increased benefits possible from spatial targeting. The average attenuation rate in the restored wetlands was approximately 45.4 kg/ha, which is much higher than the average of 3.1 kg/yr in existing wetlands (according to baseline outputs from the water quality model), but much lower than the 273 kg/ha in restored wetlands the example watershed scenario. By way of comparison, in their analysis of the cost-effectiveness of restored wetlands for controlling nitrogen loss in the Mississippi Basin, Ribaudo et al. (2001) assumed an average attenuation rate of 200 kg/ha. The results presented here imply that the average rate will depend crucially on the spatial arrangement of the restored wetlands. The difference between the qualitative results from the watershed scenario and the WRP scenario arises because the manager in the watershed scenario had a completely free hand to spatially target restoration activities; all cells were treated as available for purchase and restoration. In the WRP scenario the manager was constrained by the initial set of offered sites.

---

are efficient policies difficult to design and implement. Therefore, the results here should be interpreted as upper bounds on the gains in effectiveness possible from a spatially targeted wetlands restoration strategy.

The average density of breeding mallards on restored areas in the WRP scenario was 0.16 individuals/ha, which is virtually the same as 0.15 individuals/ha in the watershed scenario. Again, because the marginal rate of substitution between wetland area and configuration is lower for the habitat function, it follows that there would be lower gains in effectiveness from spatial targeting possible, and therefore a less pronounced difference between the two scenarios.

IV. Discussion

This case study was based on two empirically specified models of ecosystem services integrated into a numerical optimization framework. Analytic treatments of problems of spatial targeting can be enlightening (Wu and Boeggess 1999, Bhat et al. 1999), but they can only incorporate a limited amount of realism. When the crucial features of the questions at hand include spatial heterogeneity and interactions that lead to endogenous management benefits, calibrated simulation models will often provide the most effective means of addressing the problem. One goal here is to develop methods and uncover general principles that might be applicable across a range of settings, but another goal is to put actual numbers on the potential gains in effectiveness from a spatially targeted approach to selecting sites and the tradeoffs between multiple objectives. The results showed that there could be large potential gains from spatial targeting when the provision of ecosystem services depends on the configuration of the landscape, even when policy implementation is constrained by the availability of willing participants.

A cost-effectiveness framework was used here, but it can be extended to address questions of efficiency as well. By incorporating estimates of the values of water quality and species abundances one could determine the socially optimal amount and arrangement of wetlands in the landscape. In fact, delineating the PPF might be the first logical step in this larger endeavor, as it will determine the resolution of the economic information required. Consider again the PPF in Figure 7. To find the socially optimal solution (to choose the best point on the PPF), information on the relative values of habitat and water quality would be required. The solution that maximizes habitat quality would be socially optimal if $\frac{W_H}{W_W} > 2000$; i.e., if an extra mallard in the breeding season is worth more than a reduction in nitrogen loads of 2,000 kg/year. At the other end of the PPF, the solution that maximizes water quality would be socially optimal if $\frac{W_H}{W_W} < 333$. Within this range the manager has difficult choices to make; outside of it, the manager need focus only on providing the one relevant ecosystem service and need not worry about tradeoffs. We do not know where the socially optimal tradeoff between water quality and habitat quality is, but the point here is that by delineating the PPF a decision-maker can greatly narrow the set of choices that need further consideration, and bounds can be put on the resolution of the information required for that task.

Economic studies of management options for nonpoint source pollution are common, but many of these use farm-level models and assume that pollutant load reductions at the edge of the field will translate into similar reductions to receiving waters (e.g. Taylor et al. 1992, Randhir and Lee 1997). Perhaps the best example in the economics literature of the use of wetlands for controlling nonpoint source pollution is the study by Ribaudo et al. (2001), who compared the costs of source reduction versus wetlands restoration for reducing nitrogen pollution in the Mississippi Basin. They used a fully developed economic model, but did not address the question of optimal spatial targeting except at a very coarse resolution, and assumed that all wetlands would attenuate the same amount of nitrogen, no matter their placement in the landscape.

V. Conclusions

Economists have made significant contributions to methods for selecting sites for nature reserves. This paper expands on the standard reserve site selection problem by incorporating spatially explicit functions of the effects of land use changes on ecosystem services, and by assessing the tradeoffs between two environmental benefits from wetlands restoration. The results presented in this paper show that there may be significant tradeoffs between environmental benefits to consider when implementing wetlands conservation policies. The relatively small body of research on wetlands policies by economists suggests that the socially optimal rate of wetlands conversion is lower than the historical average, and possibly negative (e.g. Brown and Hammack 1973, Stavins 1990, Barbier 1994). This is not to say that no proposed wetland conversions will be beneficial, just that society would generally be better served by restoring more wetlands than are converted to other uses. The results presented in this paper suggest that both the benefits and the costs will depend on the spatial configuration of restoration activities. Furthermore, this case study provides an indication of the magnitude of the tradeoffs involved. There was a wide range of possible outcomes in the sample watershed –

from a 25% decrease in nitrogen loads with a 1.6% increase in mallard abundance, to a 34% increase in mallard abundance with a 2.8% decrease in nitrogen loads.

The natural processes that determine the levels of benefits delivered by wetlands or other set-aside lands often have strong spatial components, and in these situations spatially explicit models can help managers more effectively target restoration activities. Results from the WRP scenario show that the gains from spatially targeted site selection can be substantial, even when selecting from a pre-determined set of sites. However, the gains in effectiveness will be greater for some environmental benefits than others, depending on the nature of their production functions. In the WRP scenario, spatial targeting for water quality enhancement delivered on average three times the reduction in nitrogen loads than the solution that maximized restored wetland area, while targeting for habitat quality delivered only a 7% greater increase in mallard abundance. By most measures the potential environmental benefits from wetlands restoration appear substantial. This research demonstrates that the tradeoffs between the environmental benefits can be substantial as well, and in a spatially heterogeneous world the limited resources available for conservation can have maximum impact only if that heterogeneity is considered when designing and implementing environmental policies.

References

American Public Works Association and University of Florida, Nationwide Evaluation of Combined Sewer Overflows and Urban Stormwater Discharges: Volume I, Executive Summary, USEPA Report EPA-600/2-77-064a, 1977.

Ando, A., Camm, J., Polasky, S., and A. Solow. 1998. Species distributions, land values, and efficient conservation. *Science* 279: 2126-2128.

Barbier, E.B. 1994. Valuing environmental functions: Tropical wetlands. *Land Economics* 70(2): 155-173.

Bhat, M.G., Fister, K.R., and S. Lenhart. 1999. An optimal control model for the surface runoff contamination of a large river basin. *Natural Resource Modeling* 12(2):175-195.

Boulinier, T., Nichols, J.D., Hines, J.E., Sauer, J.R., Flaterh, C.H., and K.H. Pollock. 2001. Forest fragmentation and bird community dynamics: Inference at regional scales. *Ecology* 82(4):1159-1169.

Brown, G.M., and J. Hammack. 1973. Dynamic economic management of migratory waterfowl. *Review of Economics and Statistics* 55(1): 73-82.

Camm, J.D., Polasky, S., Solow, A., and B. Csuti. 1996. A note on optimal algorithms for reserve site selection. *Biological Conservation* 78:353-355.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N., and V.H. Smith. 1998. Nonpoint pollution of surface water with phosphorus and nitrogen. *Ecological Applications* 8(3):558-568.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L, Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Huso, M., and K. Sahr. 1997. A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation* 80:83-97.

Dale, V.H. 1997. The relationship between land-use change and climate change. *Ecological Applications* 7(3):753-769.

Flather, C.H., and J.R. Sauer. 1996. Using landscape ecology to test hypotheses about large-scale abundance patterns in migratory birds. *Ecology* 77(1):28-35.

Goldhamer, D.A., and R.L. Snyder. 1989. Irrigation Scheduling: A Guide for Efficient On-Farm Water Management. University of California, Division of Agriculture and Natural Resources, Publication 21454.

Howard, P.C., Davenport, T.R.B., Kigenyi, F.W., Viskanic, P., Baltzer, M.C., Dickinson, C.J., Lwanga, J., Matthews, R.A., and E. Mupada. 2000. Protected area planning in the tropics: Uganda's national system of forest nature reserves. *Conservation Biology* 14(3):858-875.

Johnston, C.A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Critical Reviews in Environmental Control* 21(5-6):491-565.

Kadlec, R.H., and R.L. Knight. 1996. Treatment Wetlands. Lewis Publishers, Boca Raton, Florida.

Knox, J.C. 2001. Agricultural influence on landscape sensitivity in the Upper Mississippi River Valley. *Catena* 42(2-4):193-224.

Margules, C.R., and R.L. Pressey. 2000. Systematic conservation planning. *Nature* 405: 243-253.

Martello, S., and P. Toth. 1990. Knapsack Problems: Algorithms and Computer Implementations. John Wiley and Sons, New York.

Michalewicz, Z., and D.B. Fogel. 2000. How to Solve It: Modern Heuristics. Springer-Verlag Press, Berlin.

Owens, L.B., Van Keuren, R.W. and W.M. Edwards. 1998. Budgets of non-nitrogen nutrients in a high fertility pasture system. *Agriculture, Ecosystems and Environment* 70:7-18.

Padgitt, M., Newton, D., Penn, R., and C. Sandretto. 2000. Production practices for major crops in U.S agriculture, 1990-97. Resource economics division, Economic Research Service, USDA.

Polasky, S., Camm, J.D., and B. Garber-Yonts. 2001. Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. *Land Economics* 77(1):68-78.

Pressey, R.L., Possingham, H.P., and C.R. Margules. 1996. Optimality in reserve selection algorithms: when does it matter and how much? *Biological Conservation* 76:259-267.

Reeves, C.R., editor. 1993. Modern Heuristic Techniques for Combinatorial Problems. Halsted Press, New York.

Rhandir, T.O., and J.G. Lee. 1997. Economic and water quality impacts of reducing nitrogen and pesticide use in agriculture. *Agricultural and Resource Economics Review* 26(1):39-51.

Ribaudo, M.O. 1989. Water quality benefits from the conservation reserve program. AER-606. U.S. Dept. Agr., Econ. Res. Serv., Feb.

Ribaudo, M.O., Heimlich, R., Classen, R., and M. Peters. 2001. Least-cost management of nonpoint source pollution: Source reduction versus interception strategies for controlling nitrogen loss in the Mississippi Basin. *Ecological Economics* 37:183-197.

Reeves, C.R. (editor). 1993. Modern Heuristic Techniques for Combinatorial Problems. Halsted Press, New York.

Sanchirico, J.N, and J.E. Wilen. 1999. Bioeconomics of spatial exploitation in a patchy environment. *Journal of Environmental Economics and Management* 37(2):129-150.

Stavins, R.N. 1990. Alternative renewable resource strategies: A simulation of optimal use. *Journal of Environmental Economics and Management* 19:143-159.

Stokland, J.N. 1997. Representativeness and efficiency of bird and insect conservation in Norwegian boreal forest reserves. *Conservation Biology* 11(1): 101-111.

Taylor, M.L., Adams, R.M., and S.F. Miller. 1992. Farm-level response to agricultural effluent control strategies: The case of the Willamette Valley. *Journal of Agricultural and Resource Economics* 17(1):173-185.

Tilman, D. and P. Kareiva (1997). Monographs in Population Biology; Spatial ecology: The role of space in population dynamics and interspecific interactions. Monographs in Population Biology; Spatial ecology: The role of space in population dynamics and interspecific interactions. D. Tilman and P. Kareiva. Technical Books, 36 University Place, Princeton, New Jersey 08540 ; Oxford, England, UK 1997., Princeton University Press ; Princeton University Press: xiv+368p.

Underhill, L.G. 1994. Optimal and suboptimal reserve selection algorithms. *Biological Conservation* 70:85-87.

Viessman, W., Jr., Lewis, G.L., and J.W. Knapp. 1989. Introduction to Hydrology, 3rd Edition. Harper Collins Publishers, New York.

Williams, P., Gibbons, D., Margules, C., Rebelo, A., Humphries, C., and R. Pressey. 1996. A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conservation Biology* 10(1):155-174.

Wu, J., and W.G. Boeggess. 1999. The optimal allocation of conservation funds. Journal of *Environmental Economics and Management* 38:302-321.

Woodward, R.T., and Y. Wui. 2001. The economic value of wetland services: a meta-analysis. *Ecological Economics* 37:257-270.

Appendix – The water quality model

The general specification for the water and mass fluxes across a cell are as follows:

For the cell itself:

$$P + Irr = CellET + CellLeach + CellRunoff \tag{A1}$$

For the associated ditch:

$$Runon + CellRunoff = DitchLeach + TotalRunoff \tag{A2}$$

P is the average rainfall for the cell, and the remaining terms are calculated using a basic curve number approach (Viessman et al. 1989, Goldhamer and Snyder 1989) as follows:

$$S = (1000 / CN) - 10 \tag{A3a}$$
$$R = \min\{S, P\} \tag{A3b}$$
$$S_r = S - R \tag{A3c}$$
$$Irr = \max\{0, (K_C \times ET_0 - R) / AE\} \tag{A3d}$$
$$CellET = K_C \times ET_0 \tag{A3e}$$
$$CellRunoff = \max\{0, P - S\} + \max\{0, Irr \times (1 - AE) - S_r\} \tag{A3f}$$
$$CellLeach = \max\{0, P + Irr - K_C \times ET_0 - CellRunoff\} \tag{A3g}$$

S is the total infiltration capacity and is a function of the curve number, which is determined by the hydrologic class of the soil (A, B, C, or D) and the land use type of the cell. *R* is the effective rainfall, the amount that infiltrates into the soil and is available for plant uptake. If the amount of precipitation in a month, *P*, is less than the total infiltration capacity, then all of it is assumed to infiltrate and become available for plant uptake. Otherwise the excess contributes to the cell's runoff, by equation A3e. S_r is the residual infiltration capacity, the water holding capacity of the soil after all precipitation has infiltrated. *Irr*, the amount of irrigation water applied, is determined by equation A3d and is a function of the crop coefficient, K_c, the reference crop water demand, ET_0, the amount of effective rainfall, *R*, and the irrigation application efficiency, *AE*. The model assumes that farmers always apply just enough irrigation water to meet crop demands, taking into account the amount of rainfall expected in the month and inefficiencies in their irrigation technology. For all agriculture cells, therefore, the amount of evapotranspiration will equal the crop water requirements, as determined by equation A3e. The amount of water that runs off the cell is determined by equation A3f: runoff from rainfall is generated only if the amount of precipitation exceeds the infiltration capacity, and runoff from irrigation applications is generated only if the excess irrigation water, as determined by the application efficiency, exceeds the residual infiltration capacity. Finally, the amount of water that leaches below the root zone is given by equation A3g. Leaching on the field will only occur if the amount of precipitation exceeds the crop water demand. Otherwise, all excess applied water leaves the cell as surface runoff. Some of this surface runoff, however, can leach to the groundwater in the adjacent and downstream ditches, and through any natural upland or wetland cells it encounters on its way to the receiving water body.

Native upland and wetland cells are treated slightly differently. For these cells equations A3c and A3d do not apply, and equations A3b, A3e, A3f, and A3g are modified as follows:

$$R = \min\{S, P + Runon\} \tag{A3h}$$

$$CellET = \min\{K_C \times ET_0, R\} \tag{A3i}$$

$$CellLeach = \max\{P + Runon, S\} \tag{A3j}$$

$$CellRunoff = P + Runon - CellET - CellLeach \tag{A3k}$$

The water balance for urban cells is calculated using a different approach. This is because runoff from urban areas depends on a suite of different factors, including the percent of impervious surface, detention storage, and population density. An empirical model of annual runoff and pollutant loads developed by the American Public Works Association and University of Florida (1977) was disaggregated to a monthly time step and applied to those cells that represent urban areas. The runoff model for urban cells is:

$$I = 9.6PD^{(0.573 - 0.0391 \times \log(PD_d))} \tag{A4a}$$

$$DS = 0.25 - \frac{0.1875I}{100} \tag{A4b}$$

$$AR = \left(0.15 + \frac{0.75I}{100}\right)P - 5.345DS^{0.5957} \tag{A4c}$$

$$DWF = 1.34PD \tag{A4d}$$

$$CellRunoff_m = \frac{AR \times P_m}{\sum_m P_m} + \frac{DWF}{12} \tag{A4e}$$

In the above equations I is the percent of impervious surface for the cell, PD is the population density of the cell, DS is the depression storage, AR is the annual storm runoff, and DWF is the annual dry weather flow, which is based on an average flow of 379 *l*/person/day. $CellRunoff_m$ is the total runoff from the urban cell in month m. Stormwater runoff for the cell is distributed across the months according to the distribution of rainfall, and the dry weather flow is distributed across the months evenly.

Average values for the first-order removal rate constant, k, for total nitrogen and total phosphorus from the literature on constructed wetlands for wastewater treatment are around 22 and 12 m/yr (Kadlec and Knight 1998). Values of 11 and 6 m/yr were used here.

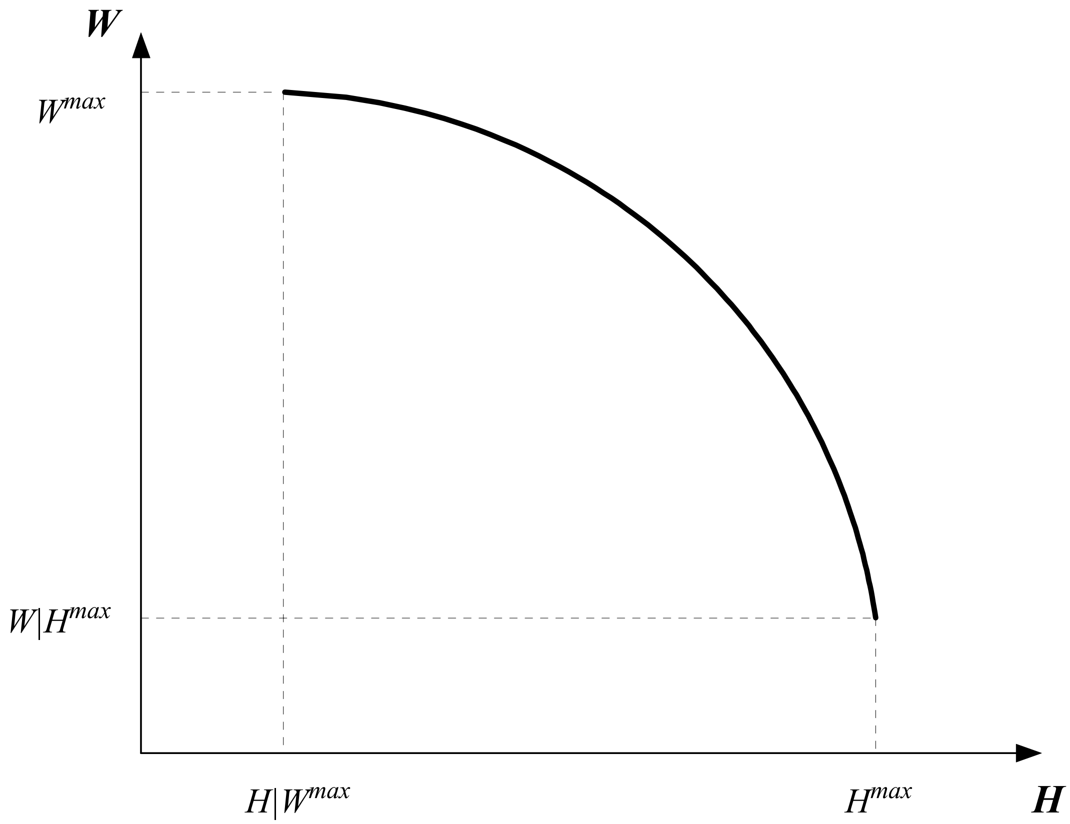


Figure 1. A hypothetical production possibility frontier for two ecosystem services from wetlands. The frontier is defined by those sets of restoration sites that yield the maximum possible amount of some weighted combination of water quality (*W*) and habitat (*H*) benefits for a fixed budget. For any point on the frontier, no rearrangement of restoration activities can yield an increase in one of the benefits without decreasing the other. All other feasible solutions will lie somewhere inside (to the left of and below) the frontier; for each of these some point on the frontier will yield more of one or both of the ecosystem services.

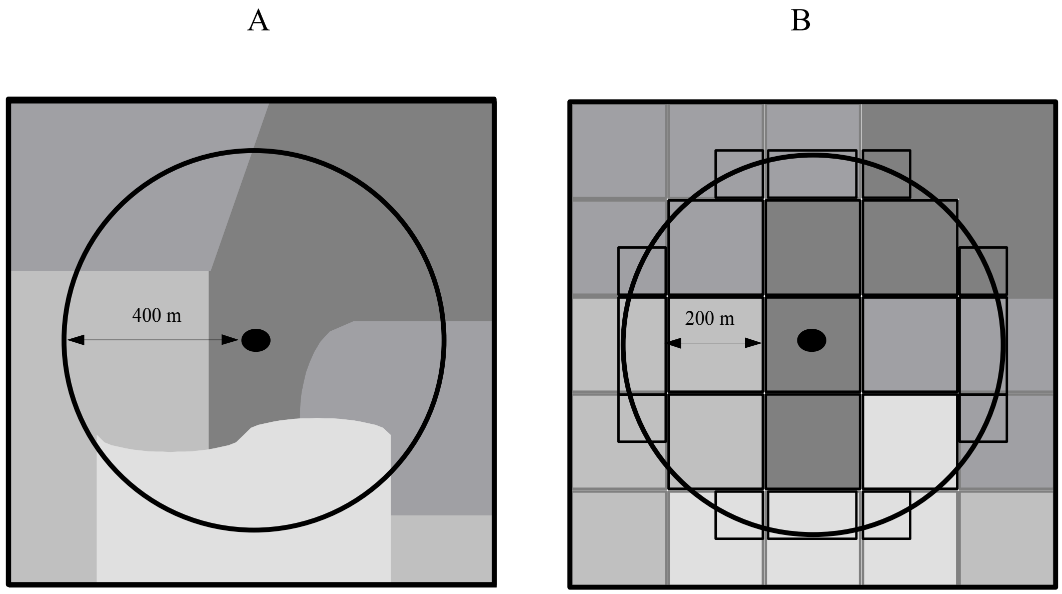


Figure 2. The mallard model was estimated based on a polygon representation of the landscape, as in panel A. In the optimization scenarios the model is applied to a grid representation, as in panel B. The 400 meter radius circular neighborhood is approximated by the polygon made up of the focal cell, the eight immediately surrounding cells, and portions of twelve of the cells surrounding those, as pictured in panel B.

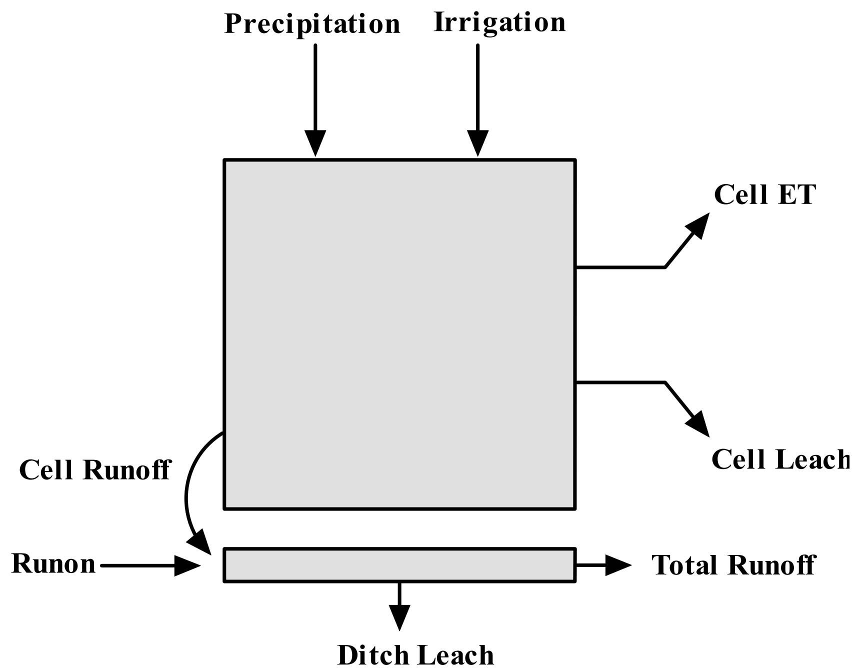
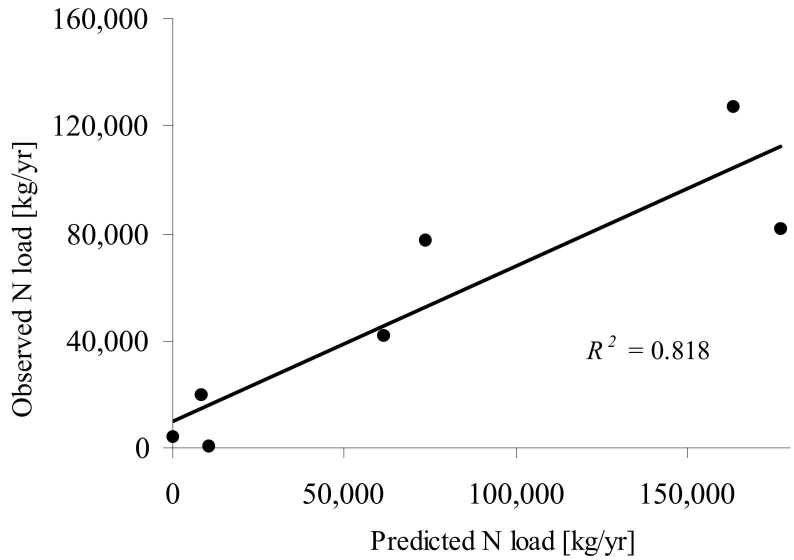


Figure 3. Basic water balance for a representative cell. In a given month the fluxes into and out of both the cell and ditch must balance. Nitrogen enters the cell with irrigation water and leaves the cell with runoff and leached water. Nitrogen enters the ditch from the cell and from runon from upstream cells, and leaves the ditch with runoff to downstream cells and leached water.

Table 1. Baseline outputs from the water quality model for the entire Central Valley

	Total fluxes
	[1000m³/yr]
Precipitation	18,379,348
Irrigation + urban water demand	37,202,691
Evapotranspiration	37,140,702
Infiltration to groundwater	13,723,558
Runoff to surface waters	4,716,129
	[kg/yr]
Nitrogen inputs	352,170,417
Nitrogen uptake	279,818,994
Nitrogen leaching	63,056,919
Nitrogen runoff	9,294,475
Phosphorus inputs	169,163,992
Phosphorus uptake	127,055,646
Phosphorus immobilized	41,430,844
Phosphorus runoff	677,500
Nitrogen attenuated in wetlands	858,515
Phosphorus attenuated in wetlands	41,882



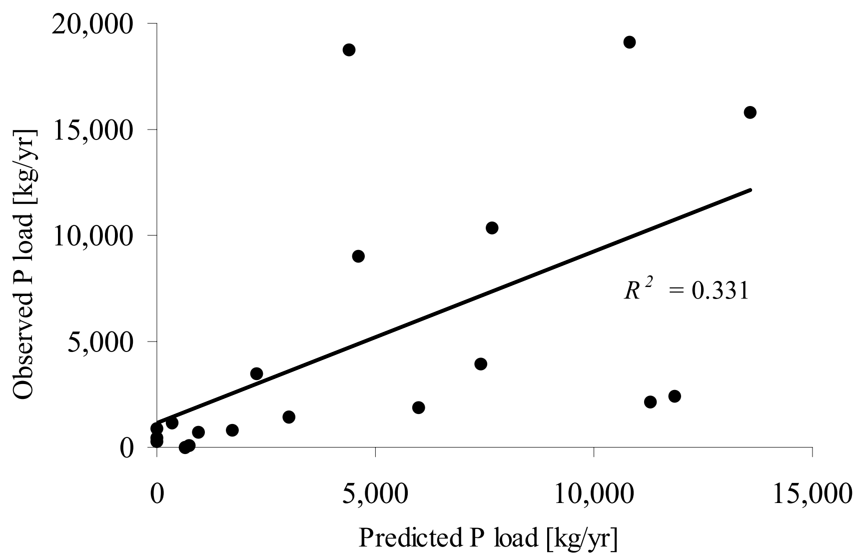


Figure 4. Observed loads at USGS monitoring stations vs. predictions from the water quality model. There were 7 observation for nitrogen and 20 for phosphorus.

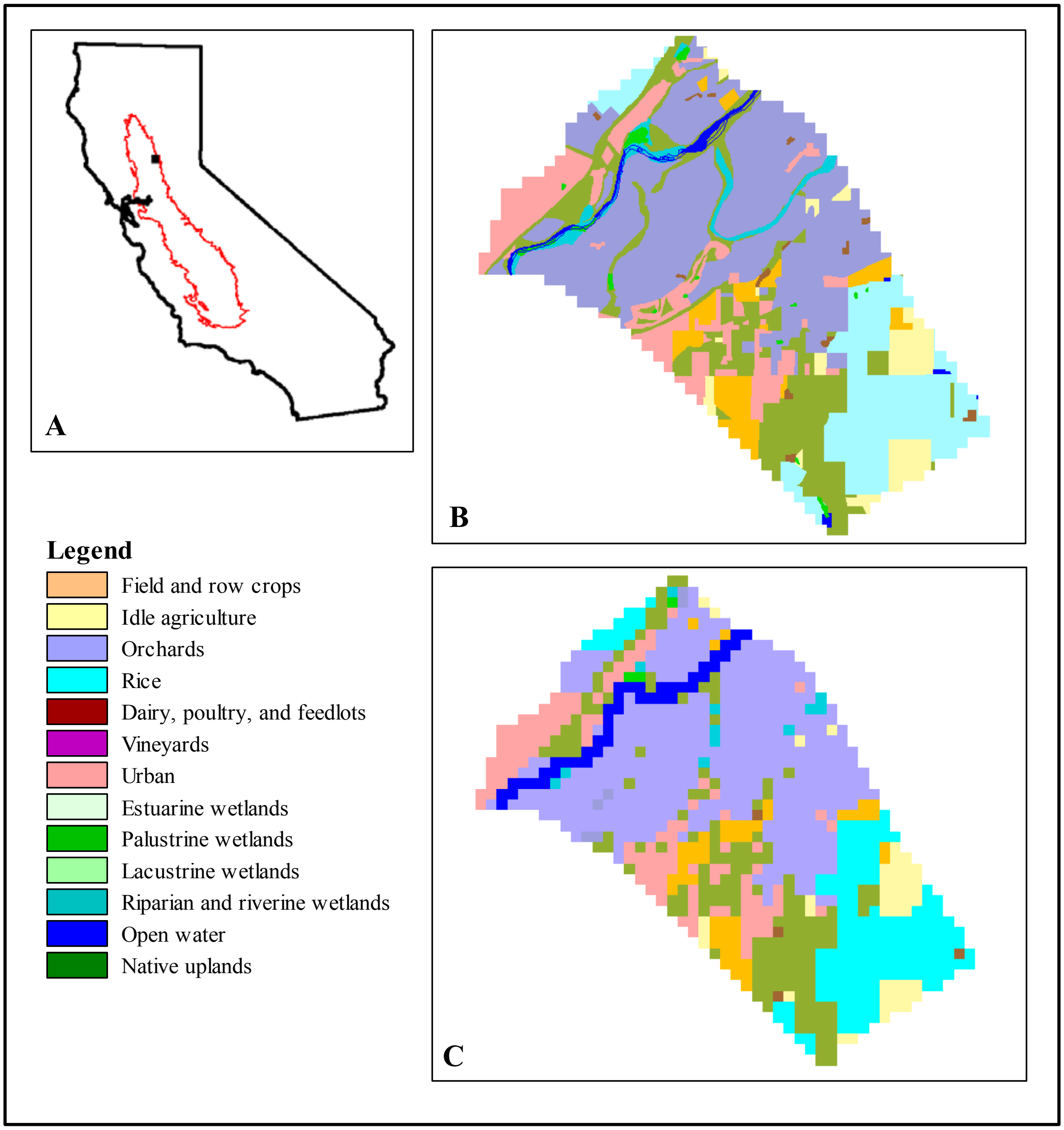


Figure 5. The watershed scenario – Panel A shows the location of the watershed in the Central Valley. Panel B shows the raw land use data, the polygon representation of the distribution of land uses in the watershed. Panel C shows the grid representation of the watershed, which was used for the optimization modeling. The legend provides a key to the land use types. The entire Central Valley covers 5.86 million hectares; the example watershed covers 4,148 hectares.

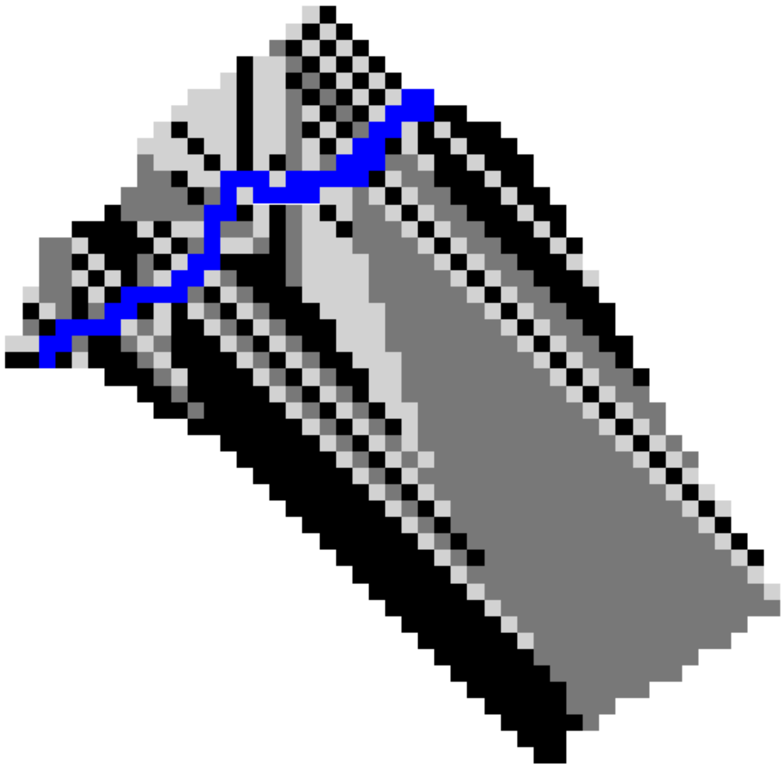


Figure 6. Drainsheds in the Yuba River reach watershed. Contiguous cells of the same color all ultimately drain to the river through a single cell. The drainsheds are independent of each other; water and mass fluxes in one drainshed never cross into another. Because of this, at each stage in the optimization algorithm only the cells in the drainshed from which the last cell was selected need to be updated.

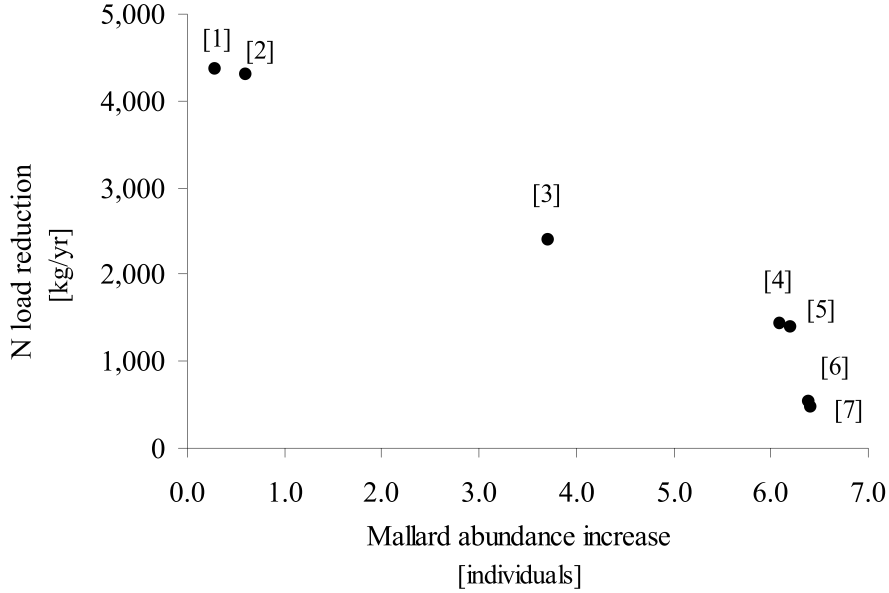


Figure 7. The PPF for nitrogen load reduction and mallard abundance increase in the Yuba River reach watershed

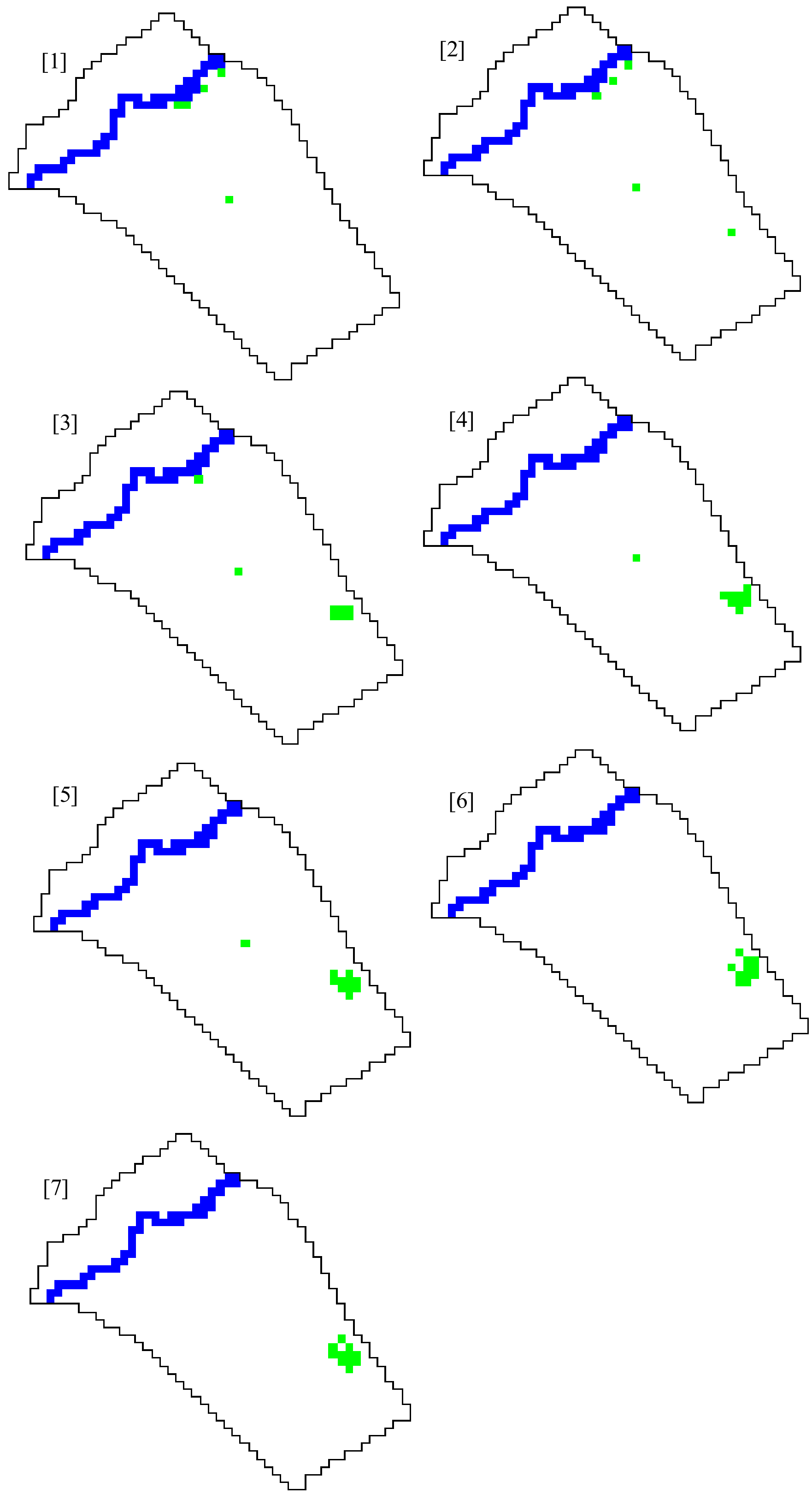


Figure 8. The spatial arrangements of wetlands restoration activities associated with the seven points on the PPF for the example watershed

Table 2. WRP scenario results

	Average	Coefficient of Variation
	[kg/yr]	
Baseline N load	9,191,920	NA
W^{max}	29,266	0.237
$W\|H^{max}$	8,120	0.607
$W\|Area^{max}$	17,334	0.295
	[individuals]	
Baseline mallard abundance	56,267[6]	NA
H^{max}	442.0	0.104
$H\|W^{max}$	285.4	0.187
$H\|Area^{max}$	405.8	0.136
	[hectares]	
Baseline wetland area	317,908	NA
$Area^{max}$	1,047	0.087
$Area\|W^{max}$	755	0.141
$Area\|H^{max}$	994	0.110

Note: The $X|Y^{max}$ notation refers to the level of X that results from the maximization of Y. For example, $Area|W^{max}$ is the area of wetlands restored when water quality benefits were maximized.

[6] The baseline value for total mallard abundance in the Central Valley was calculated by applying the regression model to a sample of 50,000 of the 1.4 million cells in the GIS representation of the study area, summing them, and multiplying by 1.4 million/50,000. This yields a slightly higher estimate of total population size than the simpler method of multiplying the average mallard density (individuals/ha) at BBS route-stops by the total area of the Central Valley, which gives an estimate of 47,373. The former estimate is preferred as it corrects for any bias in the sample of BBS route-stops with respect to the distribution of land use types in the study area. An interesting side note: application of an empirical model relating mallard breeding pairs to pond area in the Prairie Pothole region of the U.S. (Cowardin et al. 1995) yields an estimate of 50,276 for the Central Valley.