Protecting Watershed Ecosystems Through Targeted Local Land Use Policies

Christian Langpap, Dept. of Economics, Tulane University
Ivan Hascic, JunJie Wu, Dept. of Agricultural & Resource Economics, Oregon State University

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Abstract

Land-use change is arguably the most pervasive socioeconomic force driving the change and degradation of watershed ecosystems. This paper combines an econometric model of land use choice with three models of watershed health indicators (conventional water pollution, toxic water pollution, and the number of aquatic species at risk) to examine the effects of land use policies on watershed ecosystems through their effect on land use choice. The analysis is conducted using parcel-level data from four western states in the United States (California, Oregon, Washington, and Idaho). Our results suggest that incentive-based local land use policies, such as development impact fees and preferential property taxation, are most effective in improving water quality and decreasing the number of species at risk if targeted according to the current land use mix in the watersheds, while policies that attempt to change the returns to agricultural and forest land, such as reforestation payments or agricultural subsidies, are ineffective in any watershed.

Key words: Land use, targeted policy, water pollution, watershed ecosystems.

Introduction

Land-use change is arguably the most pervasive socioeconomic force driving the change and degradation of watershed ecosystems (Dale et al. 2000). The transition from relatively undisturbed natural habitat to more intensively managed land uses, such as agriculture or urban development, can have widespread implications for watershed ecosystems. For example, agricultural runoff is a major source of water pollution in both inland and coastal waters, and urban development is a leading cause of habitat destruction and species extinction around the world (Wilcove et al. 1998).

Land use decisions affecting watershed ecosystems depend in part on policy and regulations. Local land use regulations, such as zoning ordinances, development guidelines, and property acquisition programs, can directly affect land use and land use patterns. Other policies, such as development impact fees and conservation payments, may indirectly affect land use through their impact on the net returns to various land uses. *Ex ante* analysis of the likely environmental efficacy of these policies, or *ex post* assessment of their impacts, require combining a large scale economic model capable of predicting land use changes with the capacity to estimate the ecological effects of these alternative land uses at the regional scale. In addition, it is important to employ micro-level data both to achieve consistency with the underlying economic theory on which land use choice models are based and to capture accurately the significant spatial variability in economic and environmental variables and complex ecological processes (Just and Antle 1990; Antle and Capalbo 2001).

There is a large body of literature that examines land use policy at both the local scale (McMillen and McDonald 1993; Pfeffer and Lapping 1994; Kline and Alig 1999; Irwin and Bockstael 2002; 2004) and the national or regional levels (Stavins and Jaffe 1990; Plantinga et al. 2002; Schatzki 2003). These studies, however, do not examine how land use changes induced by policies affect watershed health.

There are also studies that examine the land use-water quality linkage at regional or national scales. Frissell (1993) finds that cumulative damage to aquatic habitats caused by logging, grazing, urbanization, and other land uses plays a major role in species diversity losses. Harding et al. (1998) investigate the influence of past land use on the present-day diversity of stream invertebrates and fish in North Carolina and find that past land-use activity is the best predictor of present-day aquatic diversity. Malmqvist and Rundle (2002) review the long-term trends affecting water quality and find that stream biodiversity has been significantly influenced by past land use. Dole and Niemi (2004) and Van Sickle et al. (2004) model the effect of land use decisions on water quality and stream flows in the Willamette River Basin. Hascic and Wu (2006) conduct a nationwide watershed-level analysis and find that the levels of conventional and toxic water pollution, as well as the number of endangered aquatic species, are affected by the amounts of land allocated to intensive agriculture, urban development, transportation, and mining.

A number of studies have examined the effect of conservation and/or land use policies on land use and the subsequent impact on water quality (see Wu et al. 2004 for a review of these studies). Most of these studies are undertaken at the farm or watershed level. Only a few have provided a systematic analysis of micro-level decisions and landscape changes and the resulting impacts on water quality (Wu and Segerson 1995;

Antle and Capalbo 2001; Wu et al. 2004). None of these studies have examined the effects of targeting land use policies, or used multiple measures of watershed ecosystem health.

In this article we draw from these strands of literature to examine the impact of targeted land use policies on watershed ecosystems through their effect on land use decisions. Specifically, we start by estimating an econometric model to analyze the determinants of land use choices in four western states (California, Idaho, Oregon, and Washington), including the returns to different land uses and local land use regulations. We then estimate econometric models for three measures of watershed health: conventional water pollution, toxic water pollution, and the number of aquatic species at risk. Finally, we combine the results of these econometric models to assess the effect of different land use policies on watershed ecosystems through predicted changes in land use.

Our results suggest that policies which lower urban returns, such as development impact fees, and incentive-based local policies, such as preferential property taxation, are most effective in improving water quality and decreasing the number of species at risk if targeted according to the current land use mix in the watersheds. Specifically, we find that these policies should be targeted to watersheds with high percentages of urban land or low percentages of agricultural land. On the other hand, policies that attempt to change the returns to agricultural and forest land, such as reforestation payments or agricultural subsidies, are ineffective in any watershed because these land uses are insensitive to changes in their relative returns.

The rest of this paper is organized as follows. First, the econometric models for predicting land use choice and watershed indicators are described. The data and estimation results are then discussed. Next, these econometric models are combined to simulate the effect of land use polices on watershed ecosystems, and the results of these simulations are discussed. Finally, the main results and policy implications are discussed.

The Land Use Model

The theoretical framework for the land use choice model is based on the land use decision for an individual parcel (Capozza and Li 1994; Bell and Irwin 2002). We assume that a landowner chooses a land use on his parcel on the basis of the current net returns to the various land uses, his expectation regarding the future growth of those returns, and the uncertainty surrounding returns (Capozza and Helsley 1990; Capozza and Li 1994; Schatzki 2003), given the existing local land use regulations. Let i index parcels, $k = a$ (agriculture), r (rangeland), f (forest), u (urban), o (other) index land uses, and t = 1982, 1987, 1992, 1997 index years (see discussion of data). Additionally, let π_{ikt} be the annual net return to land use k on parcel i in year t, let π_{ikt}^{G} be the expected growth of returns to land use k on parcel i in year t and π_{ikt}^{VG} its variance, and let Z_{it} be a vector of local land use regulations. Let $V_{ikt}\left(\pi_{ikt}, \pi_{ikt}^{G}, \pi_{ikt}^{VG} \mid Z_{it}\right)$ be the present value of expected net returns to land use k on parcel i in year t, conditional on local land use regulations. Then land use k is chosen on parcel i in year t if $V_{ikt} \geq V_{ilt}$ for all other $l \neq k$, i.e. if it yields the highest value.

To move to a statistical model that can be estimated with available data we must make some transformations. Data on net returns to alternative land uses do not exist at the parcel level. However, data are available to estimate the average net returns to alternative land uses at the county level. In addition, we have information about land characteristics at individual parcels and the attributes of the county (e.g. location characteristics) in which each parcel is located. With these data, we decompose V_{ikt} into two parts: V_{ikt} = $\boldsymbol{\beta}_k'\boldsymbol{v}_{ikt} + \varepsilon_{ikt}$, where $\boldsymbol{v}_{ikt}^{'} = \left(\pi_{ikt}, \pi_{ikt}^{G}, \pi_{ikt}^{VG}, Z_{it}\right)$, $\boldsymbol{\beta}_k$ is a vector of parameters to be estimated, and ε_{ikt} is a random error term. The non-random component ($\boldsymbol{\beta}_k'\boldsymbol{v}_{ikt}$) is specified as a linear function of a) the net returns to land use *k* in year *t* in the county where parcel *i* is located, b) the expected growth of net returns to land use *k* in year *t* in the county where parcel *i* is located, c) the variance of the growth of returns to land use *k* in year *t* in the county where parcel *i* is located, d) land characteristics of parcel *i*, and e) land use regulations and other characteristics of the county in which parcel *i* is located. Given this decomposition, the choice rule can be rewritten in probabilistic terms. The probability that land use *k* is chosen on parcel *i* in year *t* is

$$\text{Prob}_{ikt} = \text{Prob}(\boldsymbol{\beta}_k'\boldsymbol{v}_{ikt} + \varepsilon_{ikt} \geq \boldsymbol{\beta}_l'\boldsymbol{v}_{ilt} + \varepsilon_{ilt}, \ \forall l \neq k) \tag{1}$$

If the random errors have a Gumbel distribution, (1) can be specified as a multinomial logit model (see, e.g., Maddala 1993):

$$\text{Prob}_{ikt} = \frac{e^{\beta_k^{'} v_{ikt}}}{\sum_{l=u,a,n,f,o} e^{\beta_l^{'} v_{ilt}}}, \ k = a, r, f, u, o \tag{2}$$

The multinomial logit model has been widely used in economic analysis. In agriculture, it has been used to model farmers' land allocation decisions (Wu et al. 2004), the choice of irrigation technologies (Caswell and Zilberman 1985), and the choice of alternative crop management practices (Wu and Babcock 1998).

The Watershed Health Indicator Models

Three indicators constructed by the U.S. Environmental Protection Agency are used to describe the health of watershed ecosystems across the four states in our study area. The *conventional ambient water quality* indicator measures the number of surface water samples in a watershed with concentrations of one or more of four conventional water quality measures (phosphorus, ammonia, dissolved oxygen, pH) exceeding the national reference levels. The indicator is constructed based on water quality monitoring data collected between 1990 and 1998. The concentrations of phosphorus and ammonia and the level of dissolved oxygen and pH are important indicators of water quality. High concentrations of phosphorous and ammonia are associated with excessive eutrophication, leading to algal blooms, water turbidity, hypoxic or anoxic conditions, and changes in aquatic biodiversity (Smith 1998). Acidification can disrupt the nitrogen cycle in freshwater ecosystems (Vitousek et al. 1997) and may lead to decreased diversity of animal and plant species (Schindler 1994). Excessive eutrophication and water pollution have been linked to agricultural land and chemical uses, urban runoff, and topographic and hydrological characteristics (Wu and Babcock 1999).

The *toxic ambient water quality* indicator measures the number of surface water samples in a watershed with concentrations of one or more of four toxic pollutants (copper, nickel, zinc, chromium) exceeding the national chronic levels. This indicator is constructed based on water quality monitoring data collected between 1990 and 1998. Contamination of water bodies by heavy metals is a major concern due to their sedimentation, persistence, bioaccumulation potential, and their lethal and sublethal effects. Elevated concentrations of toxic substances affect aquatic wildlife in a number of ways, including changes in morphology, physiology, body biochemistry, behavior, and reproduction (Handy and Eddy 1990). The major anthropogenic sources of metallic pollution of water bodies include urban, industrial, and commercial land use, as well as intensive agriculture and mining (Malmqvist and Rundle 2002).

The *species-at-risk* indicator measures the number of aquatic and wetland species (plants and animals) at risk of extinction in a given watershed in 1996. Several studies have investigated the relationship between the health and abundance of aquatic organisms and their potential as a bioindicator. Amphibians have long been regarded as important indicators of environmental health and aquatic biodiversity due to their extreme susceptibility to perturbations in the environment (Blaustein and Johnson 2003). Fish are considered useful indicators of biological integrity and ecosystem health since they respond predictably to changes in both abiotic factors, such as habitat and water quality, and biotic factors, such as human exploitation and species additions (Davis and Simon 1995). Habitat alterations such as wetland drainage, wetland fragmentation, river damming and channelization, and other types of hydrologic modification have been identified as a major factor determining species composition and population abundance in aquatic ecosystems (Faurie et al. 2001).

To specify econometric models for our indicators of watershed health we note that the number of samples violating national water quality standards and the number of endangered species in a watershed represent event counts, which are usually estimated using Poisson and negative binomial models (Maddala 1983, Cameron and Trivedi 1998). Formally, an event count is defined as a realization of a nonnegative integer-valued random variable y, and the Poisson model is derived by assuming that y is

Poisson-distributed with the conditional density of y given by $f(y|\boldsymbol{x}) = e^{-\theta}\theta^{y}/y!$, where $\theta = E[y|\boldsymbol{x}]$. The log of the mean θ is assumed to be a linear function of a vector of independent regressors $\boldsymbol{x}$: $\ln \theta = \boldsymbol{x\beta}$, where $\boldsymbol{\beta}$ is a parameter vector. This specification ensures nonnegativity of θ (Cameron and Trivedi 1998).

There are two potential problems with the Poisson regression model. First, it assumes that the sample size is constant, but sample sizes often change in cross-sectional analyses. To address this problem, Maddala (1983) suggests an alternative specification: let N be the total sample corresponding to y so that the rate of occurrence is y/N, and re-parameterize the Poisson model as $\ln \theta = \ln N + \boldsymbol{x'\beta}$. In this study, the sample size is known for conventional and toxic water pollution measures, but unknown for the species-at-risk indicator. Second, the Poisson specification imposes a restriction by assuming that $E[y|\boldsymbol{x}] = V[y|\boldsymbol{x}] = \theta$ (the equidispersion property). The standard way to account for overdispersion is the NB2 model suggested by Cameron and Trivedi (1998), who derive this negative binomial model from a Poisson-gamma mixture distribution[1]. Based on these studies, the econometric models for the watershed health indicators are specified as follows:

$$\ln(CONVWQ_i) = \ln N_i^c + \beta_0 + \boldsymbol{\beta}_1^{'}\boldsymbol{l}_i^c + \boldsymbol{\beta}_2^{'}\boldsymbol{p}_i^c + \varepsilon_i^c \tag{3}$$

$$\ln(TOXICWQ_i) = \ln N_i^t + \gamma_0 + \boldsymbol{\gamma}_1^{'}\boldsymbol{l}_i^t + \boldsymbol{\gamma}_2^{'}\boldsymbol{p}_i^t + \varepsilon_i^t \tag{4}$$

$$\ln(SPERISK_i) = \delta_0 + \boldsymbol{\delta}_1^{'}\boldsymbol{l}_i^s + \boldsymbol{\delta}_2^{'}\boldsymbol{p}_i^s + \boldsymbol{\delta}_3^{'}\boldsymbol{d}_i^s + \varepsilon_i^s \tag{5}$$

[1] In addition to y being conditionally Poisson-distributed, the parameter θ is assumed to be the product of a deterministic term and a random term: $\theta = e^{x'\beta+\varepsilon} = e^{x'\beta}e^{\varepsilon} = \mu\nu$. Cameron and Trivedi show that by assuming a gamma distribution for ν (mean 1, variance α), the marginal distribution of y is the negative binomial with the first two moments $E[y|\mu,\alpha] = \mu$ and $V[y|\mu,\alpha] = \mu + \alpha\,\mu^2$.

where i indexes watersheds; N_i^c and N_i^t are the total number of samples taken to measure conventional and toxic water quality, respectively; $\boldsymbol{l}_i^c$, $\boldsymbol{l}_i^t$, and $\boldsymbol{l}_i^s$ are vectors of land use variables; $\boldsymbol{p}_i^c$ and $\boldsymbol{p}_i^t$ are vectors of physical characteristics measuring the vulnerability of individual watersheds to conventional and toxic water pollution; $\boldsymbol{p}_i^s$ is a vector of watershed characteristics that affect aquatic species; $\boldsymbol{d}_i^s$ is a vector of spatial dummies for ecosystem divisions defined by the USDA Forest Service; and ε_i^c, ε_i^t, and ε_i^s are error terms, with $\exp(\varepsilon_i)$ following the gamma distribution[2]. We estimate these models using maximum likelihood methods.

Data

Data for Estimating the Land Use Model

The data on land use were obtained from the 1982, 1987, 1992, and 1997 Natural Resources Inventories (NRI)[3] (US Department of Agriculture 2001). The NRI uses eleven broad land use categories, which we collapse into five categories to estimate the land use model: agricultural land (crop and pasture land), range land, forest, urban and built up land, and other. The total number of parcels in our sample is approximately 128,680. By using the NRI data, we can determine land use at each NRI site in each of the four years. Additionally, NRI assigns a weight (called xfactor) to each site that

[2] Since the total number of aquatic and wetland species in each watershed is unknown, ln N, which appears in the first two equations, is not present in the third equation. However, the differences in species diversity are partially accounted for by the spatial dummies representing the varying ecological conditions across the region.
[3] The NRI is conducted every five years to collect land use data at about 800,000 randomly selected sites across the U.S.

reflects the acreage it represents. Thus, we can use the xfactor for all sites in a given land use to estimate the total acreage in that land use.

The NRI data include the Land Capability Class (LCC), a land quality index, for each plot. It ranges from 1 (best quality) to 8 (worst quality). We incorporate this information into our model by defining a high-quality-land dummy, which is equal to 1 if a site is in LCC categories 1 or 2 and to 0 otherwise. To control for climate differences that may affect the productivity of different land uses, we include the NRI's climatic factor variable (WCFACT). To include a proxy for urban development pressure, the distances from the geographical center of each county to the closest large metropolitan center (Portland, Seattle, Los Angeles, San Francisco, and Boise) and to the closest major city (population $\geq$ 100,000) were estimated[4]. These distances were used to calculate a population-weighted distance index as in Hardie et al. (2001)[5]. Additionally, the distances from the geographical center of each county to the geographical center of wilderness and urban-type national parks were included as measures of distance to open space amenities, as it has been well established in hedonic studies that housing prices and thus returns from development are inversely related to distance to amenity locations (Tyrväinen and Miettinen 2002). The data on the geographical center of each county and of national parks were obtained from the U.S. Census Bureau (2000), and the distances were calculated using the website www.indo.com/distance.

[4] Because the locations of NRI sites are not identified in the dataset, we cannot include the distance from each NRI site to the closest major city as an independent variable.

[5] The population-weighted distance index was calculated using the formula $D = Pop_m/d_m^2 + Pop_c/d_c^2$, where Pop_m and Pop_c represent the population of the closest metropolitan center and major city, respectively, and d_m and d_c are the distances from the county center to the closest metropolitan center and major city.

The data on returns to agricultural and range land comes from the Bureau of Economic Analysis (U.S. Department of Commerce 2003), which reports farm profits at the county level on a yearly basis. To measure returns per acre from agriculture we use cash income for crops (in thousands of US $) as a measure of revenue and production expenditures as a measure of costs. Some expenditures can be specifically allocated to crop production (purchase of seeds and purchase of fertilizer and lime). Other expenditure categories (petroleum products, labor, and other expenditures) cannot. To allocate these expenditures, we calculate the share of total revenues that crop revenue represents and we allocate the same share of these expenditures to agricultural production costs. The resulting measure of county-level returns to agriculture is then adjusted by the Consumer Price Index. Finally, to obtain returns per acre, we use the NRI's xfactor to estimate the total acreage allocated to agriculture in each of the four years and divide total adjusted returns in each county by this amount. Returns per acre from rangeland are estimated in a similar way. We use cash income from livestock and products as a measure of revenue, and production expenses as a measure of costs. Expenses that can be directly allocated to rangeland are feed and livestock purchases; the remaining expenses are allocated in the same way as was done for agricultural returns. Two counties (out of a total of 177) were dropped from the sample due to missing returns data.

Returns to urban development and forestry were obtained from Lubowski et al. (2003). They estimate net urban returns per acre as “the median value of a recently developed parcel, less the value of structures, annualized at a 5 percent interest rate”. They estimate annual net returns from forestry by calculating the net present value of a weighted average of saw timber revenues from various forest types based on acreage, yields, prices, and costs, annualized at a five percent interest rate. We adjust their data by the CPI and scale to thousands of dollars/acre. To calculate the expected growth of returns to each land use we assume that the expected future growth is equal to the average annual growth over the past five years. That is, let $G_{ikt|t-1} = (\pi_{ikt} - \pi_{ikt-1}) / \pi_{ikt-1}$ be the growth rate of returns to land use *k* between *t* -1 and *t*. Then the expected growth rate at time *t* is

$$\pi_{ikt}^{G} = \frac{1}{5}\sum_{j=1}^{5} G_{ikt-j|t-(j+1)} \tag{6}$$

The variance of the growth rate of returns to each land use is estimated by[6]

$$\pi_{ikt}^{VG} = \frac{1}{5}\sum_{j=1}^{5} (G_{ikt-j|t-(j+1)} - \pi_{ikt}^{G})^2 \tag{7}$$

Information on county-level land use regulations was obtained from a land use policy survey conducted in 1999. The survey was conducted using the Dillman (1978) method for mail surveys, and the overall response rate was 69% (see Cho et al. 2003 for further details). Twelve counties with missing regulation data were dropped from the sample. The survey asked county land use planners whether different land use regulations were in place by 1982, 1987, 1992, and after 1993, and to judge the effectiveness of the regulation on a scale ranging from 1 to 5, where 1 corresponds to “Not Effective” and 5 to “Effective”. We divided the regulations into four categories (Incentive-Based Policies, Property Acquisition Policies, Development Guidelines, and Zoning Ordinances), and constructed a regulation index for each category in each period[7]. The index measures the fraction of the maximum possible effectiveness score achieved in each regulation

[6] For the returns to urban development and forestry in 1982, (4) and (5) are calculated averaging over 4 rather than 5 years, because we only have data starting in 1978.

[7] The following are sample policies included in each category. Development Guidelines: urban growth boundaries and housing caps; Incentive-Based Policies: preferential property taxes and density bonuses; Property Acquisitions: purchase and transfer of development rights; Zoning Ordinances: agricultural and conservation zoning.

category. For example, there are five regulations included in the Property Acquisitions category, so the maximum attainable effectiveness score for any given county in any given period is twenty five. The effectiveness scores provided by the Contra Costa county (California) land use planner for 1982 add up to 12, so the corresponding index is 0.48; for 1987 they add up to 14, and thus the index increases to 0.56.

In addition to the county-level land use regulations, two state-level regulations were included in our land use model: state land-use planning programs, which require adoption of a plan that meets state guidelines, and mandatory review of projects involving farmland conversion. These regulations were included in the empirical model as dummy variables[8].

Data for Estimating the Watershed Health Models

The watershed health indicators were obtained from the U.S. Environmental Protection Agency (EPA)'s Index of Watershed Indicators (IWI) (EPA 2004), which contains data characterizing the condition and vulnerability of aquatic systems in watersheds across the United States[9]. Land use data for the three watershed health models are drawn from the NRI. We use the following land use categories: agricultural land (including cropland, pasture land, and rangeland), forest land, urban land, rural transportation land, mining land, and other land. These land use variables are constructed as percent of total land area of the hydrologic unit and are averaged over the four NRI years (1982, 1987, 1992, 1997). We also include the total acreage (in thousands of acres)

[8] We note that Washington is the only state that has a mandatory review requirement, so this variable may capture other factors that are unique to that state.
[9] Watersheds are defined by the 8-digit hydrologic units from the nationally consistent set of watersheds in the Hydrologic Unit Classification System developed by the US Geological Service.

of irrigated land, permanent open water, and wetlands. Additionally, we control for some physical characteristics of each watershed: the average wind and water erosion rates (tons/acre/year) in the watershed, and the soil permeability index. The average wind and water erosion rates were obtained from NRI soil erosion estimates, and the soil permeability index from the IWI[10].

Finally, we include a set of spatial dummies in the species-at-risk model to capture the spatial variability in species richness across the study area. Our spatial dummy variables are derived from the USDA Forest Service's Ecosystem Divisions, which are defined as areas that share common climatic, precipitation, and temperature characteristics[11]. Summary statistics for the variables described in this section can be found in Table 1.

Results

Land Use Model

We use pooled data for 1982, 1987, 1992, and 1997 to estimate the multinomial logit land use model in (2), with "other" as the base land use. We leave out the variances of the growth of returns for agriculture and range land because they are highly correlated with the expected growth of returns[12]. Because the coefficients in a multinomial logit model

[10] The EPA constructed the index based on the State Soil Geographic (STATSGO) database of the USDA's Soil Conservation Service. The soil permeability index reflects the property of the overlying soil and is one of the controlling factors of the transport rate of contaminants through soil. The degree of soil permeability can affect the risk of contamination of ground water resources, and consequently quality of surface waters where ground water feeds rivers and lakes (EPA 2004).
[11] For details see http://www.essc.psu.edu/soil_info/soil_eco/.
[12] Including the variances would not change the main results of the land use model.

are difficult to interpret, here we only report the marginal effects, which can be found in Table 2[13]. They are calculated using

$$\partial \text{Prob}_{ikt}/\partial v_{ikt}^{j} = \text{Prob}_{ikt} \cdot [\beta_k^j - \Sigma_k (\text{Prob}_{ikt} \cdot \beta_k^j)] \tag{8}$$

where v_{ikt}^{j} and β_k^j are the j[th] elements of vectors $\boldsymbol{v}_{ikt}$ and $\boldsymbol{\beta}_k$, respectively.

The own-return marginal effects are positive and significant, and the cross-return marginal effects are negative and significant, indicating that higher returns to a land use increase the probability that it is chosen and decrease the probability that other land uses are chosen. The magnitude of these marginal effects is generally small. The same can be said about expected growth of returns from agriculture and forest. The marginal effects for expected growth of returns to urban land and rangeland, however, are negative in their own equations, suggesting that higher expected future returns decrease the present probability of choosing urban and range land uses, respectively, and increase the probabilities of choosing the other land uses. Capozza and Li (1994) show that an increase in the growth expectation for the net return to developed land does not necessarily speed up development decisions. This counter-intuitive behavior arises because when the growth expectation increases, the option value of conversion (i.e., the value of delaying the project) also increases, so that it may be worthwhile to postpone the project even though the intrinsic or net present value of the project has increased. More intuitively, urban development can only be chosen once in a given parcel, since development is irreversible. Hence, if returns to development are expected to be higher in the future, landowners may postpone development, which makes it less likely that

[13] The coefficient estimates are available from the authors upon request.

urban land use is chosen in the current period[14]. However, we have no equivalent explanation for the marginal effect of expected growth of returns to rangeland. The variances of growth of returns to forest and urban land uses have negative own- and positive cross-marginal effects, suggesting that higher variability in the growth of returns decreases the probability of choosing a land use and increases the probability of choosing others.

The marginal effects of the land use regulation indices yield mixed results. Incentive-based policies, such as preferential property taxation, that encourage farmland and forestland owners to keep their lands in those uses have positive and statistically significant effect on the probability of choosing farmland and forest land uses, but have a negative and statistically significant effect on the probability of choosing urban land use. The same can be said about property acquisition programs such as purchase or transfer of development rights. On the other hand, zoning and development guidelines have positive effects on the probability of choosing urban land, although only the marginal effects of development guidelines are statistically significant. The insignificant effect of zoning on land development may indicate that zoning affects the location of development, but not the total amount or overall probability of development. The positive effect of development guidelines is consistent with the notion that land use planning that preserves forests or open space may spur additional development. For example, Wu and Plantinga (2003) show that open space designation may not only cause leapfrog development, but also more development. Riddell (2001) reports evidence that open space programs in

[14] Comparative statics in the Capozza-Helsley (1990) model for the effect of expected growth of returns on the reservation rent, and thus on the likelihood of development, yield an undeterminate sign. Hence, the effect is not necessarily positive *a priori*.

Boulder, Colorado increased growth in residential development. Irwin and Bockstael (2004) use data from Maryland to simulate the effects of a cluster development policy intended to preserve open space and show that it is possible that, by creating a positive amenity associated with the preserved open space, the policy may attract development and exacerbate sprawl.

One possible concern with the land use model is that the regulation index variables may be endogenous. We do not believe this is the case because we model land use decisions of individual owners, who are likely to take land use regulations as given when making their land use decisions. In addition, the land use indices for a given year are based on regulations that were in place by that time, and hence land use choices in that year could not have affected the passing of the regulation. Nevertheless, we also estimated a land use model with lagged regulation variables, which are exogenous, and the qualitative results are the same as those presented here.

Watershed Health

We estimate models (3) - (5) to evaluate the effect of land use on water quality and aquatic species. Since the land uses included in these models completely describe the landscape, we avoid perfect multicollinearity by excluding forest land and using it as the reference category. Land-use variables are averages over the four NRI years (1982, 1987, 1992, 1997). The results are presented in Table 3. The goodness-of-fit measures indicate that the NB2 models fit the data much better than the Poisson model for each of the three equations. The Pearson/DF and Deviance/DF measures also indicate that the Poisson distribution assumption is inappropriate[15]. Hence, we only report results for the NB2 models.

The three columns of results in Table 3 show the coefficient estimates for the three watershed models: conventional water pollution, toxic water pollution, and species-at-risk. The coefficient for urban land is positive and statistically significant in all three models. The coefficient for agricultural land is positive in all three models as well, but it is significant only in the conventional water quality model. These results suggest that, in our study area, converting forests to developed land increases both conventional and toxic water pollution and the number of aquatic species at risk in the watershed. On the other hand, converting forest land to agriculture increases conventional water pollution, but seems to not have a significant effect on toxic water pollution or the number of species at risk.

In addition to urban development, the percent of land used for transportation and mining also have a positive and statistically significant effect on toxic water pollution. These two land uses also increase the number of species at risk in a watershed, although only transportation is statistically significant. Other significant variables in the species-at-risk model include the area of wetland and the total land area. The coefficients on both variables are positive, indicating watersheds with more species are more likely to have more endangered species.

[15] The Deviance and Pearson statistics divided by degrees of freedom with values close to 1 indicate a good fit of the regression model. Values greater (smaller) than 1 indicate over (under) dispersion, i.e. the true variance is greater (smaller) than the mean. Evidence of over (under) dispersion indicates inadequate fit (Cameron and Trivedi 1998).

Simulating the Effects of Land Use Policies on Watershed Health Indicators

In this section we link the land use choice and watershed health models to examine the effect of land use policies on conventional water pollution, toxic water pollution, and the number of species at risk. We start by using sample data and the estimated watershed models (3) - (5) to generate predicted baseline values for the three indicators of watershed health. Once the baseline is established, we can evaluate how land use policies affect watershed health. Specifically, we evaluate the effects of policies that discourage development (e.g. development impact fees) or encourage preservation or reforestation of agricultural land (e.g. conservation and reforestation payments), and of local land use regulations (incentive-based policies and property acquisitions).

To simulate the effects of policies that discourage development, we assume that they reduce the net returns from development by 5%, 25%, and 50% and use equation (2) to predict the resulting land use probabilities at each NRI site in 1997. We use these probabilities and the NRI's x-factor to predict land use acreages, and calculate the fraction of each watershed in urban, agricultural, and forest land. The results are presented in Table 4. Then we feed the predicted land use acreages into the watershed models, along with sample means for the remaining variables, to calculate watershed health indicators. Finally, we calculate percentage changes in these measures relative to those in the base scenario. Similarly, to examine the effects of policies that preserve agricultural land, we increase the returns to agriculture by 5%, 25%, and 50%. To simulate the effect of reforestation policies, we increase the returns to forestry on land that is used for agriculture[16]. Finally, we evaluate the effects of local land use regulations by increasing the value of the land use regulation indices by 5%, 25%, and 50%. Additionally, we examine the effect of increasing the value of each index to the highest value found in the data. Because the results of the land use model show that zoning and development guidelines are not effective in controlling development, we only use the Incentive-Based Policies index and the Property Acquisition index.

Given that different policies are designed to impact different land uses, we examine whether the effectiveness of these policies varies with the predominant land use in a watershed. Specifically, we separately evaluate the effects of each land use policy on the watersheds that have the highest and lowest percentages of developed land, agricultural land, and forest land. This will allow us to identify not only the most effective policies for each measure of watershed health, but also whether these policies would be more effective if targeted to specific types of watershed.

The simulation results appear in Tables 5-7. The tables show mean percentage changes relative to baseline values for conventional water pollution, toxic water pollution, and the number of species at risk, respectively. Negative changes represent reductions in the number of water samples with concentrations of conventional or toxic water quality measures exceeding reference levels, or reductions in the number of species at risk. Hence, we interpret them as improvements in water quality and watershed health. Positive changes reflect increases in the number of water samples with concentrations of conventional or toxic water quality measures above reference levels, or increases in the

[16] We also simulated the effects of forest preservation policies by increasing the returns to forestry on all land uses. The qualitative effects are equivalent to those of reforestation payments, so we do not include them here because of space considerations.

number of species at risk. Hence, we interpret them as deteriorations in water quality and watershed health. We report results for the 5% of watersheds with the highest and lowest proportions of land in urban, agricultural, and forest land uses. In the following discussion, we refer to these as the most/least urban, agricultural, and forested watersheds.

The results in Tables 5 – 7 suggest that policies which decrease the returns to urban land have the largest positive effects on water quality and species at risk. These policies are most effective in the most urban or the least agricultural watersheds, and have a smaller but still significant effect in the most forested watersheds. When urban returns fall by 50%, the conventional water quality indicator decreases by 14% relative to the baseline in the most urban watersheds and 10% in the least agricultural watersheds. The toxic water quality indicator decreases by 28% in the most urban watersheds, 12% in the least agricultural watersheds, and 6% in the most forested watersheds. The number of species at risk decreases by almost 8% in the most urban watersheds and 6% in the least agricultural watersheds. The changes in the remaining types of watersheds are considerably smaller, for the most part not exceeding 1%. To understand these results, note from Table 4 that lowering the returns to urban land leads to small increases in agricultural and forest land, and a relatively larger decrease in urban land. The latter effect dominates in the most urban watersheds. Many of the least agricultural watersheds have relatively high proportions of either urban or forest land, so decreases in urban land or further increases in forest land lead to improvements in the watershed health indicators.

Policies that aim to preserve agricultural land by increasing the returns to agriculture have negligible effects on water quality and species at risk in all types of watersheds. The effects on conventional water quality (if any) are negative, but the increase in the indicator is always less than 0.6%. The effects on toxic water quality, on the other hand, are positive, but the decreases in the indicator do not exceed 0.5%. There are essentially no effects on the number of species at risk.

Policies that encourage reforestation of agricultural land have positive effects on water quality and species at risk, but the changes are also small. The conventional and toxic water quality indicators and the number of species at risk decrease relative to the baseline when returns to forest land increase, but the changes are mostly smaller than 1%.

Table 4 suggests that these two policies are ineffective because they have small effects on land use, yielding only minor changes in the mix of land use in a watershed. Policies that increase the returns to agricultural land have essentially no effect on land use, and policies that increase the return to forest land lead to small decreases in agricultural and urban land, and a small increase in forest land. These outcomes are consistent with previous results that land use allocations are relatively unresponsive to changes in returns to alternative land uses, particularly for transitions between agriculture and forestry (Stavins and Jaffe 1990; Plantinga 1996; Wu et al. 2004).

The effects of incentive based policies depend on both the specific measure of watershed health and the type of watershed. The conventional water quality indicator decreases by almost 3% in the most urban watersheds when the index is increased to 1.0, its highest value, but the effects are minor for smaller increases in the index. The toxic water quality indicator decreases by 11% in the most urban watersheds and by almost 4%

in the most forested and least agricultural watersheds. The number of species at risk decreases by 2% in the most urban watersheds. Hence, these policies can have positive effects on watershed health in the most urban watersheds, as long as they are implemented at their highest level of effectiveness (i.e. corresponding to an index value of 1.0). As shown in Table 4, the incentive-based land use policies yield a small increase in agricultural land, a decrease in urban land, and a relatively larger increase in the fraction of forests in a watershed. The combination of a decrease in urban land a relatively large increase in forest dominate in the most urban watersheds, yielding improvements in the watershed health indicators. These policies, however, can also have relatively large negative effects on conventional water quality: the indicator increases by almost 5% in the least urban and most agricultural watersheds, and by almost 7% in the least forested watersheds. In these types of watersheds the dominant land use is agriculture, with small fractions of urban and forest land. Hence, the effect of an increase in agricultural land dominates, yielding deterioration in conventional water quality. This same pattern of effects occurs for the species at risk indicator, but the changes are smaller.

Property acquisition policies have small effects on the watershed indicators: even when the index is increased to 0.6 (its maximum sample value), the decreases in the conventional and toxic water quality indicators are less than 2%, and they are less than 0.5% for the species at risk indicator. A slightly larger negative effect on conventional water quality results if these policies are implemented on the most forested watersheds, where the indicator increases by almost 3%. The remaining effects are minor. Table 4 suggests that this may be because these policies lead to small changes in agricultural and urban land, but a relatively large decline in forest land.

In sum, the results presented in Tables 5-7 suggest that policies which lower returns to developed land, such as development impact fees, may be more effective at improving water quality and decreasing the number of aquatic species at risk than policies designed to change the returns to agricultural and forest land. Land use choices, in particular for agriculture and forestry, are insensitive to changes in their relative returns, and hence it may take large changes in returns to have a significant impact on land allocation. In contrast, the likelihood of development is relatively more responsive to changes in the returns to urban land. Incentive-based local land use policies, such as preferential property taxes, can effectively improve watershed health as well. In particular, our results suggest that significant positive effects on water quality could be achieved if more counties adopted incentive-based policies similar to the most effective ones in our sample. Property Acquisition policies, such as purchasing or transferring development rights, could also play a small role in improving conventional water quality.

The results also highlight the importance of targeting these policies according to whether watersheds have predominantly urban, agricultural, or forest land uses. By targeting the implementation of these policies to selected types of watersheds, land use planners can avoid unintended negative effects that could counter the intended positive effects on watershed health. For instance, incentive based policies have positive effects on conventional water quality in the most urban watersheds, but relatively large negative effects in the least urban or forested watersheds, or the most agricultural watersheds. Thus, if this type of policy is applied uniformly to all watersheds, improvements in some watersheds may be offset by declines in others, and the overall net effect could be negative. Additionally, implementation and enforcement of land use policies is costly, so targeting could help land use planners achieve watershed health goals more cost effectively by focusing implementation on watersheds where the policies have the largest effects. For instance, policies that lower the returns to urban land have positive effects in all types of watersheds, so applying this policy uniformly will yield positive net effects. However, this policy is not very effective in watersheds that have low proportions of urban land or high proportions of agricultural land. Hence, the same watershed health goals would likely be achieved at lower cost by not implementing the policy in these types of watersheds, and targeting only those watersheds where it is expected to have significant effects.

Our results indicate that policies which lower returns to development, such as a development fee, and incentive-based land use policies such as preferential property taxation should be targeted to watersheds with high percentages of urban or low percentages of agricultural land. To see why, note from Table 4 that these policies yield small increases in agricultural land, and relatively larger decreases in urban land. Furthermore, Table 3 indicates that converting forests to urban land and to agricultural land have comparable negative effects on the conventional water quality indicator, and conversions to urban land have larger negative effects on the toxic water quality and species at risk indicators. Hence, these policies will be more effective if targeted to watersheds with relatively less agricultural land and relatively more urban land than other types of watersheds. In these types of watersheds, the effect of the decrease in urban land dominates that of the increase in agricultural land, yielding improvements in the watershed health indicators. Thus, it makes sense to pick policies that affect land uses which are most detrimental to watershed health, and target those policies to watersheds were those land uses are predominant.

Summary and Conclusions

This paper develops an empirical framework to examine the effects of land use policies on three selected watershed health indicators in four western states of the United States. The framework consists of an econometric model of land use choice and three models of watershed health indicators (conventional water pollution, toxic water pollution, and the number of aquatic species at risk). The framework is then used to simulate the effects of policies that decrease the returns to developed land or increase the returns to agricultural and forest land, as well as of incentive-based local land use regulations and property acquisition programs.

Our results suggest that the efficacy and cost-effectiveness of the land use policies considered here could be enhanced by targeting them according to the land use mix of watersheds. By identifying the land uses that have the strongest negative effects on watershed ecosystems, policy makers can choose land use policies that have the largest effects on those land uses, and target them to watersheds where those land uses are predominant. This would allow policy makers to avoid unintended effects in watersheds with different land use mixes, where the outcome of the policies might be different. Additionally, targeting policies to watersheds where they are expected to be most effective may allow policy makers to achieve watershed health targets at lower cost by generating savings in implementation and enforcement costs.

For the watersheds considered here, we find that converting forests to developed land has much larger effects on toxic water pollution and the number of species at risk than converting forests to agriculture; however, the opposite is true in term of the effect on conventional water pollution. Policies that reduce the returns to developed land, such as development impact fees, and incentive-based local policies, such as preferential property taxation, yield small increases in the amount of agricultural land, but relatively large reductions in the proportion of urban land in highly developed watersheds. Thus, targeting these policies to watersheds with high percentages of developed land or low percentages of agricultural land is most effective in improving water quality and decreasing the number of species at risk. In contrast, policies that attempt to change the returns to agricultural and forest land, such as reforestation payments or agricultural subsidies are for the most part ineffective in any watershed because these land uses are insensitive to changes in their relative returns.

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Table 1: Summary Statistics

Variable	*Mean*	*Std. Dev.*
Land Use Model		
Ag. Returns/acre (thousands US $)	0.15	0.71
Range Returns/Acre (thousands US $)	-0.05	0.33
Forest Returns/Acre (thousands US $)	0.02	0.02
Urban Returns/Acre (thousands US $)	5.21	3.87
Development Guidelines	0.17	0.17
Incentive-Based Policies	0.24	0.24
Property Acquisition	0.11	0.15
Zoning Policies	0.26	0.25
Land Use Plan	0.76	0.43
Mandatory Review	0.18	0.38
Land Quality 1	0.06	0.23
Weighted Distance Index	51.78	121.13
Distance Wilderness Park (miles)	83.54	47.52
Distance Urban Park (miles)	245.28	157.41
Climatic Factor	6.43	19.85
Watershed Health Models		
Conventional Ambient Water Pollution		
- Total number of samples	2343.2	9040.37
- Number of samples in exceedance of the national reference levels	270.08	881.99
Toxic Ambient Water Pollution		
- Total number of samples	521.42	990.49
- Number of samples in exceedance of the national reference levels	109.16	357.75
Number of Species at Risk	4.00	3.16
Urban Land (% of watershed)[a]	3.74	9.45
Agricultural Land (% of watershed)[a]	26.52	24.80
Transportation Land (% of watershed)[a]	0.78	0.62
Mining Land (% of watershed)[a]	0.15	0.50
Other Land 1 (% of watershed)[a]	47.97	30.09
Other Land 2 (% of watershed)[a]	47.05	30.63
Irrigated Land (1,000 acres)[a]	53.35	200.68
Soil Loss due to Water Erosion (tons/acre/year)[a]	1.17	2.29
Soil Loss due to Wind Erosion (tons/acre/year)[a]	1.43	9.56
Index of Soil Permeability	3.83	0.98
Area of Water Bodies (1,000 acres)[a]	15.68	45.90
Area of Wetlands (1,000 acres)	14.67	37.34
Total Land Area (1,000 acres)[a]	804.17	660.84

[a] Four-year averages (1982, 1987, 1992, 1997).

Table 2: Estimates of Marginal Effects for the Multinomial Logit Model of Land Use Choice

Variable	*Agriculture*	*Range*	*Forest*	*Urban*	*Other*
Constant	-0.1154***	-0.1052***	-0.0414***	-0.0303***	0.2923***
Ag. Returns	0.0035***	-0.0006***	-0.0004***	-0.0001***	-0.0025***
Expected Growth of Ag. Returns[a]	0.7000***	-0.1000***	-0.0700***	-0.0200***	-0.5000***
Range Returns	-0.0137***	0.1092***	-0.0113***	-0.0040***	-0.0803***
Expected Growth of Range Returns[a]	0.0060*	-0.0500*	0.0050*	0.0020*	0.0300*
Forest Returns	-0.1371***	-0.1767***	1.1587***	-0.0396***	-0.8053***
Expected Growth of Forest Returns[a]	-0.0400*	-0.0500*	0.3000*	-0.0100*	-0.2000*
Variance Growth of Forest Returns	0.0007***	0.0009***	-0.0060***	0.0002***	0.0040***
Urban Returns	-0.0004***	-0.0005***	-0.0003***	0.0034***	-0.0220***
Expected Growth of Urban Returns	0.0037***	0.0048***	0.0031***	-0.0336***	0.0220***
Variance Growth of Urban Returns	0.0262***	0.0337***	0.0216***	-0.2350***	0.1535***
Incentive-Based Policies	0.0399***	-0.0053*	0.0659***	-0.0051***	-0.0955***
Property Acquisition	-0.0993***	0.0871***	-0.1275***	-0.0039**	0.1436***
Development Guidelines	0.1066***	0.0408***	-0.0837***	0.0193***	-0.0830***
Zoning Policies	-0.1005***	-0.0141***	0.0473***	0.0015	0.0658***
Land Use Plan	-0.0326***	-0.0101***	-0.0541***	-0.0259***	0.1227***
Mandatory Review	0.0282***	-0.0219***	0.0834***	0.0062***	-0.0960***
High-Quality Land	0.2141***	0.0319***	-0.0019	-0.0405***	-0.2036***
Weighted Distance Index[a]	0.0300***	-0.00700	0.0100***	0.0300***	-0.0600***
Distance Wilderness Park[a]	0.0400***	-0.1000***	0.3000***	-0.0200***	-0.2000***
Distance Urban Park[a]	0.0300***	-0.0090	-0.4000***	-0.1000***	0.5000***
Climatic Factor	0.0007***	0.0012***	-0.0003***	0.0002***	-0.0018***

*, **, *** indicate significance at α = 10%, 5%, and 1%.
[a] Marginal effects are multiplied by 1000 to facilitate presentation. Observations: 512,201. % Correct Predictions: 60.7%

Table 3: NB2 Estimates for the Watershed Health Indicator Models

Land Use	*Conventional Water Quality*	*Toxic Water Quality*	*Species at Risk*
Constant	-2.718***	-5.132***	0.905***
	(< 0.0001)	(0.0017)	(< 0.0001)
Urban Land	0.013*	0.034*	0.007*
	(0.0708)	(0.0745)	(0.0812)
Agricultural Land	0.018***	0.002	0.002
	(0.0002)	(0.9225)	(0.4058)
Transportation Land		1.444**	0.301***
		(0.0206)	(0.0012)
Mining Land		0.4972*	0.113
		(0.0928)	(0.1054)
Other Land 1[a]	0.012***		
	(0.0018)		
Other Land 2[b]		0.039**	0.002
		(0.0286)	(0.4149)
Irrigated Land	1.67 E-04	-0.0075*	-5.52 E-05
	(0.6245)	(0.0861)	(0.7524)
Soil Loss due to Water Erosion	0.017		
	(0.7246)		
Soil Loss due to Wind Erosion	-0.021		
	(0.5743)		
Index of Soil Permeability	-0.124		
	(0.1368)		
Area of Water Bodies			-2.39 E-04
			(0.7957)
Area of Wetlands			0.005***
			(0.0057)
Total Land Area			1.80E-04***
			(0.0048)
Observations	155	31	279
Deviance/DF	1.20	1.43	0.96
Pearson χ^2/DF	0.93	0.93	1.03

p- values in parenthesis

*, **, *** indicate significance at $\alpha = 10\%$, 5%, and 1%, respectively.

Estimates for spatial dummies not reported. Available upon request.

[a] Other land 1 includes transportation land, minor land (incl. mining land), CRP land, and federal land.

[b] Other land 2 includes minor land (excl. mining land), CRP land, and federal land.

Table 4: Mean Percentage of Watershed in Each Land Use

Land Use Policy	*% of Watershed in Ag. Land*	*% of Watershed in Urban Land*	*% of Watershed in Forest Land*
Baseline	23.69%	5.25%	10.99%
Urban Returns			
↓Urban Returns 5%	23.74%	5.01%	11.03%
↓Urban Returns 25%	23.94%	4.16%	11.17%
↓Urban Returns 50%	24.14%	3.30%	11.30%
Agriculture Returns			
↑ Ag. Returns 5%	23.69%	5.25%	10.99%
↑ Ag. Returns 25%	23.71%	5.24%	10.99%
↑ Ag. Returns 50%	23.74%	5.24%	10.98%
Reforestation Payments on Ag. Land			
↑ Forest. Returns 5%	23.67%	5.24%	11.04%
↑ Forest. Returns 25%	23.61%	5.23%	11.24%
↑ Forest. Returns 50%	23.53%	5.20%	11.53%
Incentive-Based Policies			
↑ Inc.-Based Index 5%	23.72%	5.23%	11.13%
↑ Inc.-Based Index 25%	23.86%	5.16%	11.64%
↑ Inc.-Based Index 50%	24.02%	5.08%	12.25%
↑ Inc.-Based Index to 1.0	25.51%	4.60%	16.96%
Property Acquisition			
↑ Prop.-Acq. Index 5%	23.71%	5.24%	10.86%
↑ Prop.-Acq. Index 25%	23.79%	5.22%	10.37%
↑ Prop.-Acq. Index 50%	23.90%	5.18%	9.80%
↑ Prop.-Acq. Index to 0.6	24.03%	5.01%	5.90%

Table 5: Mean Percentage Change in Conventional Water Pollution Relative to Baseline

Land Use Policy	*Most Urban Land*	*Least Urban Land*	*Most Ag. Land*	*Least Ag. Land*	*Most Forest Land*	*Least Forest Land*
Urban Returns						
↓Urban Returns 5%	-1.32%	-0.02%	-0.06%	-0.93%	-0.20%	-0.03%
↓Urban Returns 25%	-5.89%	-0.08%	-0.26%	-4.22%	-0.91%	-0.13%
↓Urban Returns 50%	-14.03%	-0.22%	-0.82%	-10.13%	-2.19%	-0.38%
Agriculture Returns						
↑ Ag. Returns 5%	0.05%	0.00%	0.00%	0.01%	0.00%	0.00%
↑ Ag. Returns 25%	0.25%	0.00%	0.01%	0.04%	0.01%	0.01%
↑ Ag. Returns 50%	0.53%	0.00%	0.02%	0.08%	0.02%	0.01%
Reforestation Payments on Ag. Land						
↑ Forest. Returns 5%	-0.10%	-0.06%	-0.02%	-0.05%	-0.05%	0.00%
↑ Forest. Returns 25%	-0.51%	-0.32%	-0.12%	-0.26%	-0.24%	-0.01%
↑ Forest. Returns 50%	-1.08%	-0.72%	-0.25%	-0.54%	-0.51%	-0.03%
Incentive-Based Policies						
↑ Inc.-Based Index 5%	-0.05%	0.08%	0.07%	-0.05%	-0.03%	0.11%
↑ Inc.-Based Index 25%	-0.26%	0.39%	0.33%	-0.27%	-0.15%	0.54%
↑ Inc.-Based Index 50%	-0.56%	0.79%	0.65%	-0.57%	-0.29%	1.08%
↑ Inc.-Based Index to 1.0	-2.93%	4.50%	4.70%	-0.86%	-1.00%	6.55%
Property Acquisition						
↑ Prop.-Acq. Index 5%	0.10%	-0.02%	0.00%	0.09%	0.05%	-0.03%
↑ Prop.-Acq. Index 25%	0.48%	-0.08%	0.01%	0.42%	0.25%	-0.15%
↑ Prop.-Acq. Index 50%	0.91%	-0.15%	0.03%	0.79%	0.48%	-0.28%
↑ Prop.-Acq. Index to 0.6	1.29%	-0.45%	-1.12%	0.90%	2.92%	-1.77%

Table 6: Mean Percentage Change in Toxic Water Pollution Relative to Baseline

Land Use Policy	*Most Urban Land*	*Least Urban Land*	*Most Ag. Land*	*Least Ag. Land*	*Most Forest Land*	*Least Forest Land*
Urban Returns						
↓Urban Returns 5%	-4.2%	-0.14%	-0.29%	-1.73%	-0.79%	-0.18%
↓Urban Returns 25%	-17.4%	-0.68%	-1.37%	-7.32%	-3.57%	-0.84%
↓Urban Returns 50%	-28.1%	-1.27%	-2.56%	-12.09%	-6.32%	-1.58%
Agriculture Returns						
↑ Ag. Returns 5%	-0.04%	0.00%	0.00%	0.00%	0.00%	0.00%
↑ Ag. Returns 25%	-0.19%	0.00%	0.00%	-0.01%	0.00%	0.00%
↑ Ag. Returns 50%	-0.39%	0.00%	0.00%	-0.01%	0.00%	0.00%
Reforestation Payments on Ag. Land						
↑ Forest. Returns 5%	-0.05%	0.00%	-0.01%	-0.01%	-0.03%	0.00%
↑ Forest. Returns 25%	-0.25%	-0.01%	-0.04%	-0.06%	-0.16%	-0.01%
↑ Forest. Returns 50%	-0.53%	-0.01%	-0.07%	-0.11%	-0.34%	-0.01%
Incentive-Based Policies						
↑ Inc.-Based Index 5%	-0.07%	0.00%	-0.03%	-0.05%	-0.08%	0.00%
↑ Inc.-Based Index 25%	-0.34%	-0.01%	-0.13%	-0.25%	-0.29%	0.01%
↑ Inc.-Based Index 50%	-0.69%	-0.02%	-0.23%	-0.50%	-0.54%	0.02%
↑ Inc.-Based Index to 1.0	-11.23%	0.01%	-1.12%	-3.74%	-3.71%	-0.34%
Property Acquisition						
↑ Prop.-Acq. Index 5%	-0.07%	-0.01%	-0.01%	-0.12%	-0.01%	-0.01%
↑ Prop.-Acq. Index 25%	-0.45%	-0.06%	-0.06%	-0.64%	-0.06%	-0.07%
↑ Prop.-Acq. Index 50%	-1.05%	-0.12%	-0.13%	-1.33%	-0.13%	-0.13%
↑ Prop.-Acq. Index to 0.6	-1.56%	-0.86%	-1.06%	-0.11%	0.04%	-1.21%

Table 7: Mean Percentage Change in the Number of Species at Risk Relative to Baseline

Land Use Policy	*Most Urban Land*	*Least Urban Land*	*Most Ag. Land*	*Least Ag. Land*	*Most Forest Land*	*Least Forest Land*
Urban Returns						
↓Urban Returns 5%	-0.96%	-0.01%	-0.03%	-0.78%	-0.22%	-0.02%
↓Urban Returns 25%	-4.36%	-0.06%	-0.15%	-3.54%	-0.97%	-0.09%
↓Urban Returns 50%	-7.61%	-0.12%	-0.29%	-6.23%	-1.70%	-0.17%
Agriculture Returns						
↑ Ag. Returns 5%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
↑ Ag. Returns 25%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
↑ Ag. Returns 50%	0.00%	0.01%	0.01%	0.00%	0.00%	0.00%
Reforestation Payments on Ag. Land						
↑ Forest. Returns 5%	-0.03%	0.00%	0.00%	-0.02%	-0.01%	0.00%
↑ Forest. Returns 25%	-0.15%	0.00%	-0.02%	-0.10%	-0.07%	0.00%
↑ Forest. Returns 50%	-0.31%	-0.01%	-0.05%	-0.22%	-0.14%	0.00%
Incentive-Based Policies						
↑ Inc.-Based Index 5%	-0.04%	0.01%	0.00%	-0.05%	-0.02%	0.01%
↑ Inc.-Based Index 25%	-0.22%	0.03%	0.02%	-0.24%	-0.09%	0.03%
↑ Inc.-Based Index 50%	-0.46%	0.07%	0.03%	-0.49%	-0.17%	0.06%
↑ Inc.-Based Index to 1.0	-2.02%	0.45%	0.36%	-1.38%	-0.84%	0.70%
Property Acquisition						
↑ Prop.-Acq. Index 5%	0.00%	0.00%	0.00%	0.00%	0.01%	0.00%
↑ Prop.-Acq. Index 25%	-0.03%	-0.01%	-0.01%	-0.03%	0.03%	-0.01%
↑ Prop.-Acq. Index 50%	-0.11%	-0.02%	-0.02%	-0.11%	0.06%	-0.03%
↑ Prop.-Acq. Index to 0.6	0.02%	-0.04%	-0.32%	0.09%	0.32%	-0.40%