Land Use and Watershed Health in the United States

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ABSTRACT. *This national-scale, watershed-level analysis provides an empirical assessment of land use impacts on water quality and aquatic ecosystems in the United States. Results suggest that the level of conventional water pollution in a watershed is significantly affected by the amount of land allocated to intensive agriculture and urban development, while the level of toxic water pollution is significantly affected by the amount of land allocated to transportation and mining. We examine the relationship between land use, water quality, and aquatic species extinction and discuss the implications of the results for the design and implementation of the water quality trading policy.* (JEL Q24, Q53, Q57)

I. INTRODUCTION

Land use change is arguably the most pervasive socioeconomic driving force affecting watershed ecosystems (Dale et al. 2000). Runoff from agricultural lands is a leading cause of water pollution both in inland and in coastal waters. The drainage of wetlands and irrigation water diversions have brought many wildlife species to the verge of extinction. Urban land development has also been linked to many environmental problems, including urban runoff, water pollution, and loss of wildlife habitat. Habitat destruction, fragmentation, and alteration associated with urban development have been identified as the leading causes of biodiversity decline and species extinctions (McKinney 2002; Rottenborn 1999). Land use in coastal areas and further inland is also a major threat to the health, productivity, and biodiversity of the marine environment in the United States and throughout the world (Intergovernmental Conference 1995).

Although there is a large amount of scientific evidence that land use affects water quality and ecosystems, the relative impacts of alternative land uses on water quality and ecosystems have rarely been quantified. Such information is essential for the design and evaluation of policies aiming at protecting water quality and ecosystems, which typically involve various landscape management options, such as altering management practices on cultivated croplands, retiring land from crop production, or restoring some land to its natural state (e.g., Carpenter et al. 1998, Wu et al. 2004).[1] Recently, several states in the United States have begun to use water quality trading as a policy tool to control water pollution and to mitigate the ecological impacts of land use, and many other states are actively considering the policy (U.S. Environmental Protection Agency 2003a).[2] Water quality trading is an inno-

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[1] To increase adoption of these management options, many policies have been implemented at the federal and state levels. These include various conservation programs established under the U.S. Farm Bills, such as the Conservation Reserve Program and the Environmental Quality Incentive Program, and numerous mandatory measures authorized under the Clean Water Act and the Endangered Species Act.

[2] The objectives of the policy are to encourage voluntary trading programs that facilitate implementation of TMDLs, reduce the cost of compliance with the Clean Water Act regulations, establish incentives for voluntary reductions, and promote watershed-based initiatives (U.S. Environmental Protection Agency 2003a). The EPA's 2003 Water Quality Trading Policy supports trading to achieve nutrient (e.g., total phosphorus and nitro-

vative, market-based approach that allows one source to meet its regulatory obligations by using pollutant reductions created by another source that has lower pollution control costs. As marketable pollution permits, it has the potential to achieve water quality goals at a lower social cost. However, much information is needed to implement the water quality trading policy. At the local watershed level, data are needed on the relative impacts of alternative land uses and conservation practices on water pollution and ecosystems. At the state or national level, the information is needed to develop general guidelines for trading and to target geographic areas or land use changes that are most effective to achieve overall national goals. The objective of this study is to provide such information by evaluating the effect of alternative land uses on selected indicators of water quality and watershed ecosystems.

Numerous studies have examined the structure and functions of various components of ecosystems at the watershed or river-basin scales (see discussion in Sections 2 and 3). Although these studies have provided piecemeal evidence that land use affects water quality and ecosystems, no study, to our knowledge, has analyzed multiple aspects of watershed health at the regional or national scales. Watershed ecosystems are complex assemblages of plants, animals, and microbes interacting with each other and their environment. The complexity of ecosystems requires a system approach. This study treats watersheds as ecosystems by analyzing the interaction between water quality and aquatic health as affected by land use and other human activities in watersheds covering the lower 48 states of the United States. It concentrates on water pollution related to agricultural and urban runoff, eutrophication, and toxic pollution. It analyzes how water quality affects the status of wetland and aquatic species[3] and whether land-use changes exacerbate the impacts. This watershed-level analysis provides a "big-picture" of the ecological impact of land use at the national scale. Such a "big-picture" analysis is useful because large-scale changes in land use are needed to improve the overall health of the nation's ecosystems.

The remainder of this paper is organized as follows. Section 2 discusses several selected indicators of watershed health in the United States. Section 3 reviews the biological and ecological literature to identify the critical relationships among land use, water quality, and wildlife abundance and presents the empirical specification of these relationships. Section 4 provides justifications for the empirical specification and describes the estimation methods. Section 5 discusses the data. Section 6 presents the empirical results. Section 7 discusses major findings and policy implications. Section 8 concludes the paper.

II. WATERSHED INDICATORS IN THE UNITED STATES

Freshwater ecosystems, in addition to being valuable in their own right, are indispensable for the functioning of terrestrial ecosystems, and are largely responsible for maintaining and supporting overall environmental health (U.S. Environmental Protection Agency 2004a). In this study, three indicators were selected to describe the health of aquatic resources across the United States. These indicators were retrieved from the U.S. Environmental Protection Agency (USEPA)'s Index of Watershed Indicators, which contains data characterizing the condition and vulnerability of aquatic systems in watersheds across the United States (U.S. Environmen-

gen) and sediment load reductions, as well as cross-pollutant trading for oxygen-related pollutants. Water quality trading is currently being implemented or actively considered in about ten states (U.S. Environmental Protection Agency 2003b).

[3] Aquatic organisms are exceptionally vulnerable to the outside environmental conditions and their health provides an early indicator of the state of the environment (Blaustein, Wake, and Sousa 1994; Hartwell and Ollivier 1998). Aquatic ecosystems are also characterized by the highest levels of species endangerment and extinction rates (Stein, Kutner, and Adams 2000).

FIGURE 1
CONVENTIONAL AMBIENT WATER QUALITY (% SAMPLES EXCEEDING CRITERIA)

tal Protection Agency 2004b). Each of the indicators is discussed below.

The *conventional ambient water quality* indicator (CONVWQ) measures the number of surface water samples in a watershed with concentrations of one or more of the 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 data sufficiency threshold requires that each watershed must contain at least 20 observations representing a minimum of five sites over the nine-year period. Figure 1 shows the percentage of surface water samples that exceed the national reference levels for the four conventional ambient water quality measures across the 2,100 watersheds in the contiguous Unites States. Conventional water quality appears to be a nation-wide problem, with more frequent violation of the USEPA standard in the Midwest, the Gulf coast, and the Atlantic coast.

The concentrations of phosphorus and ammonia and the level of dissolved oxygen and pH are important indicators of water quality. It has been well documented that excessive eutrophication associated with high concentrations of phosphorus and ammonia may cause algal blooms, increase water turbidity, generate hypoxic or anoxic conditions, and change aquatic biodiversity (e.g., Carpenter et al. 1998; Smith 1998). The level of dissolved oxygen (DO) is affected by a number of factors, including eutrophication, photosynthesis of plants and planktonic algae, decomposition of all organic matter, as well as abiotic factors such as temperature and atmospheric pressure (e.g., Deaton and Winebrake 2000; Faurie et al. 2001).[4] Acidification can disrupt the nitrogen cycle in freshwater ecosystems (Vitousek et al. 1997) and has been identified with the occurrence of decreased diversity of animal and plant species (Schindler 1994). The causes and effects of conventional ambient water pollution and the related literature are summarized in Table A1 in Appendix A.

The *toxic ambient water quality* indicator (TOXICWQ) measures the number of

[4] Depletion of DO levels is closely related to biological oxygen demand (BOD). BOD is the indicator of pollution by biodegradable organic matter present in water. It is the amount of oxygen required to completely oxidize a quantity of organic matter by biological processes (Keyes 1976).

surface water samples in a watershed with concentrations of one or more of the four toxic pollutants (copper, nickel, zinc, chromium) exceeding the national chronic levels. This indicator, however, does not capture pollution by toxic compounds other than the selected four heavy metals. It is constructed based on water quality monitoring data collected between 1990 and 1998. The data sufficiency threshold is the same as for CONVWQ. Figure 2 displays the percentage of surface water samples that exceed the national chronic levels for the four toxic pollutants. It shows a clustered pattern of toxic water pollution nationwide, although limited data availability may obscure the overall spatial pattern. Watersheds in the Rocky Mountains and parts of the eastern and southern United States have more serious toxic water pollution problems.

Contamination of water bodies by heavy metals is a major concern due to their sedimentation, persistence, and bioaccumulation potential, and their lethal and sub-lethal effects. Elevated concentrations of toxic substances affect aquatic wildlife in a number of ways. These include changes in morphology, physiology, body biochemistry, behavior, and reproduction (e.g., Skidmore 1964; Handy and Eddy 1990). The causes and effects of toxic ambient water pollution and the related studies are summarized in Table A2 in Appendix A.

The *species-at-risk* indicator (SPERISK) measures the number of aquatic and wetland species (plants and animals) at risk of extinction in a given watershed in 1996. Figure 3 shows the indicator across the contiguous United States. The figure shows that no area in the United States has been spared the threat to aquatic biodiversity, although the southern states and the West Coast have more aquatic species at risk of extinction than other parts of the country.

Aquatic ecosystems are characterized by a great deal of biodiversity. 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 (e.g., Hartwell and Ollivier 1998; Welsh and Ollivier 1998; Blaustein and Johnson 2003). Fish are considered useful indicators of biological integrity and ecosystem health since

FIGURE 2
TOXIC AMBIENT WATER QUALITY (% SAMPLES EXCEEDING CRITERIA)

FIGURE 3
AQUATIC AND WETLAND SPECIES AT RISK OF EXTINCTION (NUMBER OF SPECIES)

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). The causes and mechanisms of biodiversity losses with the corresponding literature are summarized in Table A3 in Appendix A.

III. EMPIRICAL MODELS

A recursive equation system is estimated to evaluate the impacts of land use on water quality and aquatic ecosystems. The system consists of three equations, representing models of conventional water quality, toxic water quality, and species at risk of extinction, respectively. Land use affects both conventional and toxic water quality, which in turn affect the number of species at risk of extinction. Each model is discussed below.

The Conventional Water Quality Model

The conventional water quality model captures the relationship between different types of land uses and their effect on water quality via the processes of eutrophication and dissolved oxygen depletion. Although eutrophication is a natural process that occurs in virtually all water bodies, its anthropogenic acceleration is a major concern (e.g., Laws 1993; Schnoor 1996). Excessive eutrophication and water pollution have been linked to agricultural land and chemical uses, urban runoff, and topographic and hydrological characteristics (see Table A1 for a summary of major causes of conventional water pollution). Based on previous studies, the conventional water quality model is specified as

$$\ln(CONVWQ_i) = \ln N_i^c + \beta_0 + \boldsymbol{\beta}_1'\boldsymbol{l}_i^c + \boldsymbol{\beta}_2'\boldsymbol{p}_i^c + \boldsymbol{\beta}_3'\boldsymbol{d}_i^c + \varepsilon_i^c, \quad [1]$$

where i is the watershed index, N_i^c is the total number of samples taken to measure conventional water quality, $\boldsymbol{l}_i^c$ is a vector of land- and chemical-use variables affecting conventional water pollution, $\boldsymbol{p}_i^c$ is a vector of physical characteristics measuring the vulnerability of individual watersheds to conventional water pollution, $\boldsymbol{d}_i^c$ is a vector of spatial dummies, and ε_i^c is an error term, with $\exp(\varepsilon_i^c)$ following the gamma distribution. The justification for the empirical specification is given in the next section.

The Toxic Water Quality Model

The toxic water quality model represents the relationship between different types of land uses and the presence of heavy metals in a watershed.[5] The major anthropogenic sources of metallic pollution of water bodies include urban, industrial, and commercial land use and mining (see Table A2 for a summary). The whole life cycle of a metallic pollutant, from ore extraction and processing to manufacturing, domestic and industrial use, and disposal, may cause water contamination (e.g., Keyes 1976; Fergusson 1982). Domestic uses, sewage, urban runoff, and traffic also contribute to heavy metal contamination (Alloway 1995). Intensive agriculture is a major non-point source of metals, with the main sources being impurities in fertilizers, sewage sludge, manures from intensive hog and poultry production, and pesticides (Alloway 1995). Based on these previous studies, the toxic water quality function is specified as

$$\ln(TOXICWQ_i) = \ln N_i^t + \gamma_0 + \boldsymbol{\gamma}_1' \boldsymbol{l}_i^t + \boldsymbol{\gamma}_2' \boldsymbol{p}_i^t + \boldsymbol{\gamma}_3' \boldsymbol{d}_i^t + \varepsilon_i^t, \qquad [2]$$

where N_i^t is the total number of samples taken to measure toxic water pollution, $\boldsymbol{l}_i^t$ is a vector of land- and chemical-use variables affecting toxic water pollution, $\boldsymbol{p}_i^t$ is a vector of physical characteristics measuring the vulnerability of individual watersheds to surface water pollution, $\boldsymbol{d}_i^t$ is a vector of spatial dummies, and ε_i^t is an error term, with $\exp(\varepsilon_i^t)$ following the gamma distribution.[6]

The Species-at-Risk Model

The third equation in the system models the effect of water quality on aquatic ecosystems. Previous studies have identified several factors potentially affecting the quality of aquatic environment (see Table A3 for a summary).

The decline in biodiversity of both animal and plant species has been linked to a number of conventional water pollution problems, including excessive eutrophication (e.g., Vitousek et al. 1997). The mechanisms by which an algal bloom eventually leads to changes in species diversity in eutrophic systems, as well as the effects of eutrophication on species biomass and diversity, are varied (see references in Table A3 for details).

Changes in species diversity and abundance have also been attributed to elevated concentrations of toxic substances. The toxicity of a compound varies across different species, individuals, age, life history, DO levels, water hardness, water pH, and the level of pollutant concentration (chronic vs. acute) (e.g., Handy and Eddy 1990; Laws 1993).

Habitat alterations and changes in physical conditions of habitats, such as wetland drainage, wetland fragmentation, river damming and channelization, and other types of hydrologic modification, have been identified as another major factor determining species composition and population abundance in aquatic ecosystems (Faurie et al. 2001).[7]

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[5] The discussion is focused on the sources of metallic pollution since the USEPA's toxic water quality indicator (TOXICWQ) measures pollution by selected heavy metals (Cr, Cu, Ni, Zn).

[6] The conventional and toxic water quality models contain both, the erosion rate variables as well as land use variables. While erosion rates are affected by land use, they also characterize the physical vulnerability of a watershed to water pollution.

[7] For example, Frissell (1993) studied the causes of ichthyofaunal impoverishment in drainage basins of the Pacific Northwest and California, and found that cumulative damage to aquatic habitats caused by logging, grazing, urbanization, and other land uses plays a major role in species diversity losses. Richter et al. (1997) assessed the threats to freshwater fauna in the United States through an experts survey and identified three leading threats nationwide: altered sediment loads and nutrients inputs from agricultural nonpoint pollution; interference from exotic species; and altered hydrologic regimes associated with impoundment operations. Czech, Krausman, and Devers (2000) found that urbanization endangers more species in the mainland United States than any other human activity. Harding et al.

The literature seems to be conclusive in identifying nutrient loading, toxic pollution, and habitat alterations as the major factors affecting the abundance and diversity of aquatic life. Accordingly, the species-at-risk equation is specified as follows

$$\ln(SPERISK_i) = \delta_0 + \delta_1 CONVWQ_i + \delta_2 TOXICWQ_i + \delta_3' \boldsymbol{l}_i^s + \delta_4' \boldsymbol{d}_i^s + \varepsilon_i^s, \quad [3]$$

where $\boldsymbol{l}_i^s$ is a vector of land-use and habitat variables, $\boldsymbol{d}_i^s$ is a vector of spatial dummies, and ε_i^s is an error term, with $\exp(\varepsilon_i^s)$ following the gamma distribution. Since the total number of aquatic and wetland species in each watershed is unknown a corresponding ln N component, which appears in the other 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 United States. Equation [3], together with [1] and [2], constitute our equation system. The justifications for the specification of [1]–[3] and the estimation method are discussed in the next section.

IV. ESTIMATION METHOD

The Poisson and negative binomial models have been suggested for estimating the number of occurrences of an event, or event counts (Maddala 1983, 51; Cameron and Trivedi 1998). In this study, an event count is the number of samples violating the national water quality standard or the number of endangered species in a watershed. Formally, an event count is defined as a realization of a nonnegative integer-valued random variable y. The Poisson model is derived by assuming that y is Poisson-distributed with the conditional density of y equal to $f(y|\mathbf{x}) = (e^{-\theta}\theta^y)/y!$, where $\theta = E[y|\mathbf{x}]$. The log of the mean θ is assumed to be a linear function of a vector of independent regressors $\mathbf{x}$: $\ln\theta = \mathbf{x}'\boldsymbol{\beta}$, 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. One is that it assumes that the sample size is constant (Maddala, 53). But sample sizes often change in cross-sectional analyses. To address this problem, Maddala suggests an alternative specification. Let N be the total sample corresponding to y so that the rate of occurrence is y/N. With the sample size information, the Poisson model can be reparameterized as $\ln\theta = \ln N + \mathbf{x}'\boldsymbol{\beta}$. In this study, the sample size is known for conventional and toxic water pollution measures, but unknown for the species-at-risk indicator.

Another problem with the Poisson specification is the restriction imposed by the equidispersion property.[8] Table 1 shows that conditional mean and conditional variance are likely to be different for each of the three dependent variables. The standard way to account for overdispersion is the NB2 model suggested by Cameron and Trivedi (1998). They derive this negative binomial model from a Poisson-gamma mixture distribution (Cameron and Trivedi, 100–102). In addition to y being conditionally Poisson-distributed, parameter θ is assumed to be the product of a deterministic term and a random term, $\theta = e^{\mathbf{x}'\beta+\varepsilon} = e^{\mathbf{x}'\beta}\, e^{\varepsilon} = \mu\nu$. 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 $V[y|\mu, \alpha] = \mu + \alpha\mu^2$. The model of the form $\ln\theta = \mathbf{x}'\boldsymbol{\beta} + \varepsilon$ is estimated using maximum likelihood methods, along with the dispersion parameter α.

Instrumental variable (IV) techniques are used to estimate the equation system [1]–[3]. First, equations [1] and [2] are

(1998) investigated the influence of past land use on the present-day diversity of stream invertebrates and fish in river basins in North Carolina and found that past land-use activity, in particular agriculture, was the best predictor of present-day aquatic diversity.

[8] The property $E[y|\mathbf{x}] = V[y|\mathbf{x}] = \theta$ is referred to as the equidispersion property of the Poisson.

estimated as NB2 models using the GENMOD procedure in SAS. These equations are then used to predict CONVWQ and TOXICWQ. Finally, equation [3] is estimated as a NB2 model using the predicted values of CONVWQ and TOXICWQ.

V. DATA

The data used in this study come from three main sources: The USEPA's Index of Watershed Indicators (IWI), the USDA's National Resources Inventories (NRI), and the NOAA's Coastal Assessment and Data Synthesis System. All datasets have been retrieved by the eight-digit hydrologic units, the nationally consistent set of watersheds in the Hydrologic Unit Classification System developed by the USGS. The dataset includes about 2,100 watersheds.[9] Table 1 describes the variables selected for this study and basic statistics.

Land Use and Other Human Impacts

The USDA's NRI contain detailed data on land use, land cover, and natural resource conditions on nonfederal lands in the United States. The data are collected every five years from the same statistically based sample sites and are classified by state, county, major land resource area, and hydrologic unit. All variables used in this study were retrieved from the 1997 NRI database, which also contains data from previous three NRIs (1982, 1987, 1992). Unless mentioned otherwise, all variables are constructed as the percentage of total land area of the hydrologic unit and are averaged over the four NRI years (1982, 1987, 1992, 1997). This study uses the following NRI land-use categories: *cultivated cropland* (CC), *noncultivated cropland* (NONCC), *pastureland* (PAST), *rangeland* (RANG), *forestland* (FO), *urban land* (UR), *rural transportation land* (TR), *minor land* (MINOR) - with subcategory of *mining land* (MIN), *CRP land* (CRP) (measuring the land enrolled in the Conservation Reserve Program), and *federal land* (FED) measuring the land owned by the federal government.[10] These land-use categories completely describe the landscape. Hence, in order to avoid perfect multicollinearity, forestland (FO) was excluded from the regression models and used as the reference land use. *Irrigated land* (IRRIGacres) reflects the area that shows evidence of being irrigated during the year of the inventory or of having been irrigated during two or more of the last four years.

Population density (POPDEN) was calculated as the number of persons per acre in the watershed based on the 1990 Census population data. *Total storage* (STORAGE) measures the total storage capacity of dams and reservoirs in a given watershed. The volume of impounded water is an indicator of the degree of hydrologic modification. These variables were retrieved from USEPA's IWI database. The fertilizer and pesticide use variables (FERTUSE, PESTUSE) were obtained from NOAA's dataset, which contains data on the application of nitrogen and phosphorus fertilizers and numerous pesticides in agricultural production.[11]

Watershed Physical and Habitat Characteristics

Watershed ecosystems are affected not only by human activities but also by their own vulnerability. The vulnerability of a watershed to ecosystem damages is determined by a number of physical and habitat characteristics, including the following: total land area of the watershed (LANDacres); area of the watershed covered by permanent open water (WATERacres) con-

[9] Smith, Schwarz, and Alexander (1997) suggest that these watersheds are a logical choice for characterizing national-level water quality because they represent a systematically developed and widely recognized delineation of U.S. watersheds, and provide a spatially representative view of water quality conditions.

[10] Detailed definitions of the land use/cover categories can be found in the *NRI glossary*.

[11] The data come from two sources—The National Center for Food and Agricultural Policy and the Census of Agriculture. The dataset includes statistics on 185 and 208 chemical compounds for the years 1987 and 1992, respectively (NOAA 1999). Only the 1992 data were used in this study, since NOAA expressed some reservations about the reliability of the 1987 vintage.

TABLE 1

DEFINITIONS OF VARIABLES AND DESCRIPTIVE STATISTICS

Variables	Description	Data Source	Years	Observations	Mean	Std. Dev.	Min.	Max.
CONVWQ	Conventional water quality: number of samples in exceedance of national reference levels for concentrations of phosphorus, ammonia, dissolved oxygen, and pH in surface waters	USEPA - IWI	1990–1998	1,344	586.94	2,593.92	0	56,476
N^C	Total number of samples taken to measure conventional water quality	USEPA - IWI	1990–1998	1,344	3469.1	11,889.01	20	204,168
TOXICWQ	Toxic water quality: number of samples in exceedance of national chronic levels for concentrations of copper, nickel, zinc, and chromium in surface waters	USEPA - IWI	1990–1998	758	70.86	234.16	0	4,323
N^T	Total number of samples taken to measure toxic water quality	USEPA - IWI	1990–1998	758	731.16	1,121.56	20	13,059
SPERISK	Species at risk: number of aquatic and wetland species at risk of extinction	USEPA - IWI	1996	1,595	4.56	4.94	1	47
POPDEN	Population density: based on 1990 U.S. Census (persons/acre)	USEPA - IWI	1990	2,062	18.31	65.06	0	1,314.74
SOILPERM	Index of soil permeability	USEPA - IWI	1998	2,109	4.17	1.21	0	8.96
STORAGE	Total storage capacity of dams and reservoirs (1,000 acre-feet)	USEPA - IWI	1995-96	1,915	262.03	1,277.3	0.001	28,255
UR	Urban land (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	4.39	8.82	0	87.47
TR	Rural transportation (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	1.18	0.68	0	3.39
CC	Cultivated cropland (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	18.7	23.99	0	91.73
NONCC	Noncultivated cropland (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	2.46	3.21	0	31.83
PAST	Pastureland (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	6.95	9.28	0	71.99
RANG	Rangeland (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	18.97	26.25	0	99.98

FO	Forestland (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	24.68	27.43	0	99.55
CRP	CRP land (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	1.01	2.06	0	16.72
MINOR	Minor land (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	2.75	5.09	0	81.41
MIN	Mining land (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	0.27	1.33	0	41.52
OMINOR	Other minor land (excl. mining land) (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	2.48	4.82	0	81.41
FED	Land owned by the federal government (% watershed land area)	USDA - NRI	1982,87,92,97	2,109	18.9	27.99	0	100
IRRIGacres	Irrigated land (1,000 acres)	USDA - NRI	1982,87,92,97	2,109	31.86	107.14	0	3240.18
USLE	Soil loss due to water erosion (tons/acre/year)	USDA - NRI	1982,87,92,97	2,109	1.86	1.97	0	18.64
EIWIND	Soil loss due to wind erosion (tons/acre/year)	USDA - NRI	1982,87,92,97	2,109	2.08	7.16	0	146.5
WETLANDacres	Area of palustrine and estuarine wetlands (1,000 acres)	USDA - NRI	1997	2,109	52.65	102.72	0	1524.9
LANDacres	Total land area of watershed (1,000 acres)	USDA - NRI	1982,87,92,97	2,109	895.3	570.23	0	5510.58
WATERacres	Area of water bodies in watershed (1,000 acres)	USDA - NRI	1982,87,92,97	2,109	23.43	57.99	0	860
FERTUSE	Fertilizer use: average annual nitrogen and phosphorus fertilizer use in agriculture (lbs per acre of fertilized area)	NOAA	1982–1991	2,074	359.48	2,908.05	0	60814.15
PESTUSE	Pesticide use: average annual pesticide use in agricultural production (lbs per acre of agricultural area)	NOAA	1992	2,071	2.07	5.48	0.0015	129.38

Note: Statistics for the NRI-based variables were computed for the four-year (1982, 1987, 1992, 1997) averages.

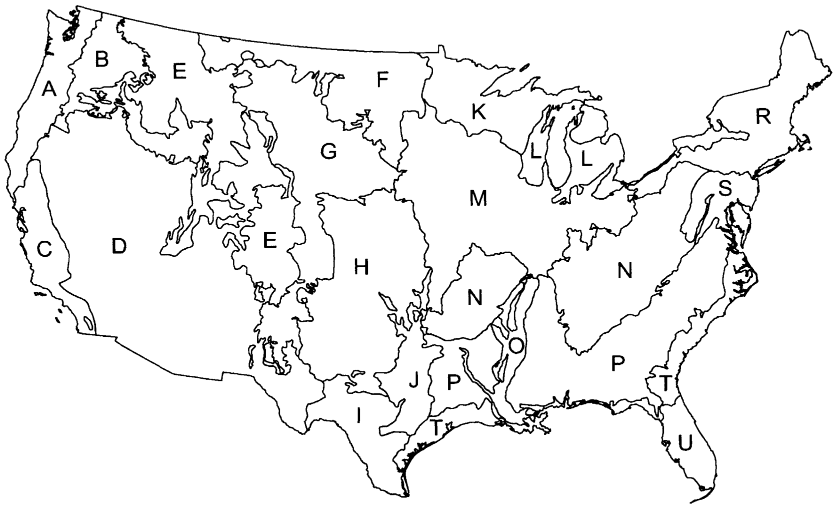

FIGURE 4
LAND RESOURCE REGIONS

structed as the sum of NRI census water and small water areas; area of palustrine and estuarine wetlands (WETLANDacres), as defined by the Cowardin classification system; average wind erosion rate in the watershed (EIWIND), based on the NRI wind erosion estimates; average water erosion rate in the watershed (USLE), based on the NRI soil erosion estimates (USDA 2000); and soil permeability index (SOILPERM) based on the IWI data.[12]

Spatial Dummies

Spatial dummies are included in order to capture some of the spatial variability across the large study area. Two sets of spatial dummy variables are used in this analysis—Land Resource Regions (20 regions) and Ecosystem Divisions (21 divisions). Land Resource Regions, defined by the USDA's Soil Conservation Service (Figure 4), are characterized by a particular pattern of soils, climate, water resources, and land uses.[13] Ecosystem Divisions (Figure 5) are the second level in a four-level ecoregion hierarchy. The USDA's Forest Service defines divisions as areas that share common climatic, precipitation, and temperature characteristics.[14]

VI. ESTIMATION RESULTS

The equation system [1]–[3] is estimated to evaluate the effect of land use on water quality and aquatic species. 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 likelihood ratio tests and the Pearson/DF and deviance/DF measures also indicate that the Poisson distribution

[12] The US 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 groundwater resources, and consequently quality of surface waters where ground water feeds rivers and lakes (USEPA 2004b).

[13] For detailed characterization of the Land Resource Regions see Soil Survey Staff (1981) or http://www.soilinfo.psu.edu/soil_lrr/.

[14] For detailed characterization of the Ecosystem Divisions see Bailey (1995) or http://www.essc.psu.edu/soil_info/soil_eco/.

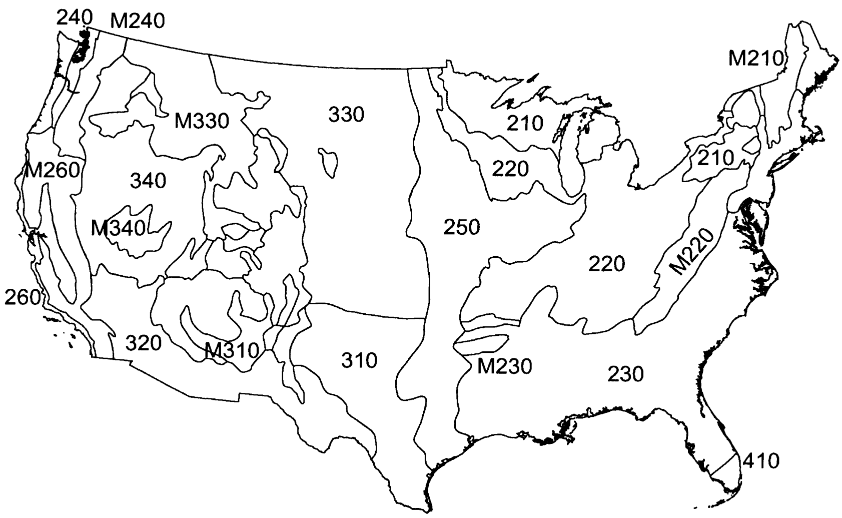

FIGURE 5
Ecosystem Divisions

assumption is inappropriate.[15] Thus, only results from the NB2 models are reported in this paper.

The Conventional Water Quality Model

Table 2 presents parameter estimates for two specifications of the conventional water quality model. The basic specification uses shares of alternative land uses as explanatory variables, and the alternative model uses population density and the average annual fertilizer use per acre as explanatory variables instead of the land use variables. The coefficients on urban land (UR), cultivated cropland (CC), and pastureland (PAST) in the basic model are positive and statically significant at the 1% level, or better. Given that forest is used as a reference land use, these results indicate that converting forests to developed land, cultivated cropland, or pastureland increases conventional water pollution. Although the impacts of crop production and urban runoff on water quality have been well documented and the results concerning these land uses are expected, the impact of pastureland on water pollution is less intuitive. A common characteristic of cultivated cropland and pastureland is that they both are treated with fertilizer application. The USDA defines *pastureland* (PAST) as area that is managed primarily for the production of introduced forage plants and where management usually consists of cultural treatments such as fertilization and weed control. This is in contrast to *rangeland* (RANG) which is defined as area covered with native grasses, grasslike plants, forbs or shrubs suitable for grazing and browsing, with little or no chemicals or fertilizer being applied. As expected, the coefficient on EIWIND is positive, and significant, indicating that wind erosion exacerbates conventional water quality problems.

In the alternative conventional water quality model, the land use variables are replaced with population density and fertilizer use. The coefficient on population density (POPDEN) is positive and statis-

[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).

TABLE 2

ESTIMATED COEFFICIENTS FOR THE CONVENTIONAL AND TOXIC WATER QUALITY MODELS WITH THE NB2 SPECIFICATION

	Conventional Water Quality Model				Toxic Water Quality Model			
	(1a) Basic Model		(1b) Alternative Model		(2a) Basic Model		(2b) Alternative Model	
Variables	Coefficient	St. Error	Coefficient	St. Error	Coefficient	St. Error	Coefficient	St. Error
Intercept	−2.0283***	(0.2343)	−2.8338***	(0.1704)	−2.2533***	(0.5432)	−4.0067***	(0.3612)
UR	0.0118***	(0.0026)			0.0077	(0.0055)		
TR	0.0293	(0.0547)	0.1168**	(0.0534)	0.4847***	(0.1338)	0.3084**	(0.1276)
CC	0.0120***	(0.0018)			−0.0044	(0.0040)		
NONCC	−0.0042	(0.0085)			−0.0297	(0.0189)		
PAST	0.0105***	(0.0033)			−0.0192***	(0.0074)		
RANG	0.0030	(0.0022)			0.0060	(0.0054)		
CRP	−0.0074	(0.0164)	0.0067	(0.0165)	0.0931**	(0.0428)	0.0490	(0.0404)
MINOR	0.0063	(0.0043)	0.0045	(0.0048)				
MIN					0.0771*	(0.0446)	0.0833*	(0.0451)
OMINOR					0.0031	(0.0100)	0.0068	(0.0105)
FED	−0.0010	(0.0020)	−0.0041**	(0.0017)	0.0075	(0.0046)	0.0080**	(0.0038)
POPDEN			0.0011***	(0.0003)			0.0007	(0.0009)
FERTUSE			1.87E-05	(1.29E–05)				
PESTUSE							0.0090	(0.0205)
IRRIGacres	4.55E–05	(0.0002)	0.0004*	(0.0002)	0.0006	(0.0008)	0.0005	(0.0007)
USLE	0.0059	(0.0147)	0.0233	(0.0144)	−0.0215	(0.0298)	−0.0204	(0.0287)
EIWIND	0.0071*	(0.0039)	0.0102**	(0.0044)	0.0198	(0.0231)	0.0198	(0.0233)
SOILPERM	0.0122	(0.0202)	−0.0232	(0.0200)	−0.1107**	(0.0529)	−0.0699	(0.0489)
Spatial Dummies								
Region A	−1.0548***	(0.2092)			−0.1969	(0.8050)	1.4202*	(0.7515)
Region B	−0.2282	(0.2071)	0.9348***	(0.1785)	−1.7410**	(0.7247)	0.0149	(0.6589)
Region C			1.0441***	(0.2108)			1.7400***	(0.5257)
Region D	−0.1524	(0.1853)	1.0157***	(0.1441)	−0.9701**	(0.4440)	0.8493**	(0.3395)
Region E	−0.3728**	(0.1862)	0.7692***	(0.1422)	0.0548	(0.4326)	1.7217***	(0.3292)
Region F	−0.2089	(0.2241)	1.2792***	(0.1839)	−1.6023***	(0.5458)	0.1675	(0.4296)
Region G	−0.2588	(0.2207)	0.9947***	(0.1859)	−1.2550**	(0.5321)	0.7425*	(0.4229)
Region H	−0.5266***	(0.1964)	0.8520***	(0.1537)	−1.6144***	(0.4592)	0.2986	(0.3127)
Region I	−0.7479***	(0.2452)	0.5604***	(0.2157)	−1.8632**	(0.7395)	−0.0697	(0.6809)
Region J	−0.5753**	(0.2269)	0.7808***	(0.1824)	−1.8120***	(0.5402)	−0.1876	(0.4327)
Region K	−0.4070**	(0.2074)	0.7393***	(0.1550)	−0.8486*	(0.4859)	0.6742*	(0.3503)
Region L	−0.5025**	(0.2196)	0.9896***	(0.1697)	−1.4709***	(0.4532)		
Region M	−0.3184	(0.2056)	1.2040***	(0.1471)	−1.2738***	(0.4398)	0.1721	(0.2759)
Region N	−0.5711***	(0.2023)	0.5746***	(0.1380)	−0.2660	(0.4314)	1.1134***	(0.2527)
Region O	−0.1898	(0.2586)	1.1123***	(0.2255)	−0.9566*	(0.5520)	0.5320	(0.4422)
Region P	−0.3155	(0.1997)	0.7999***	(0.1388)	0.0225	(0.4424)	1.6200***	(0.2562)
Region R	−0.7252***	(0.2074)	0.3400**	(0.1591)	−0.0187	(0.4587)	1.5065***	(0.3079)
Region S	−0.7002***	(0.2265)	0.4889***	(0.1859)	−0.3380	(0.4668)	1.1424***	(0.3433)
Region T	−0.0703	(0.2024)	1.1393***	(0.1519)	−0.2382	(0.4358)	1.4365***	(0.2812)
Region U	−0.0984	(0.2443)	1.2755***	(0.2107)	−0.0239	(0.5260)	1.2853***	(0.4368)
Dispersion	0.4702***	(0.0193)	0.4905***	(0.0201)	1.1707***	(0.0695)	1.2031***	(0.0714)
Observations	1,343		1,330		757		751	
Deviance/DF	1.1311		1.1288		1.2087		1.2068	
Pearson X2/DF	1.0740		1.0761		1.3636		1.3519	
Log Likelihood	4,943,982		4,933,349		211,359		211,129	

Notes: The dependent variables are ln(CONVWQ) and ln(TOXICWQ), respectively. Forest land serves as the reference land use. Standard errors are deviance-scaled.

* = Significant at the 10% level; ** = significant at the 5% level; *** = significant at the 1% level.

tically significant at the 1% level. This is consistent with the coefficient on urban land use in the basic model. However, the coefficient on fertilizer use (FERTUSE) is statistically insignificant at the 10% level, although it is positive as expected.

Both models of conventional water quality are estimated using the NB2 assumption, and land-use variables are averages over the four NRI years (1982, 1987, 1992, 1997). The qualitative results, however, are robust to alternative specifications of functional forms (linear or log-linear OLS), to error term distribution assumptions (Poisson or NB2), to data selections (based solely on the 1997 NRI data or the four-year average), and to the choice of dummy variables (using the 20 regions or 41 production areas as spatial dummies).[16] Thus, there is strong empirical evidence that intensive agriculture and urban development contribute to conventional water quality problems in the United States.

The sign and magnitude of the coefficients of spatial dummies indicate the degree of water quality concerns in these regions relative to the chosen reference unit. For example, selecting Region C (central and southern California valleys) as the reference dummy yields all coefficients of the spatial dummies negative, approximately a half of them significant. This is not surprising given that Region C has a large agricultural sector and is experiencing rapid urbanization. On the other hand, if Region A (western part of U.S. Pacific Northwest) is selected as a reference region, all spatial dummies have a positive coefficient. This is expected for a region characterized by extensive forests and relatively low levels of urbanization.

Both the continuous land use variables and the spatial dummies tend to identify the same factors affecting water pollution. The reason may be that the dummies capture the vulnerability of a region for water pollution (e.g., high natural background concentrations), with intensive farming and urbanization exacerbating the problem. Furthermore, while the continuous regressors capture only the extent of a particular land use, the dummies are also likely to capture the intensity of land use as well as differences in cropping systems.

The Toxic Water Quality Model

Estimation results for two specifications of the toxic water quality model are also presented in Table 2. The basic model uses land use shares as explanatory variables, and the alternative one uses population density and the average annual pesticide use per acre as explanatory variables. Results show that toxic water pollution in a watershed is significantly affected by the amount of land allocated to mining (MIN) and transportation (TR) according to both specifications. Surface water samples taken in watersheds with more land allocated to mining and transportation and less to forests are more likely to have toxic pollutant concentrations (copper, nickel, zinc, chromium) above the national chronic level. However, the coefficients on the shares of urban land and cultivated cropland are insignificant in the basic model, nor are the coefficients on population density and pesticide use in the alternative specification. These results are also robust to alternative specification of functional forms, as well as to the choice of data and spatial dummies.[17] From a national perspective, mining and transportation are the two major causes of toxic water pollution.

[16] A third set of spatial dummies, the 41 production areas, was constructed by aggregating the Major Land Resource Areas (MLRA). The USDA defines a MLRA as a geographic area that is characterized by a particular pattern of soils, climate, water resources, land uses, and type of farming. One way of aggregating the MLRAs yields the 20 Land Resource Regions. We clustered the MLRAs into 41 spatial units based on their geographic proximity, climate, land cover, and other characteristics (http://www.nrcs.usda.gov/technical/land/mlra/mlralegend.html). Results of the conventional water quality model using the 41 production areas as spatial dummies are qualitatively and quantitatively very similar to the results reported in Table 2. No changes in signs of the key variables of interest occur. The significance level of TR improves to 5% and 1% in the basic and alternative models, respectively.

[17] Using the 41 production areas as spatial dummies in the toxic water quality model causes no changes in

TABLE 3
ESTIMATED COEFFICIENTS FOR THE SPECIES-AT-RISK MODEL WITH THE NB2 SPECIFICATION

	Species-at-Risk Model			
	(3a) Basic Model		(3b) Alternative Model	
Variables	Coefficient	St. Error	Coefficient	St. Error
Intercept	1.5207***	(0.0979)	−0.3989	(0.3548)
Endogenous Variables				
CONVWQ	4.52E–05***	(1.54E–05)	4.79E–05***	(1.52E–05)
TOXICWQ	2.17E-04	(2.81E–04)	2.85E–04	(3.24E–04)
Exogenous Variables				
STORAGE	−3.91E–05	(5.54E–05)	−4.22E–05	(5.51E–05)
WATERacres	6.21E–04	(5.39E–04)	7.67E–04	(5.51E–04)
WETLANDacres	1.46E–04	(3.76E–04)	1.06E–04	(3.75E–04)
LANDacres	4.13E–04***	(7.13E–05)	4.03E–04***	(7.13E–05)
Spatial Dummies				
Division 210	−0.7437***	(0.1491)	1.1844***	(0.3661)
Division M210	−0.6372**	(0.2748)	1.2905***	(0.4314)
Division 220	−0.0432	(0.0963)	1.8852***	(0.3440)
Division M220	−0.0085	(0.1340)	1.9031***	(0.3558)
Division 230			1.9158***	(0.3513)
Division M230	−0.0062	(0.4377)	1.9193***	(0.5483)
Division 240	−0.8800	(0.8474)	0.9963	(0.9190)
Division 250	−0.8494***	(0.1310)	1.0763***	(0.3520)
Division 260	0.2135	(0.3728)	2.3407***	(0.5402)
Division M260	0.1480	(0.4190)	2.0709***	(0.5319)
Division 310	−0.6352***	(0.2251)	1.2910***	(0.3947)
Division M310	−0.5105	(0.3527)	1.4242***	(0.4699)
Division 320	−0.4595*	(0.2436)	1.4750***	(0.3988)
Division 330	−1.5923***	(0.1921)	0.3399	(0.3743)
Division M330	−1.3573***	(0.1729)	0.5716	(0.3671)
Division 340	−1.8586***	(0.3323)		
Division 410	−1.7944**	(0.8890)	−0.1689	(1.0075)
Dispersion	0.3820***	(0.0309)	0.3782***	(0.0308)
Observations	614		611	
Deviance/DF	0.9846		0.9844	
Pearson X2/DF	1.1030		1.1011	
Log Likelihood	3,527		3,534	

Notes: The dependent variable is ln(SPERISK). Standard errors are deviance-scaled.
* = Significant at the 10% level; ** = significant at the 5% level; *** = significant at the 1% level.

As for the spatial dummies, Region C (central and southern California valleys) and Region L (parts of Great Lakes states) serve as the reference dummies in the basic and alternative models, respectively. The generally high levels of toxic contamination in the former compared to the relatively low levels in the latter region can explain the different results. It is noteworthy that in the basic model only two regions (Region E and P) have a positive sign, though insignificant. The former is characterized by high concentration of mining, the latter by sprawling urban growth.

signs of the key land-use variables. In the basic model, significance of CC and NONCC improves to the 1% and 10% levels, respectively. MIN becomes insignificant. In the alternative model, both TR and MIN become insignificant.

The Species-at-Risk Model

Parameter estimates for two specifications of the species-at-risk model are presented in Table 3. The results show that water pollution, both conventional and toxic, increases the number of aquatic and wetland species at risk of extinction in a watershed, ceteris paribus. However, the toxic pollution variable, TOXICWQ, is statistically insignificant. Since the species-at-risk model focuses on aquatic species, land use variables such as shares of developed land and cultivated cropland are not included as independent variables in the final model. When land use variables are also included in the species-at-risk model, they tend to be insignificant. Those significant often have signs consistent with interpretations of representing the size of habitat. Thus, land use variables are not included in the final model except those describing the size and conditions of habitat (WATERacres, WETLANDacres, LANDacres, STORAGE). These results suggest that land use affects aquatic species mainly through its impact on water quality.

One critical issue related to the analysis of interactions between habitat conditions and species richness is that it is not obvious whether the presence of rare or endangered species in a watershed necessarily indicates (1) poor environmental conditions leading to species endangerment; or (2) quite the opposite, high-quality conditions providing habitat for species not found elsewhere. To address this issue, we include several groups of control variables in the regression model. First, a group of acreage variables (WATERacres, WETLANDacres, LANDacres) are included to account for the size of the watershed and its aquatic habitat. Large watersheds with more wetlands and larger areas of water bodies are more likely to have more species including those at risk of extinction. Second, a set of spatial dummies (Ecosystem Divisions) are included to account for the species richness across the watersheds. We assume that an area with low species diversity is likely to have only few rare species (ceteris paribus), while an area with high species diversity is more likely to have many rare species. The third group of variables (CONVWQ, TOXICWQ, STORAGE) are included to control for the human impacts. Hence, this specification will help isolate the partial effects of habitat size, species diversity, and selected habitat quality variables.

The reference dummies chosen in this case include subtropical Division 230 characterized by high levels of aquatic biodiversity, and the temperate desert Division 340 characterized by generally low level of aquatic biodiversity. Results in Table 3 show that coefficients of the more biologically diverse ecoregions in the eastern United States (e.g., Divisions 220, M220, 230, M230) and California (Divisions 260, M260) tend to be greater than those of the steppe and desert ecoregions in the western United States (Divisions 310, M310, 320, 330, M330, 340).

In sum, there is evidence that poor water quality is likely to intensify the stress on aquatic ecosystems and contribute to species endangerment. The mutual interdependence of watershed health and water quality implies a need for systemic policies. For this reason, a policy aimed at decreasing the threats to biodiversity has to address the problems of conventional and toxic water quality.

VII. MAJOR FINDINGS AND POLICY IMPLICATIONS

The effect of alternative land uses on water quality and watershed ecosystems is evaluated using the empirical models. Table 4 shows the elasticities of the three selected watershed indicators with respect to land-use variables that are statistically significant in at least one equation (at the 10% level or better) and are robust to empirical specifications. The elasticities are calculated using the formula shown in Appendix B and are evaluated at the sample mean. Because forestland is used as a reference land use in both the conventional and toxic water quality models, the impacts of alternative land uses should be interpreted relative to the impact of forest land use.

TABLE 4
ESTIMATED ELASTICITIES OF WATERSHED INDICATORS WITH RESPECT TO ALTERNATIVE LAND USES

	Dependent Variables					
	CONVWQ		TOXICWQ		SPERISK	
Independent Variables	(1a)	(1b)	(2a)	(2b)	(3a)	(3b)
CONVWQ					0.0266***	0.0281***
TOXICWQ					0.0153	0.0202
UR	0.0519***		0.0338		0.0019**	
TR	0.0346	0.1381**	0.5731***	0.3646**	0.0097	0.0113
CC	0.2244***		−0.0823		0.0047	
PAST	0.0730***		−0.1334***		−0.0001	
MIN			0.0209*	0.0226*	0.0003	0.0005
POPDEN		0.0195***		0.0130		0.0008
IRRIGacres	0.0015	0.0128*	0.0176	0.0146	0.0003	0.0007

Notes: Elasticities are evaluated at the sample means. The variables represent conventional water quality (CONVWQ), toxic water quality (TOXICWQ), species at risk of extinction (SPERISK), % urban land (UR), % transportation land (TR), % cultivated cropland (CC), % pastureland (PAST), % mining land (MIN), population density (POPDEN), and irrigated acreage (IRRIGacres).
* = Significant at the 10% level; ** = significant at the 5% level; *** = significant at the 1% level.

The first two columns of Table 4 show the impact of alternative land uses on conventional water quality. There is strong evidence that converting forests to intensive agriculture and urban development contributes to conventional water pollution in the United States. A 1% increase in cultivated cropland (1,674 acres for an average watershed) increases the number of samples in exceedance of the national reference level for conventional water quality by 0.22% in an average watershed, while a 1% increase in developed land (393 acres for an average watershed) increases the number of samples in exceedance of the national reference level for conventional water quality by 0.05%. Converting forestland to pasture also increases conventional water pollution. A 1% increase in pastureland (622 acres for an average watershed) increases the number of samples in exceedance of the national reference level for conventional water quality by 0.07%.

Columns 3 and 4 of Table 4 show the impact of alternative land uses on toxic water quality. Converting forestland to transportation or mining will significantly increase the probability of toxic water pollution. A 1% increase in the amount of land allocated to transportation and mining (106 acres and 24 acres, respectively, in an average watershed) is expected to increase the number of samples in exceedance of the national chronic level for toxic water quality by 0.57% and 0.02%, respectively.

The last two columns of Table 4 show the impact of alternative land use variables on the number of endangered species in an average watershed. The conventional water quality variable is statistically significant at the 1% level. A 1% increase in the percent of samples exceeding the national reference level for conventional water quality is expected to increase the number of endangered aquatic species by about 0.03%. However, the toxic water quality measure is insignificant in the model of endangered species at the 10% level. Although land use variables are not included as independent variables in the species-at-risk model, they affect aquatic species indirectly through their impacts on water quality. These indirect impacts of land uses are also estimated and reported in the last two columns of Table 4 (see the formula in Appendix B). A 1% increase in acreages of developed land and transportation increases the number of endangered aquatic species by 0.002% and 0.01%, respectively.

As shown above, because land is not equally divided among alternative uses, a 1% increase in alternative land uses represents different acres. For example, developed land accounts for only 4.39% of total

TABLE 5
THE PER-ACRE IMPACT OF ALTERNATIVE LAND USES ON THE SELECTED WATERSHED INDICATORS

	Estimated Impact (%) ($\times 10^{-04}$)					
	CONVWQ		TOXICWQ		SPERISK	
One Acre of Land Use Category	(1a)	(1b)	(2a)	(2b)	(3a)	(3b)
UR	1.3180***		0.8600		0.0482**	
TR	3.2726	13.0459**	54.1381***	34.4464**	0.9177	1.0633
CC	1.3403***		−0.4915		0.0281	
PAST	1.1728***		−2.1445***		−0.0018	
MIN			8.6116*	9.3041*	0.1321	0.1881

Notes: The variables represent conventional water quality (CONVWQ), toxic water quality (TOXICWQ), species at risk of extinction (SPERISK), % urban land (UR), % transportation land (TR), % cultivated cropland (CC), % pastureland (PAST), and % mining land (MIN).
* = Significant at the 10% level; ** = significant at the 5% level; *** = significant at the 1% level.

land area on average, while cultivated cropland accounts for 18.7% of total land area on average. Thus, a 1% increase in developed land translates to 393 acres, while a 1% increase in cultivated cropland translates into 1,674 acres. To compare the impacts of alternative land uses on water quality, the results are converted to the per-acre impacts in Table 5 using the formula shown in [B5] in Appendix B. For an average watershed, converting 1% of forestland (2,210 acres) into urban development increases the percentage of samples in exceedance of the national reference level for conventional water quality by 0.29%.[18] It has about the same impact as converting the same amount of forests to cultivated cropland, but has a slightly larger impact than converting forests to pastureland on a per-acre basis.

Converting forestland to highways and mining both increases toxic water pollution, but transportation has a much larger impact than mining on a per-acre basis. Converting 1% of forestland to transportation will increase the percentage of samples in exceedance of the national chronic level for toxic water quality by 7.61% in an average watershed based on the alternative model, which is about four times larger than the impact of converting the same amount of forestland to mining. Transportation also has a larger impact on endangered aquatic species than mining and urban development on a per-acre basis because it has a larger impact on toxic water pollution.

These results have important implications for water quality trading policies under consideration in many states of the United States. Water quality trading is an innovative, market-based approach that allows one source to meet its regulatory obligations by using pollutant reductions created by another source that has lower pollution control costs. Our results show that trading for reducing conventional water pollution should focus on intensive agriculture and urban development, while trading for reducing toxic water pollution should focus on transportation and mining. In an average watershed, one acre of urban development on forestland can be offset by converting one acre of cultivated cropland to forests in terms of impact on conventional water quality. However, to offset the impact on toxic water quality of one acre of highways built in forests, 3.7 acres of mining must be converted to forests. It is important to note that these results are estimated based on the marginal effects. For large land-use changes, the models instead of the elasticities must be used to calculate the approximate trading ratios. In addition, because the marginal effects of land use on water quality are not constant, the amount of land use conversions needed to offset a negative water quality impact will

[18] 1.3180e-4 per acre × 2,210 acres = 0.29, using results in Table 5.

be different for watersheds with different size and mix of land uses. Trading ratios may also be different for inter-watershed trading than intra-watershed trading. For example, equation [B5] shows that one acre of urban development will have a much larger effect on water quality and species in a small watershed than in a large watershed. Thus, in general, the environmental effect of one acre of development cannot be offset by purchasing one acre of development rights in another watershed. The empirical models estimated in this study can be used to calculate trading ratios for both intra- and inter-watershed trading.

The empirical results also have implications for the design and evaluation of conservation programs. Soil erosion, particularly wind erosion, is found to increase both conventional and toxic water pollution. Thus, conservation programs that aim at reducing soil erosion will contribute to improving water quality and aquatic habitat. Given that biodiversity and endangered species are significantly affected by water quality and land uses, it is important to take an ecosystem approach in the design of policies for protecting biodiversity and endangered species (Main, Roka, and Noss 1999).

The most important legislative initiative for the species protection in the United States is the Endangered Species Act (ESA). Compared to the previous legislative efforts, ESA expanded the available conservation measures to include all methods and procedures necessary to protect the species rather than emphasizing only habitat protection (Switzer 2004). Although the ESA has been recognized as "a powerful and sensible way to protect biological diversity" (Carroll et al. 1996, 2), it has been subjected to extensive criticism from both natural scientists (e.g., Carroll et al. 1996; Switzer 2004) and economists (e.g., Brown and Shogren 1998) for ineffective use of scientific information and for sidelining economics. Our empirical results suggest that habitat conditions, particularly water quality, are important factors determining the number of aquatic species at risk of extinction in a watershed.

VIII. CONCLUSION

Land use issues are a manifestation of the fundamental economic fact of scarcity. The limited land supply implies that more land in one use means less land being left for an alternative use. Although markets play a central role in land allocation, they may fail to allocate land efficiently in the presence of externalities and improper incentives. For example, market prices of developed land may not reflect the environmental damages caused by urban runoff, nor do they account for the loss of wildlife habitat. These externalities may cause developed land being overvalued. However, it is difficult to develop policies to correct market failures in land allocation because of lack of information on the relative impacts of alternative land uses on water quality and ecosystems.

This study has important implications for water quality trading policies under consideration in many states of the United States. Our results show that trading for reducing conventional water pollution should focus on agricultural and urban land use, while trading for reducing toxic water pollution should focus on transportation and mining. In an average watershed, one acre of urban development on forestland can be offset by converting one acre of cultivated cropland to forests in terms of impact on conventional water quality; however, to offset the impact on toxic water quality of one acre of highways built in forests, 3.7 acres of mining must be converted to forests. In general, trading ratios are different for watersheds with different sizes and mixes of land use. The models estimated in this study can be used to calculate such trading ratios.

This study accentuates the "big picture" analysis by examining the relationship between land use, water quality, and aquatic species extinction across the United States. This nation-wide analysis has two major limitations. First, the ecological impact of land use is inherently a spatial issue. Conducting a spatially explicit analysis may yield valuable insights. However, dimensionality and data limitations prevent us from considering locations of economic activities within a watershed. Second, interactions among land use, water quality, and watershed health are intrinsically dynamic. There also may be time lags between land use changes and their ecological impacts. Explicitly modeling these interactions and time lags would be an important topic for future research. Currently, considering these spatial and temporal dimensions is constrained by the lack of spatially explicit, time-series data. In particular, our analysis highlights the need for time-series data on species abundance and habitat characteristics. Despite these limitations, this study provides information for the design and evaluation of land use-based policies that aim at improving water quality and ecosystems. Considering spatial and temporal dimensions in future research with improved data will provide additional insights. This study provides a useful first step in setting priorities for effective and efficient restoration actions.

APPENDIX A

TABLE A1
THE CAUSES AND EFFECTS OF IMPAIRED CONVENTIONAL AMBIENT WATER QUALITY

Causes	References	Effects	References
Cultural (Excessive) Eutrophication			
Discharge of organic waste, treated and untreated sewage Nutrient loading caused by urban and agricultural runoff Agricultural practices (e.g., fertilizer and chemical application rates, crop management practices) Topographic and hydrological characteristics	Carpenter et al. 1998 Faurie et al. 2001 Laws 1993 Ryszkowski 2002 Schindler 1977 Schnoor 1996 Anderson, Opaluch, and Sullivan 1985 Barbash et al. 2001 De Roo 1980 Gilliam and Hoyt 1987 Kellogg et al. 1992 Malmqvist and Rundle 2002 Smith, Alexander, and Wolman 1987 Wu et al. 1997 Wu and Babcock 1999	Increased growth of algae (algal blooms), aquatic weeds, and other phytoplankton Increased water turbidity Wide fluctuations of dissolved oxygen concentration causing hypoxic or anoxic conditions Changes in species composition and biomass, loss in faunal and floral diversity Adverse effects on aesthetic and recreational values	Brouwer, Thomas, and Chadwick 1991 Carpenter et al. 1998 Faurie et al. 2001 Laws 1993 Mason 1977 Ryszkowski 2002 Sayer et al. 1999 Schindler 1990, 1994 Schnoor 1996 Seehausen, van Alpen, and Witte 1997 Smith 1998 Vitousek et al. 1997
Dissolved Oxygen Depletion			
Abiotic factors (including temperature and atmospheric pressure) Biotic factors (including photosynthesis of plants and planktonic algae)	Faurie et al. 2001	Oxygen shortages leading to fish kills and changes in aquatic biodiversity	Carpenter et al. 1998 Smith 1998
Organic waste (including domestic, farm), industrial effluents, or urban runoff	Alloway 1995 Deaton and Winebrake 2000 Fergusson 1982		
Acidification			
Atmospheric nitrogen deposition	Vitousek et al. 1997	Disruption of the nitrogen cycle in freshwater ecosystems	Vitousek et al. 1997
		Decreased faunal and floral diversity	Schindler 1988, 1990, 1994

TABLE A2
THE CAUSES AND EFFECTS OF IMPAIRED TOXIC AMBIENT WATER QUALITY

Causes	References	Effects	References
Metal mining (including ore extraction, smelting, and processing)	Fergusson 1982 Keyes 1976 McGowen and Basta 2001 Malmqvist and Rundle 2002	*Contamination of sediments* in aquatic environments (metallic pollution is highly persistent in time)	Erichsen Jones 1958 Hare, Carignan, and Herta-Diaz 1994 Tessier et al. 1993 Sengupta 1993 Welsh and Denny 1980
Industrial processes (e.g., metallurgy, electronics, electrical manufacturing, petroleum refining, or chemical industry) Contamination may occur by: Emission of aerosols and dusts and consequent atmospheric deposition; Discharge of effluents into water ways; Creation of waste dumps in which metals become corroded and leached in the underlying soil.	Alloway 1995 Fergusson 1982 Stephenson 1987	*Bioaccumulation* of metallic contaminants in aquatic organisms	Handy and Eddy 1990 Laws 1993 Novotny and Olem 1994 Van der Zanden and Rasmussen 1996 Walker 1990 Novotny and Olem 1994 Skidmore 1964 Van der Zanden and Rasmussen 1996 Walker 1990
Domestic uses (sewage, urban runoff, and traffic)	Alloway 1995 Fergusson 1982 Malmqvist and Rundle 2002		
Intensive agriculture (e.g., impurities in fertilizers, sewage sludge, manures from intensive hog and poultry production, pesticides)	Alloway 1995		

TABLE A3
THE CAUSES AND EFFECTS OF CHANGES IN AQUATIC BIODIVERSITY

Causes	References	Effects	References
Conventional Water Pollution			
Excessive eutrophication and its ramifications, e.g., algal blooms creating generally uninhabitable environment, with some bloom-forming species being toxic; and oxygen shortages caused by senescence and decomposition of nuisance plants	Carpenter et al. 1998 Ryszkowski 2002 Sayer et al. 1999 Schindler 1990, 1994 Schnoor 1996 Seehausen et al. 1997 Smith 1998 Vitousek et al. 1997	Changes in species composition and biomass of aquatic fauna and flora caused by:dominance of a few highly competitive species tolerant of high nutrient concentrations; reduced habitat heterogeneity; and higher competition and predation	Brown 1987 Carpenter et al. 1998 Deaton and Winebrake 2000 Laws 1993 Mason 1977 Sayer et al. 1999 Smith 1998
Acidification	Schindler 1990, 1994		

(Table continued on following page)

TABLE A3
THE CAUSES AND EFFECTS OF CHANGES IN AQUATIC BIODIVERSITY (CONTINUED)

Causes	References	Effects	References
Toxic Water Pollution			
Toxicity of a compound varies across species, the individuals, their ages, life histories, and various environmental conditions such as pollutant concentration (chronic vs. acute), dissolved oxygen levels, water hardness, and pH.	Handy and Eddy 1990 Laws 1993 Skidmore 1964 Watras and Bloom 1992	Changes in morphology, physiology, body biochemistry, behavior, and reproduction	Handy and Eddy 1990 Laws 1993 Skidmore 1964 Waldichuk 1979
		Fish kills and increased stress	Skidmore 1964
		Reduction in photosynthesis	Laws 1993
		More complex response (additive, synergistic, or antagonistic effects) may occur due to simultaneous exposure to several to metallic contaminants	Skidmore 1964
Habitat Alterations			
Changes in physical condition of aquatic habitats, incl. water temperature, water currents, depth of the water column, turbidity, area of open water, sediment type and particle size, organic content of sediments	Faurie et al. 2001	Changes in species composition and population abundance in aquatic ecosystems	Faurie et al. 2001 Ehrlich and Ehrlich 1981 Harding et al. 1998
Blockage of migratory routes by dams	Angermeier 1995	Isolation of populations caused by habitat alterations (e.g., dam construction) can indirectly affect extinction rates of other species (e.g., non-migratory fish)	Angermeier 1995 Winston, Taylor, and Pigg 1991
Changes in riparian conditions	Raphael and Bisson 2003 Wipfli et al. 2002		
Location versus size of suitable habitat (e.g., fragmentation, connectedness)	Bockstael 1996 Lamberson et al. 1992 Montgomery, Brown, and Adams 1994	Disruption of wildlife interactions, changing wildlife populations, and communities	Rottenborn 1999
Disturbances associated with urban development, including noise, human presence, exotic species, habitat fragmentation	Rottenborn 1999	Decline in species richness along the urban-rural gradient, with the lowest richness usually found in the urban core	Czech, Krausman, and Devers 2000 McKinney 2002
Importance of the land use-ecosystem linkage at the regional or national scales; Land use, including logging, grazing, mining, industrial activities, fertilizer use, urban development; Cumulative damage to aquatic habitats caused by human land use; Regional versus local land-use pattern	Czech, Krausman, and Devers 2000 Ehrlich and Ehrlich 1981 Frissell 1993 Harding et al. 1998 Malmqvist and Rundle 2002 Richter et al. 1997 Rivard et al. 2000		

APPENDIX B

Calculation of Elasticities in Table 4

For notational simplicity let C = CONVWQ, T = TOXICWQ, S = SPERISK, and X_k denote an exogenous land use variable such as urban land or cultivated cropland. Elasticities in Table 4 were calculated as follows

$$\varepsilon_k^C = \frac{X_k}{C}\frac{\partial C}{\partial X_k} = \frac{\partial \ln C}{\partial X_k}X_k = \beta_k X_k, \qquad [B1]$$

$$\varepsilon_k^T = \frac{X_k}{T}\frac{\partial T}{\partial X_k} = \frac{\partial \ln T}{\partial X_k}X_k = \gamma_k X_k, \qquad [B2]$$

$$\varepsilon_k^S = \frac{X_k}{S}\frac{\partial S}{\partial X_k} = \frac{\partial \ln S}{\partial X_k}X_k = \delta_k X_k, \text{ if } X_k \text{ affects } S \text{ directly} \qquad [B3]$$

$$\varepsilon_k^S = \left(\frac{\partial \ln S}{\partial X_k}\right)X_k = \left(\frac{\partial \ln S}{\partial C}\frac{\partial C}{\partial X_k} + \frac{\partial \ln S}{\partial T}\frac{\partial T}{\partial X_k}\right)X_k = (\delta_1\beta_k C + \delta_2\gamma_k T)X_k, \text{ if } X_k \text{ affects } S \text{ through } C \text{ and } T. \qquad [B4]$$

All elasticities are evaluated at the sample mean. Variances of the elasticity estimates from [B1]–[B3] are calculated by $V(\varepsilon_k^j) = X_k^2 \cdot V(\beta_k \text{ or } \gamma_k \text{ or } \delta_k)$, where $j = C, T, S$. Variances of the elasticity estimates from [B4] were calculated using the Delta method (Zhou 2002, 669): $V(\varepsilon_k^S) = [V(\delta_1)\beta_k^2 + V(\beta_k)\delta_1^2]\overline{C}^2\overline{X}_k^2 + [V(\delta_2)\gamma_k^2 + V(\gamma_k)\delta_2^2]\overline{T}^2\overline{X}_k^2$.

Calculation of Factor Changes in Table 5

Each land-use variable in the models, X_k, is constructed as percentage of total land area in the watershed. Thus, a 1% change in X_k represents $(1\% \cdot LANDacres \cdot X_k/100)$ acres. Because ε_k^j measures changes in indicator j ($j = C, T, S$) as a result of a 1% change in X_k, a per-acre impact of variable X_k can be calculated by

$$\varphi_k^j = \frac{\varepsilon_k^j}{1\% \cdot LANDacres \cdot X_k/100} = \frac{\beta_k \text{ or } \gamma_k \text{ or } \delta_k}{LANDacres \cdot 10^{-4}}, \qquad [B5]$$

where the denominator is evaluated at the mean of X_k and *LANDacres*. φ_k^j measures the change in indicator j as a result of one acre increase in X_k.

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