Ecological Applications, 14(2), 2004, pp. 555–570

A TEST OF THE ENVIRONMENTAL KUZNETS CURVE USING LONG-TERM WATERSHED INPUTS

Sarah E. Gergel,[1,6] Elena M. Bennett,[2] Ben K. Greenfield,[2,7] Susan King,[3] Christine A. Overdevest,[4] and Basil Stumborg[5,8]

[1]*National Center for Ecological Analysis and Synthesis, Santa Barbara, California 93101 USA*
[2]*Center for Limnology, University of Wisconsin, Madison, Wisconsin 53706 USA*
[3]*Department of Chemical Engineering, Montana State University, Bozeman, Montana 59717 USA*
[4]*Department of Sociology and Rural Sociology, University of Wisconsin, Madison, Wisconsin 53706 USA*
[5]*Department of Agricultural and Applied Economics, University of Wisconsin, Madison, Wisconsin 53706 USA*

Abstract. The Environmental Kuznets Curve (EKC) postulates that environmental quality is initially degraded with increasing economic prosperity, until reaching some turning point where environmental quality improves with increases in wealth. Tests using environmental indicators beyond those that affect human health have been less supportive of the EKC idea. We hypothesize that environmental changes impacting human health are more likely to show evidence of an EKC than variables less directly related to human health. Furthermore, the EKC hypothesis assumes that ecological damage is reversible, and we argue that this assumption is not always valid. We test for evidence of an EKC in Dane County, Wisconsin, using non-point-source pollution time series data for Lake Mendota throughout the 20th century. We examine metals deposited in lake sediments (cadmium, chromium, copper, and lead), other non-point-source pollutants such as sulfur and dissolved reactive phosphorus (DRP), and water clarity (measured by Secchi disk depth). We relate changes in ecological variables to changes in real wealth per capita (RWPC) in Dane County over time. The EKC did not describe the relationship between all ecological and economic indicators tested; however, several variables were related to RWPC. Our strongest results (for Secchi depth, DRP, and copper) show *increasing* pollution with increasing wealth. Secchi depth and DRP are related to water quality and clarity, which have value to society but less direct, immediate health consequences. P pollution may also be fairly irreversible over short time scales. The best models and plots for cadmium, chromium, and lead suggest improvements in environmental quality with increases in RWPC, although these trends were not statistically significant. Results for sulfur were inconclusive. Overall, wealth did not explain much of the variability in any of the ecological variables examined here, suggesting that consideration of additional factors are necessary to explain their dynamics. Economic prosperity cannot be expected to improve all aspects of the environment, but may be biased toward aspects that are ecologically reversible phenomena or of concern to human health.

Key words: ecological economics; economic and ecological indicators; Environmental Kuznets Curve; heavy metals; irreversibility; Lake Mendota, Wisconsin; non-point-source pollution; phosphorus; water clarity; wealth.

Introduction

As the global population increases, concern over the impacts of human activities on environmental resources is growing. Some economists have suggested that the negative impacts of development can be mitigated by economic growth or that societies can ''grow'' out of environmental problems (Stern 1998). The World Bank has adopted this logic as an argument for sustainable development in Our Common Future (World Commission on Environment and Development 1987). Others have adopted similar positions:

> *[T]he strong correlation between incomes and the extent to which environmental protection measures are adopted demonstrates that, in the longer run, the surest way to improve your environment is to become rich.*
>
> —Beckerman (1992)

The logical extension of this rationale has policy implications:

> *Economic growth appears to be a powerful way for improving environmental quality . . . [this suggests]*

Manuscript received 2 December 2002; revised 12 May 2003; accepted 10 June 2003; final version received 7 July 2003. Corresponding Editor (ad hoc): R. L. Naylor.

[6] Present address: Department of Forest Sciences, University of British Columbia, Vancouver, British Columbia, Canada V6T 1Z4.

[7] Present address: San Francisco Estuary Institute, Oakland, California 94621 USA.

[8] Present address: BC Hydro, Vancouver, British Columbia, Canada V6B 5R3.

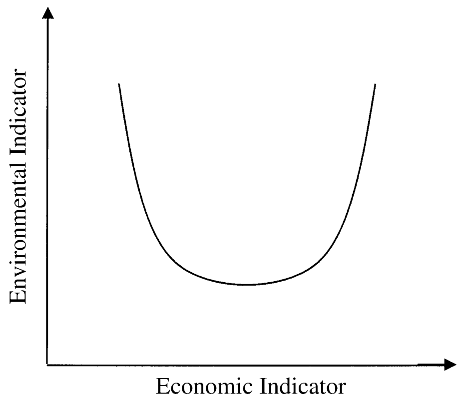

FIG. 1. Conceptual diagram of the EKC. The EKC hypothesis suggests that environmental quality is high initially in areas with little economic development. Then, environmental quality is degraded with increasing economic development until reaching a point where environmental quality improves with increasing levels of wealth.

that the environment need no particular attention, either in terms of domestic environmental policy, international pressure or assistance; resources can best be focused on achieving rapid economic growth.
—Panayotou (1993)

An important rationale behind such arguments comes from the Environmental Kuznets Curve (EKC) hypothesis (Panayotou 1993). The EKC, also called the U-Curve, suggests that high environmental quality exists in areas with little economic development and is initially degraded with increasing economic development. Eventually, once economic development proceeds past some critical turning point, environmental quality will improve (Fig. 1). The putative mechanism is that societies surpass some threshold level of wealth that allows them to ''purchase'' environmental quality through various means. Hypothesized causes for improved environmental quality with economic growth include technological advances, regulation, awareness, and education (Stern 1998). The name of the Environmental Kuznets Curve is in reference to work by economist Simon Kuznets, who postulated a U-curve relationship between income inequality and income levels (Kuznets 1955); he did not, however, introduce the idea of an Environmental Kuznets Curve.

Empirical support for an EKC has been demonstrated for some pollutants, generally those with local health effects that can be easily mitigated (Barbier 1997). For example, sulfur emissions and concentrations (which have high impacts on human health) have been found to decrease at higher levels of income (de Bruyn 1997, McConnell 1997). Evidence supporting an EKC has also been presented for SO_2, dark matter, ambient sulfur oxides, suspended particulates, NO_x, CO, and deforestation (Shafik and Bandyopadhyay 1992, Grossman and Krueger 1993, Panayotou 1993, Selden and Song 1994, Cole et al. 1997, Zaim and Taskin 2000, Bhattarai and Hammig 2001). An EKC has not been demonstrated unequivocally for any pollutant by more than one group of researchers (Ekins 1997, Rothman and de Bruyn 1998). Furthermore, while there is some evidence for an EKC for air pollutants, few other types of pollutants have been studied. More recent studies have examined the relationship between wealth and threatened and endangered species, wilderness area protection, number of hazardous waste sites, biochemical oxygen demand, chemical oxygen demand, nitrogen, pH, and suspended solids for water quality and total suspended particulates for air quality (Vincent 1997, Bohara et al. 2001, Gawande et al. 2001, Naidoo and Adamowicz 2001, Skonhoft and Solem 2001). Studies that have expanded the range of environmental variables have generally not found broad support for EKC relationships.

We propose that improvements in environmental quality with increasing economic development might be found only for pollutants for which there are both the *possibility* and *incentive* for management or mitigation. The EKC hypothesis assumes that ecological damage is reversible (Stern 1998), an assumption that is not necessarily valid (Carpenter et al. 1999). Biophysical characteristics of ecological indicators may influence whether or not environmental degradation is reversible through technological or behavioral fixes, export of the problem, or ecological processing (e.g., biogeochemical transformations or hydrologic flushing). Furthermore, incentives for mitigation may be influenced by the perceived value or threat associated with a change in an ecological variable. Under such an econometric view, variables linked most directly to human health impacts would be of utmost concern, followed by variables linked to quality of life or enjoyment of the environment, followed by variables that do not impact human health or quality of life. This potential hierarchy of value provides testable hypotheses for understanding which ecological variables might be expected to relate to wealth. We expect indicators that people have greater incentive to control (e.g., those that relate directly to human health) or are more easily mitigated to be more likely to exhibit an EKC relationship than those that are less closely related to human interests or are more difficult (or impossible) to mitigate.

We evaluate the long-term relationship between economic development and environmental quality in Lake Mendota, Dane County, Wisconsin, USA (Fig. 2), and test for evidence of an EKC. Lake water and sediments sequester pollutants and contaminants associated with human activities and provide a measure of the ecological impacts of humans in a watershed. Heavy metals, including lead, chromium, cadmium, and copper, are generated by industrial activities, fossil fuel burning, and urban runoff (de Luca et al. 1991, Benjamin and Honeyman 1992, Bannerman et al. 1993) and are stored

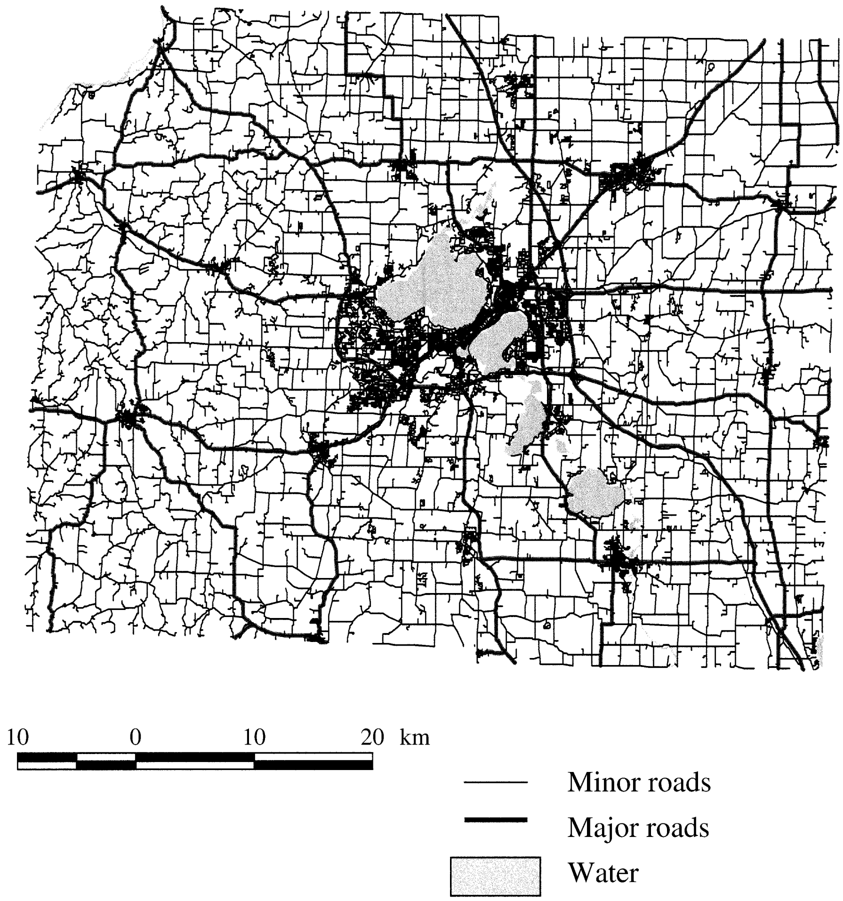

FIG. 2. Map of Dane County, Wisconsin, USA, showing Lake Mendota.

in lake bottom sediments. Thus, lake bottom sediments can provide an accurate historical record of such pollutants (Charles and Hites 1987). Elevated levels of sulfur in sediments as well as high dissolved reactive phosphorus (DRP) and low water clarity in the water column of lakes are indicative of pollution from urban and agricultural runoff, e.g., fertilizer, plant debris, and sewage (Brock 1985). Lakes with low water clarity and high DRP are often at higher risks for noxious algal blooms and are considered less desirable for recreation (Wilson and Carpenter 1999). Water clarity can easily be measured using a standard Secchi disk technique. Thus, heavy metals, Secchi disk depth, DRP, and sulfur in Lake Mendota are useful measures of environmental quality in Dane County.

Past critiques of EKC studies have focused on problems such as weak empirical evidence, faulty statistics, and the use of panel data. Panel or cross-sectional data use ecological and economic data from several countries to cover the range of economic development, rather than tracking one country throughout its economic development. It has been argued that Environmental Kuznets Curves estimated from panel data are inaccurate because they do not capture the dynamic interplay between economic growth and environmental pressure (de Bruyn et al. 1998). Instead, long-term data (from one area through time) are needed to examine the dynamics of areas experiencing growth (Stern 1998). The use of long-term, as opposed to cross-sectional data, is uncommon in tests of EKC, although it is arguably the most appropriate way to test the idea.

Our study is distinct for several reasons. We test for the EKC in one location over approximately 100 years (1900–2000) during a period of rapid economic growth (Fig. 3). We also examine a range of ecological indicators: indicators related directly to human health (heavy metals [World Health Organization 1977, 1988, 1992, 1998]) as well as ecological variables related to human use and enjoyment of lakes that are valued by society (water clarity [Steinnes 1992, Michael et al. 1996]). We hypothesize that biophysical aspects of the ecological indicator (such as its biogeochemical properties), as well as its potential human health impacts, may play a role in whether an EKC relationship is found for a particular pollutant. Our data provide an opportunity to examine some of the reasons and conditions under which we might expect economic development to improve environmental quality and to examine the

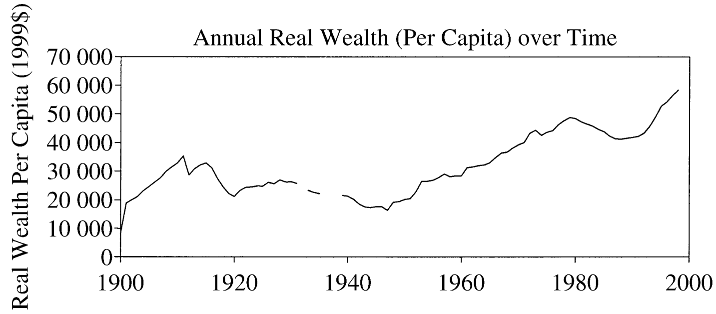

FIG. 3. Real wealth per capita (RWPC) in 1999 dollars in Dane County (1900–1998). Wealth was measured as the state assessed value of all property (personal property and real estate) from Dane County tax records.

validity of some of the assumptions underlying the Environmental Kuznets Curve hypothesis.

METHODS

Study area

Dane County, Wisconsin (population: 426 526) is a primarily agricultural but rapidly urbanizing area, with one large city (Madison, population 208 054) and several smaller surrounding cities (e.g., Sun Prairie, population ~20 000 and Mazomanie, population: ~1500). The 686-km^2 watershed of Lake Mendota is almost entirely contained within Dane County, Wisconsin, with a small portion in Columbia County (Fig. 2). The upland soils are well-drained silt loams, while the lowlands are poorly drained silts (Cline 1965). In 1995, the watershed land use was ~86% agricultural, 9% urban, 4% wetlands, and 1% forest (Soranno et al. 1996). The proportion of the watershed in agriculture has been at this level since about 1870 (Lathrop 1992*a*), but is currently changing through the conversion of agricultural land to urban uses (Dane County Regional Planning Commission 1992).

Ecological indicators

We used the following annual measures of ecological variables related to water quality and sediment deposition in Lake Mendota from previously assembled published and unpublished sources: mean annual dissolved reactive phosphorus (DRP), mean annual Secchi disk depth, and annual deposition rates of cadmium, chromium, copper, lead, and sulfur in lake sediments. DRP is the concentration of phosphorous (milligrams per liter of H_2O) in a sample taken from the lake surface. We used the average of all DRP samples taken during a given year for 45 years between 1926 and 1997 (Fig. 4a). DRP data were obtained from Lathrop et al. (1996) and from R. C. Lathrop (*unpublished data*) for the years 1994–1997. Detailed collection and analytical methods are available in Lathrop (1992*b*). Because phosphorus is a limiting nutrient in lakes, increased nutrients (such as DRP) in the water column result in greater productivity and potentially noxious algal blooms.

Greater algal growth also results in reductions in water clarity, which can be monitored using Secchi disk depth readings. Secchi disk depth is the depth at which a standard-sized disk is no longer visible when lowered beneath the lake surface. Secchi depth (in meters) was measured at multiple times over 73 noncontinuous years between 1900 and 1999 (Fig. 4b). Data from 1900–1993 were obtained from Lathrop et al. (1996). More recent data were obtained from the on-line North Temperate Lakes, Long Term Ecological Research database (NTL-LTER 2000). We pooled the data from 1900 to 1993, using the mean of five periods as available: spring turnover, early stratification, summer, destratification, and fall turnover, as defined by Lathrop et al. (1996). Data for all five time periods were not available for every year sampled.

Agriculture and construction in the watershed surrounding a lake can increase sediment inputs to lakes, which accumulate at the lake bottom. Sediment accumulation can also increase due to the sinking of the increased growth of vegetation and algae in the lake's water column caused by nutrient inputs. Scuba divers collected a sediment core from the University Bay basin (43°04.95′ N, 89°24.68′ W) of Lake Mendota in October 1995. The core was sectioned at 1-cm intervals over the upper 10 cm of the core and sectioned into 2-cm intervals in lower sections of the core. Sediments were weighed, freeze-dried, and reweighed to determine percent water and split for chemical composition analyses and sediment dating. All sediment sampling and laboratory analyses adhered to ultraclean sampling procedures (Patterson and Settle 1976, Hurley et al. 1996).

Fifteen 1–2-cm intervals were dated using the lead-210 radiometric method. Lead-210, a radioactive decay product of radium-226, may be used to accurately determine the rates of sediment and trace contaminant deposition (Charles and Hites 1987) for a core representing the recent 150–200 years of lake sediment deposition. Sediment accumulation rates over time were calculated using the constant rate of lead-210 supply model (Appleby and Oldfield 1984) and rates for non-dated sediment intervals were estimated by linear interpolation from the nearest dated intervals. Error in accumulation rates varies from 5% (one standard deviation) of reported values in most recent sediments to 20% in sediments deposited 100 years ago (S. King *unpublished data*).

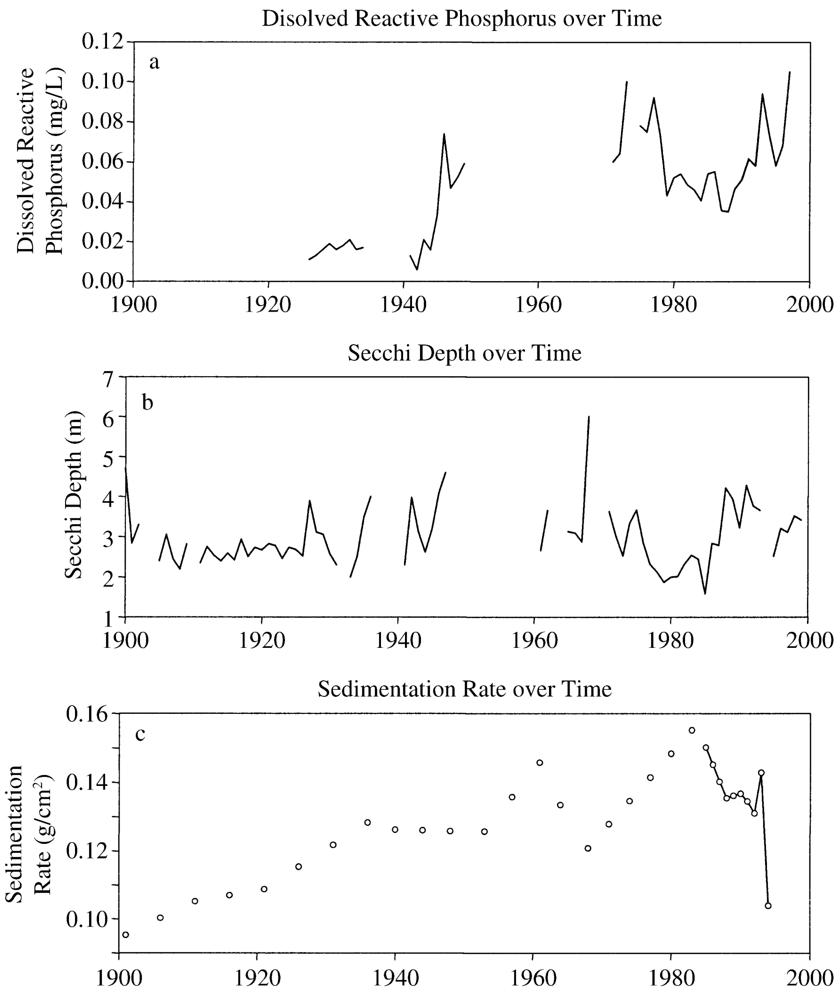

FIG. 4. (a) Dissolved reactive phosphorus concentrations in the water column of Lake Mendota (mg/L) (Lathrop et al. 1996; R. C. Lathrop, *unpublished data*). Annual means are presented; years with missing data are blank. (b) Secchi disk depth (m) in Lake Mendota, which indicates water clarity. Data are means from open water season. Data through 1993 were obtained and summarized from Lathrop et al. (1996). (c) Annual sedimentation rates (g sediment·cm^{-2}·yr^{-1}) in Lake Mendota. An in-depth discussion of trends for metals, sulfur, and sediment deposition rates data will be presented elsewhere (S. King, *unpublished manuscript*).

For each sediment core interval, chemical composition was determined by treating 20 mg of sediment with trace-metal-grade HF and HNO_3 and digesting with a modified microwave digestion technique (Eggiman and Betzer 1976, Walker 1994). Diluted samples were analyzed for lead (Pb), chromium (Cr), cadmium (Cd), copper (Cu), and sulfur (S) content with a Perkin-Elmer Plasma II ICP-OES (inductively coupled plasma/optical emission) spectrophotometer (Perkin-Elmer, Boston, Massachusetts, USA) using standard operating conditions as per the manufacturer's software. Metal deposition rates represent the mass concentration of metal (micrograms of metal per gram of sediment) multiplied by the annual sediment accumulation rate (grams of sediment per square centimeter per year) derived from dated sediments (S. King, *unpublished data*).

Economic indicators

Annual real wealth per capita (RWPC) was our economic indicator. We determined annual wealth using the State of Wisconsin's equalized values for total assessed real estate and personal property for Dane County from 1899 until 2000 (Fig. 3) (Wisconsin Department of Taxation Bulletin, as available, 1900–2000). To correct for inflation, wealth was converted to constant (1999) dollars by dividing annual wealth by a deflator factor of the Consumer Price Index (CPI) from 1900 to 2000 (R. C. Stahr, *unpublished data*). We accounted for changes in the population of Dane County by dividing real wealth by the annual population estimates to obtain RWPC. The annual population of Dane County was estimated by interpolation of the decadal census data from on-line sources (U.S. Census

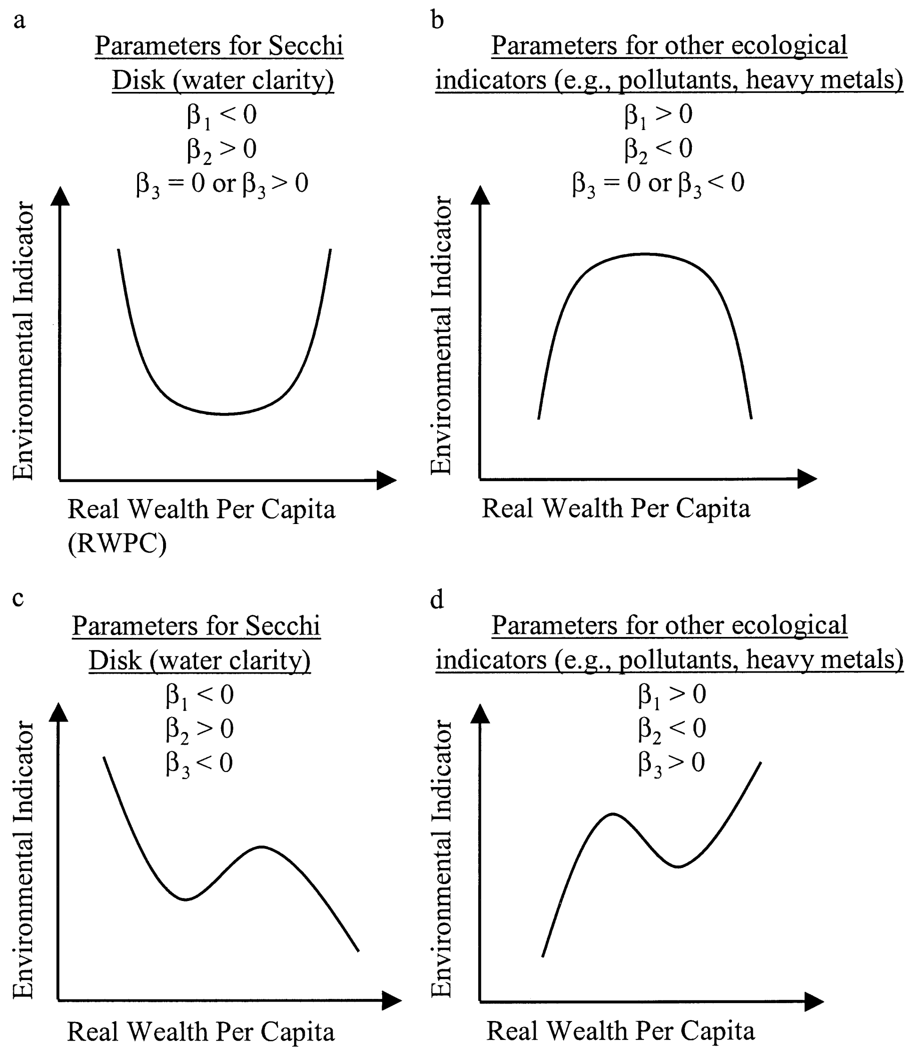

FIG. 5. Parameters expected for Model 2 that would provide evidence for an Environmental Kuznets Curve. The signs of the parameters (β_x) are different for ecological variables for which an *increase* in the value of the variable improves environmental quality (a) such as increasing Secchi disk depth (or water clarity) vs. those ecological variables for which an increase in the value of the variable would be considered a *decrease* in environmental quality (b) such as dissolved reactive phosphorus, cadmium, chromium, copper, lead, and sulfur. A possible downturn in the EKC is shown in (c) and (d) suggested by the significance of the $RWPC^3$ (quadratic term) in the models of ecological and economic indicators.

Bureau 2001). The period of record includes both economic growth and downturns (Fig. 3).

Previous EKC studies have used real gross domestic product per capita rather than RWPC. Regional estimates of real income per capita are not available for extended time periods in the United States, although estimates of wealth are. However, both are estimates of economic well-being and prosperity. Moreover, given that changes in income become capitalized into changes in this measure of wealth, either measure of economic prosperity is appropriate for testing the EKC hypothesis.

Statistical analyses

We examined the relationship between ecological indicators and RWPC using two general (and complementary) approaches. Our primary approach used ordinary least-squares regression to compare different models using an Akaike Information Criterion (AIC) (Hilborn and Mangel 1997). Explained in more detail below, this approach is useful for comparing among different models that contain different numbers of parameters and identifying the ''best'' or most appropriate model given the data. We then determined if the best model was of the form that suggested an EKC relationship based on the signs (±) of the parameter estimates for RWPC (Fig. 5a and b). If an EKC relationship was not the best fit for the data, we discussed which model was the most likely candidate. As a secondary source of information we examined plots of RWPC vs. the different ecological indicators to *qualitatively* evaluate general trends. These plots were particularly useful to

TABLE 1. Summary of dependent variables used in analyses.

Variable	N	Mean	1 SD	Minimum	Maximum
Secchi depth (m)	75	2.98	0.74	1.57	6.00
Dissolved reactive phosphorus (mg/L)	45	0.048	0.026	0.006	0.105
Cadmium ($\mu g \cdot cm^{-2} \cdot yr^{-1}$)	19	0.155	0.135	0.000	0.437
Chromium ($\mu g \cdot cm^{-2} \cdot yr^{-1}$)	19	3.96	0.56	3.11	5.20
Copper ($\mu g \cdot cm^{-2} \cdot yr^{-1}$)	19	2.44	0.83	0.87	4.35
Lead ($\mu g \cdot cm^{-2} \cdot yr^{-1}$)	19	7.71	5.40	0.22	17.77
Sulfur ($\mu g \cdot cm^{-2} \cdot yr^{-1}$)	19	978.2	284.6	439.4	1362.3

interpret the general magnitude and direction of the relationships for the variables whose statistical analyses were suspect due to small sample size.

Hypothesized models.—We compared three potential models using ordinary least-squares (OLS) regression in SAS version 8 (2000). We tested variants of the following model that relates environmental quality to RWPC:

$$E_t = \beta_0 + \beta_1 \mathrm{RWPC}_t + \beta_2 \mathrm{RWPC}_t^2 + \beta_3 \mathrm{RWPC}_t^3 + e_t$$

where E_t is an ecological variable at time t and β_0, β_1, β_2, and β_3 are constants for the intercept, linear, quadratic, and cubic forms of RWPC, and e_t represents normally distributed errors. We tested three contrasting models of the relationship between environmental and economic quality.

Model 1 (linear):

$$E_t = \beta_0 + \beta_1 \mathrm{RWPC}_t$$

Model 2 (quadratic):

$$E_t = \beta_0 + \beta_1 \mathrm{RWPC}_t + \beta_2 \mathrm{RWPC}_t^2$$

Model 3 (cubic):

$$E_t = \beta_0 + \beta_1 \mathrm{RWPC}_t + \beta_2 \mathrm{RWPC}_t^2 + \beta_3 \mathrm{RWPC}_t^3.$$

Model 1 represents a linear relationship between the dependent ecological variable and RWPC. Model 2 represents the EKC hypothesis only if the parameter estimates for the β_x terms are of the appropriate sign (Fig. 5a and b, and explained further in *Support for the EKC hypothesis*). In the case of Model 3, if the sign of β_3 is opposite the sign of β_2, then an additional downturn or upturn on the curve is suggested (Fig. 5c and d).

Akaike information criterion (AIC).—For each ecological indicator, we determined which of the three hypothesized models was the most appropriate fit for the data using Akaike Information Criterion (AIC) scores for each model. AIC scores are based on information theory and rest on the assumption that while no particular model is true, the model with the smallest AIC value is the most appropriate (or best) choice of the models under consideration. AIC is calculated as follows:

$$\mathrm{AIC} = -2 \log[\mathcal{L}(\hat{\theta} \mid y)] + 2K$$

where $[\mathcal{L}(\hat{\theta} \mid y)]$ is the log likelihood and K = the number of parameters estimated in the model including the intercept and σ^2. AIC is based on the principle of parsimony, which attempts to use ''the smallest possible number of parameters for adequate representation of the data'' (Box and Jenkins 1970). The first part of the equation decreases as more parameters are added to the model, while the second term ($2K$) increases as more parameters are added to the model (Burnham and Anderson 1998). In this way, AIC corrects for over-fitting (i.e., the fact that a better fit can often be obtained merely by adding more parameters to a model fit by OLS regression). This approach is particularly useful since the models compared in this study are nested. Stated simply, if we did not employ a technique to identify our best model that accounted for the different number of parameters in the candidate models, our analyses might inherently favor more parameter-rich models over simpler models.

Furthermore, we used a modified version of the equation above, a corrected AIC score ($\mathrm{AIC_c}$) that is recommended to account for small sample sizes (Burnham and Anderson 1998):

$$\mathrm{AIC_c} = -2 \log[\mathcal{L}(\hat{\theta} \mid y)] + 2K[n/(n - K - 1)]$$

where n = the sample size.

The ''best'' model for each ecological indicator was the model with the lowest $\mathrm{AIC_c}$ score (Burnham and Anderson 1998). The bigger the difference in $\mathrm{AIC_c}$ scores between the best model and other competing models, the less likely the other models were plausible (Burnham and Anderson 1998). Burnham and Anderson (1998) suggest that ''models for which $\Delta\mathrm{AIC} \leq 2$ have ''substantial support'' and should be considered further as candidate models. As such, we used the general rule that a fuller (more parameter-rich) model $\mathrm{AIC_c}$ must be at least two $\mathrm{AIC_c}$ points lower than a competing smaller (fewer parameter) model to warrant the use of more parameters.

We also determined the r^2 and the overall significance levels for each model (F test) as well as for each parameter (t test). Clearly, the identification of the best model using an $\mathrm{AIC_c}$ score could recommend a model that does not have a statistically significant F value for the overall model. However, the primary goal of this research is to determine if the EKC is the *most reasonable* relationship of the three models. Furthermore, the significance levels of several models and their parameters were influenced by the autocorrelation corrections (explained in more detail in *Correcting for autocorrelation*). Thus, while the non-AIC criteria (r^2,

TABLE 2. Results from ordinary least-squares regression analysis comparing Models 1, 2, and 3 (from *Methods*) to examine the relationship between real wealth per capita (RWPC) and several ecological indicators. For each indicator, the r^2 and overall significance levels (P) are shown for each of three different models. Following these are the parameter estimates and significance levels (P). The best candidate models as determined by AIC_c scores are shown in bold below and in Fig. 6.

Ecological indicator	AIC_c	r^2	P	β_1	P	β_2	P
Secchi (m; n = 63)							
Model 1	**134.067**	**0.17**	**0.0037**	$\mathbf{-4.305 \times 10^{-6}}$	**0.5819**		
Model 2	136.265	0.17	0.0103	-2.5431×10^{-5}	0.6406	3.02×10^{-10}	0.6950
Model 3	138.566	0.17	0.0233	-1.0628×10^{-4}	0.6414	2.656676×10^{-9}	0.6826
Dissolved reactive phosphorus (mg/L; n = 39)							
Model 1	**−207.691**	**0.63**	**<0.0001**	$\mathbf{3.307836 \times 10^{-7}}$	**0.2479**		
Model 2	**−206.979**	**0.65**	**<0.0001**	$\mathbf{-1.850 \times 10^{-6}}$	**0.2728**	$\mathbf{3.2273 \times 10^{-11}}$	**0.1914**
Model 3	−204.355	0.65	<0.0001	-5.20×10^{-6}	0.5466	1.30846×10^{-10}	0.5973
Cadmium (Cd) deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$; n = 19)							
Model 1	**−19.2274**	**0.09**	**0.2247**	$\mathbf{-4.289 \times 10^{-6}}$	**0.2247**		
Model 2	−16.3718	0.10	0.4134	-2.288×10^{-5}	0.4848	2.74142×10^{-10}	0.5670
Model 3	−13.1279	0.13	0.5462	9.248×10^{-5}	0.6202	-3.265×10^{-9}	0.5642
Chromium (Cr) deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$; n = 19)							
Model 1	**32.5296**	**0.19**	**0.1907**	$\mathbf{-2.659 \times 10^{-5}}$	**0.0616**		
Model 2	**33.6708**	**0.28**	**0.0755**	$\mathbf{1.3636 \times 10^{-4}}$	**0.2702**	$\mathbf{-2.403 \times 10^{-9}}$	**0.1887**
Model 3	37.3737	0.28	0.1688	2.7851×10^{-4}	0.6923	-6.76286×10^{-9}	0.7509
Copper (Cu) deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$; n = 19)							
Model 1	**45.8051**	**0.26**	**0.0255**	$\mathbf{4.61 \times 10^{-5}}$	**0.0255**		
Model 2	48.8021	0.27	0.0801	-3.689×10^{-5}	0.8383	1.224051×10^{-9}	0.6449
Model 3	49.8222	0.37	0.0684	-1.474×10^{-3}	0.1447	4.530998×10^{-8}	0.1393
Lead (Pb) deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$; n = 19)							
Model 1	**122.274**	**0.02**	**0.6053**	$\mathbf{-7.423 \times 10^{-5}}$	**0.6053**		
Model 2	124.884	0.05	0.6690	9.0041×10^{-4}	0.5038	-1.43704×10^{-8}	0.4673
Model 3	128.565	0.05	0.8396	0.002754	0.7224	-7.12246×10^{-8}	0.7615
Sulfur (S) deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$; n = 6)							
Model 1	72.009	0.87	0.0473	−0.0534	0.0224	−0.5543	0.3108
Model 2	**70.224**	**0.90**	**0.1436**	**0.3726**	**0.5503**	$\mathbf{-4.726 \times 10^{-6}}$	**0.5010**
Model 3	**69.942**	**0.91**	**0.4471**	**−5.7016**	**0.8869**	$\mathbf{1.31 \times 10^{-4}}$	**0.8837**

F and t test results) were useful in evaluating closely competing models (within two AIC_c points of each other), they were problematic for models requiring an autocorrelation correction. As a result, the AIC_c scores were used as the primary statistical tool to consistently compare models.

Support for the EKC hypothesis.—Once the best model for each ecological variable was identified, we assessed whether that best model supported an EKC for that variable. Support for the EKC hypothesis was indicated by two specific criteria: (1) Model 2 must have been identified by the AIC_c as the best (or one of the very closely competing) candidate models for that ecological variable, *and* (2) parameter estimates for the β_x terms of the model must be of the appropriate sign to create an EKC. The appropriate sign depends on whether an increase in the ecological variable represents an increase or decrease in environmental quality (Fig. 5a and b). For most of our environmental indicators, an increase in the indicator denotes increased pollution and thus, lower environmental quality (DRP, Cd, Cr, Cu, Pb, and S). For these variables, an EKC is suggested if $\beta_1 > 0$, $\beta_2 < 0$, $\beta_3 = 0$ (Fig. 5b). In the case of Secchi disk measurements, increasing Secchi depths are indicative of higher water clarity, and thus better water quality (see Fig. 5a). Thus, Model 2 suggests an EKC if $\beta_1 < 0$, $\beta_2 > 0$, $\beta_3 = 0$ for Secchi depth.

Correcting for autocorrelation.—Autocorrelation of the error terms is a common problem with regression analysis involving time series data. For all models, we used ''proc autoreg'' in SAS to test for positive and negative autocorrelation in the first five time lags using generalized Durbin-Watson statistics. If autocorrelation was detected, it was corrected using the standard technique of adding a lag of the dependent variable to the model as another independent variable (SAS 2000):

$$E_t = \beta_0 + E_{t-1} + \beta_1 \mathrm{RWPC}_t + \beta_2 \mathrm{RWPC}_t^2 + \beta_3 \mathrm{RWPC}_t^3.$$

For the models containing the lagged dependent variable, the more appropriate Durbin-h and Durbin-t tests were used to test for autocorrelated errors (SAS 2000).

Correcting for autocorrelation also introduced new caveats to data interpretation. While necessary, the addition of the lagged term of the dependent variable was generally found to have four major impacts on our results: (1) it decreased the sample size for the response variable (as data points could not be analyzed if lag_{t-1}

TABLE 2. Extended.

β_3	P	lag$_1$	P
		0.46914	**0.0012**
		0.46138	0.0017
-2.1513×10^{-14}	0.7150	0.45035	0.0028
		0.72952	**<0.0001**
		0.66547	**0.0002**
-8.7741×10^{-16}	0.6917	0.63169	0.0013
3.474×10^{-14}	0.5309		
4.28103	0.8371		
-4.3285×10^{-13}	0.1481		
5.58217×10^{-13}	0.8078		
−0.5074	**0.4069**		
$\mathbf{-1.004 \times 10^{-9}}$	**0.8796**	**−0.4453**	**0.6591**

did not exist); (2) it inflated the model r^2 values; (3) it increased the statistical significance of some models; and (4) it decreased the significance of parameter estimates for wealth within the models. However, the signs (±) of the parameter estimates were unchanged due to the addition of the lagged parameter, so interpretation of the signs of the parameter estimates (as well as the AIC$_c$ scores) were considered more robust than interpretation of significance levels and r^2 values. Primary consideration of AIC$_c$ scores and the signs of parameter estimates allowed us to maintain consistency in our evaluation criteria among different ecological indicators, as not all indicators required an autocorrelation correction.

Autocorrelation corrections were needed for Secchi depth and dissolved reactive phosphorus, the variables with the longest time series, as well as for sulfur deposition. In the case of sulfur deposition, the reduction in sample size to $n = 6$ greatly reduced our ability to test our hypotheses. For DRP and Secchi, adding the autocorrelation correction always increased the r^2 of the model and reduced the significance levels of the parameter estimates for the RWPC$_x$ variables.

RESULTS

Statistically significant relationships were obtained for Secchi depth, DRP, copper deposition, and sulfur deposition models. A model of marginal significance ($P = 0.0755$) was obtained for chromium deposition (Table 2). There were no significant models for cadmium or lead. Also, due to the autocorrelation correction and resulting small sample size for sulfur ($n = 6$) the results for sulfur are tenuous; nevertheless, we report them for context.

According to the best model as determined by AIC$_c$ scores, water quality decreases with increasing wealth, whether measured in terms of Secchi depth or DRP. Secchi depth (water clarity) decreases linearly with increasing wealth, as indicated by the sign of β_1 (Table 2) as well as Fig. 6a. In contrast to water clarity, increases in DRP are indicative of *decreased* water quality, and DRP increases either linearly or exponentially with real wealth (Table 2, Fig. 6b and c). Both Secchi depth and DRP models included a lagged dependent variable term. The best model for DRP explained 63% of the variability in DRP. While the results are not shown here, the linear model for DRP and the parameter estimate for RWPC were both highly significant before the lagged dependent variable was added to the model. The best model for Secchi depth only explained 17% of the variability, and while not shown here, the F value for this model was not significant before the addition of the lagged term. Our data do not support an EKC as the most likely relationship between RWPC and Secchi depth or DRP.

The best models for cadmium, chromium, and lead suggest a linear relationship with RWPC as determined by the AIC$_c$ scores. Plots (Fig. 6d,e,f,h) and the signs of β_1 (Table 2) indicate a negative relationship for all three variables. None of these models had significant F values, however, so the trend of increasing environmental quality with wealth is not statistically significant. Furthermore, Models 1 and 2 for chromium were indistinguishable (with AIC$_c$ scores within two points of each other) and should both be considered good candidate models. Parsimony would suggest the linear model most appropriate. However, unlike Model 1 ($P = 0.1907$), Model 2 for chromium was marginally significant ($P = 0.0755$), with the parameter estimates suggesting an EKC relationship (Table 2). This model suggests chromium initially increases with wealth ($\beta_1 > 0$), but then decreases ($\beta_2 < 0$) at higher values of wealth around ~$30 000 (see Fig. 6). The EKC response model (Model 2 with appropriate signs for the parameters) also explains more of the variability ($r^2 = 0.28$) than the linear model ($r^2 = 0.19$). Thus, for chromium, an EKC may be a weak possibility.

The results for heavy metals cadmium, chromium, and lead contrast with those of copper and sulfur. The best models show copper increasing linearly with increasing wealth (Fig. 6g), with a positive parameter

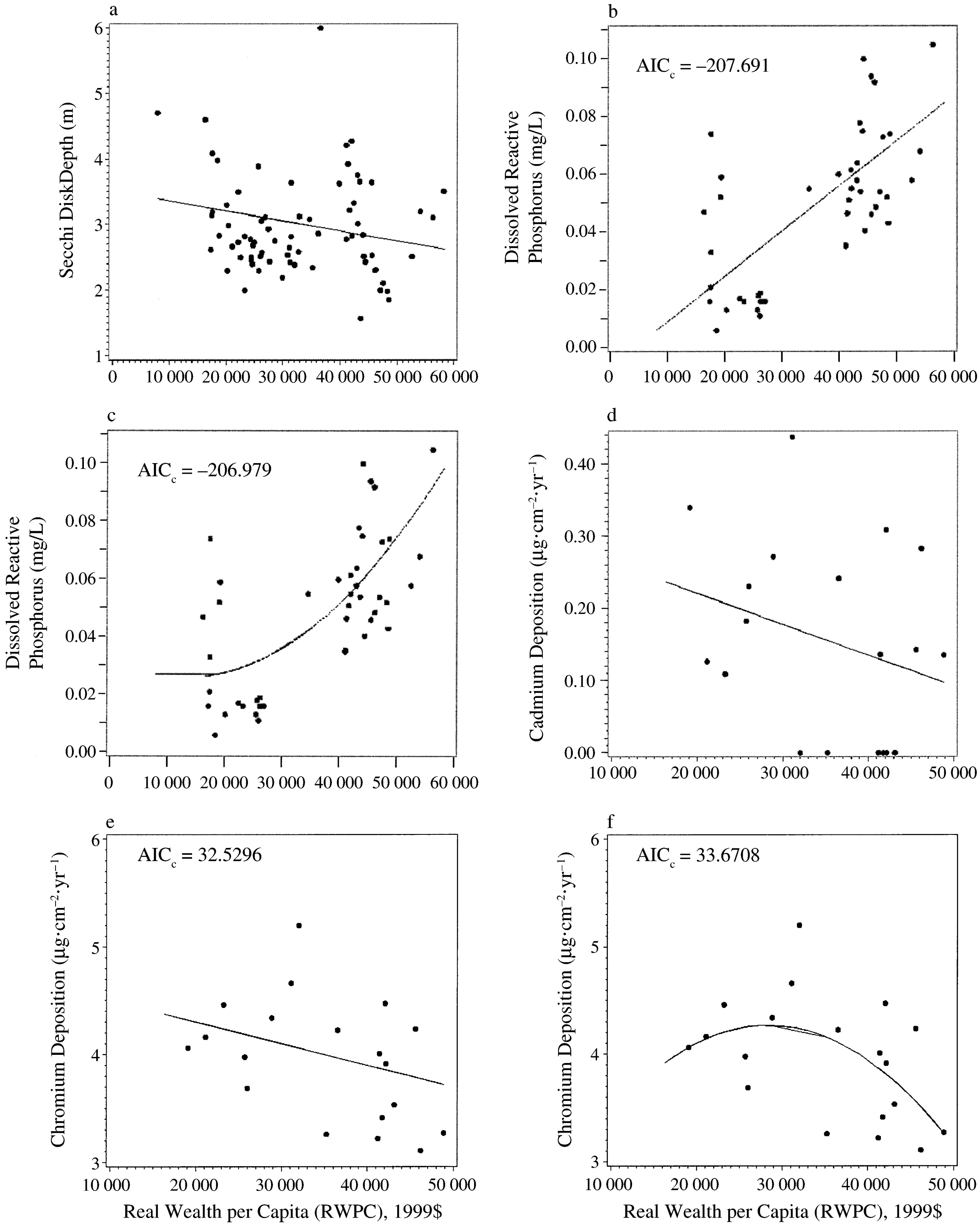

FIG. 6. Ecological variables from Lake Mendota (∼1900–2000) related to RWPC in Dane County, Wisconsin. Each plot shows the data and model for the best model as determined from the Akaike Information Criterion scores (AIC$_c$) used for model comparisons. When the best model was not more than two points lower than the next competitive model, both competing models and AIC$_c$ scores are shown (e.g., as in panels b and c). (a) Secchi disk depth (m), a measure of water clarity. An increase in Secchi depth indicates clearer lake water, which is often indicative of better water quality. The final model results shown in Table 2 also include the lag of Secchi disk depth to correct for serial autocorrelated errors. The model shown here

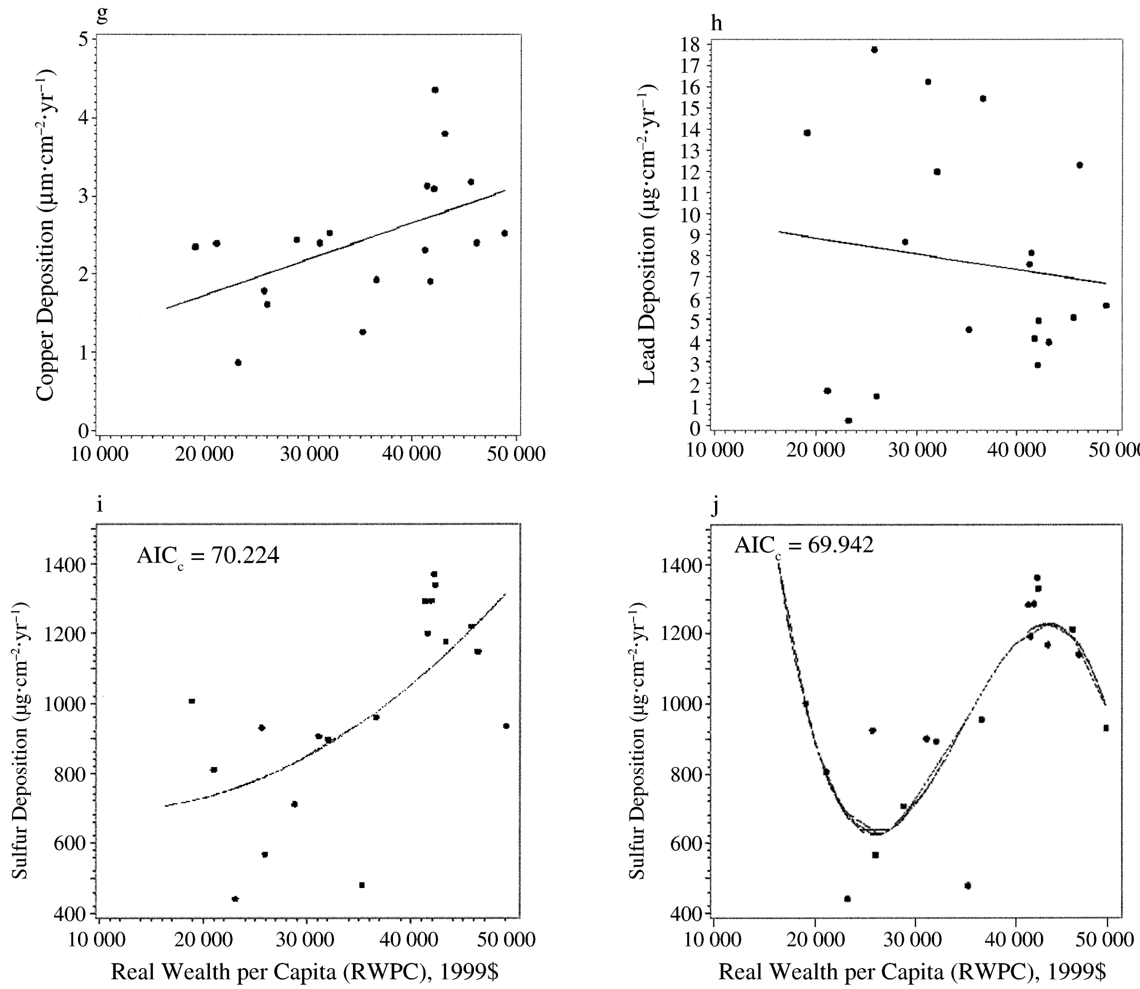

FIG. 6. Continued.

estimate for β_1 (Table 2). This model also has a significant overall F value, while Models 2 and 3 are marginally significant, $P = 0.08$ and 0.07, respectively (Table 2). The best model (Model 1: linear) explained 26% of the variability in copper. This model for copper is one of the most statistically rigorous relationships we found; not only was it determined to be the most likely model by the AIC_c, but this model also had a significant F value and significant parameter estimates (β_1) (Table 2). Although the results are not shown here, a model with only $RWPC^2$ had a slightly lower AIC_c value than even the linear model, and supported the same general trend of increasing copper with increasing wealth.

Correcting for temporal autocorrelation in the sulfur model greatly reduced our sample size ($n = 6$), rendering the statistical results and any interpretation for sulfur highly suspect. The AIC_c scores suggest Model 3 as the best model (Fig. 6i and j). However, the second best model (Model 2) is within only two points of the

←

does not include this term. (b, c) Dissolved reactive phosphorus (mg/L), a common constituent of non-point-source pollution. The final model results shown in Table 2 include the lag of DRP to correct for serial autocorrelated errors. The model shown here does not include this term. (d) Cadmium deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$). Cadmium, as well as the pollutants in (e) and (f), is a heavy metal produced by industrial activities, burning of fossil fuels, and urban runoff. (e, f) Chromium deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$). (g) Copper deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$). (h) Lead deposition ($\mu g \cdot cm^{-2} \cdot yr^{-1}$). (i, j) Sulfur deposition ($\mu g \cdot cm^{-2} yr^{-1}$). The final model results shown in Table 2 also include the lag of sulfur deposition to correct for serial autocorrelated errors. The model shown here does not include this term. Results for sulfur are extremely problematic due to low sample size when the lagged dependent variable is used in the regression equation to correct autocorrelation problems.

best model, and should be considered equally valid, suggesting sulfur deposition is exponentially related to RWPC. However, the only model with a significant *F* value is the linear one (Model 1). Our low sample size does not allow us to distinguish different models for sulfur. However, the plots and the signs of parameter estimates for all three candidates suggest that over most of the range in wealth examined here, sulfur deposition increases, but may hint at a downturn at the highest levels of wealth.

Discussion

Our analysis indicates that different measures of environmental quality show different relationships with wealth, and suggests that the EKC is not a universal theory describing the relationship between all ecological and economic indicators. Most of our variables did not show definitive support for an EKC relationship. However, several of our variables were related to RWPC to varying degrees. When considering water clarity (Secchi disk depth) and dissolved reactive phosphorous (DRP) levels, environmental quality most likely decreases with increasing wealth. The best models and plots for cadmium, chromium, and lead suggest improvements in environmental quality with increases in real wealth per capita as compared to other models, although the trends are not statistically significant, indicating the relationship is extremely weak. A clear relationship was established between copper and RWPC, suggesting more pollution (and thus lower environmental quality) with increased wealth. Overall, our strongest (most statistically rigorous) results for Secchi depth, DRP, and Cu show *increasing* pollution in Lake Mendota with increasing wealth in Dane County. Overall, wealth did not explain much of the variability in our ecological indicators.

We hypothesized that biophysical characteristics of a pollutant (such as its biogeochemical properties) could influence the likelihood of finding an EKC relationship. More specifically, we hypothesized that an EKC was more likely for pollutants whose impacts are reversible, as the EKC hypothesis depends on the reversibility of environmental problems. It is unlikely that the assumption of reversibility for excessive P pollution is always reasonable for lakes, because time lags associated with phosphorus cycling make it less likely to respond quickly to mitigation. Phosphorus can be recycled to algae in the water column by resuspension and digenesis of lake bottom sediments, particularly during oxygen deficits. Recycling of phosphorus can continue eutrophic conditions even after external inputs are reduced (National Research Council 1992). The recycling of sulfur is also similar to that of phosphorus as conditions of anoxia in the sediment result in the reduction of sulfate ions into reduced sulfur forms (i.e., HS^-) in the water column (Stumm and Morgan 1995).

In addition, large amounts of P are bound to soils in the uplands surrounding Lake Mendota, as in many other fertilized watersheds (Bennett et al. 1999). This sediment-bound P persists as a long-term source after other P inputs (such as fertilizer application) are reduced, resulting in considerable time lags between P reduction efforts and improvements in lakes (Bennett et al. 1999, Reed-Andersen 1999). Such time lags could mask any relationship between wealth and environmental pollutants. Furthermore, even if the effects of P pollution could hypothetically be reversed, determining in which particular lakes eutrophication is reversible is problematic. For example, Lake Mendota is one of the world's most studied lakes, benefiting from consistent long-term P measurements. Theoretical eutrophication models suggest only a ~58% probability that the lake is reversible with P reductions alone (Carpenter et al. 1999).

Non-point pollution is also diffuse and thus hard to measure and regulate (Novotny and Olem 1994), making mitigation a challenge. While economic loses due to non-point pollution are substantial (Postel and Carpenter 1997, Wilson and Carpenter 1999) economic benefits to industries causing pollution result in economic trade-offs between action and inaction (Carpenter et al. 1999). However, significant effort has been expended to combat non-point P pollution in the Lake Mendota watershed, including reduction of sewage inputs (1970s) (Lathrop et al. 1998), a non-point pollution control program (initiated in the 1980s) (Lathrop et al. 1998), and food web manipulations (Lathrop et al. 2002). Thus, we suggest that time lags in P cycling and challenges in monitoring non-point-source pollution are more plausible explanations for why an EKC is not observed than the idea that mitigation has not been attempted.

Secchi disk depth also has several features that may make it less likely to exhibit an EKC. Water clarity does not have direct impacts on human health, although it does impact property values (Wilson and Carpenter 1999) and enjoyment of the lake for fishing, swimming, and other recreational uses. While Secchi depth is influenced by non-point-source pollution from nutrients such as phosphorus, some water bodies may naturally have shallow Secchi depths due to other chemical constituents (e.g., dissolved organic carbon) and grazing by zooplankton (Lathrop et al. 1999). Thus, when considering an EKC for water clarity in other lakes, it is important to note that shallow Secchi depths can also occur as a result of nonanthropogenic causes.

Phosphorus and Secchi disk depth provide reasonable examples of our hypothesis that an EKC is less likely for ecological variables that are difficult to mitigate or reverse over shorter time scales. DRP was positively related to RWPC, suggesting a decrease in environmental quality with increasing wealth. As previously discussed, remediation of P pollution may exhibit significant time lags (making it less immediately reversible), and is not directly related to human health. Secchi depth, an indicator of water clarity, decreased with increasing RWPC, also suggesting environmental quality worsened with increasing RWPC. The relationship of wealth to P and Secchi depth is consistent with our hypothesis that given their indirect relationship with human health and biophysical characteristics, they may be less likely to exhibit an EKC.

We also hypothesized that potential human health impacts of pollutants could affect whether an EKC relationship exists, working under the econometric view that incentives and motivation for mitigation are greater for pollutants that impact human health than for those that do not. Trace metals were the type of indicator we expected to be more likely to show evidence of a Kuznets curve, as heavy metal pollution can seriously impact human health (World Health Organization 1977, 1988, 1992, 1998). Historically, heavy metal inputs to Lake Mendota were produced from industrial point sources. Herbicides used to control algal growth (copper sulfate applications directly to water) were used decades ago in large nearby lakes (Lathrop and Johnson 1979, Lathrop et al. 1992). Currently, the primary source of many metals to Lake Mendota are non-point sources such as urban stormwater and atmospheric deposition. Sources of metals in stormwater are parking lots and roads with high traffic densities (from mechanical wearing of automobile braking components), and metal roofs (Bannerman et al. 1993, Makepeace et al. 1995). The effects of trace metals tend to remain localized since they generally are not recycled in the water column. Instead, trace metals are bound to particles and are deposited in lake sediments fairly rapidly (Stumm and Morgan 1995).

Although not definitive, the most probable patterns for Cd, Cr, and Pb are declining trends with increasing wealth over the time period considered here (as suggested by the AIC_c scores and a qualitative assessment of the plots of these variables). These patterns were not statistically significant, however. It should also be noted that the results for cadmium are less reliable than for other metals since its levels were near method detection limits. Reductions in these heavy metals may be reasonable when viewed in light of establishment of industrial discharge regulations and reductions in the use of leaded gasoline (e.g., Federal Water Pollution Control Act 1972, Clean Water Act 1977). If this decrease is legitimate, we suggest that it would also be in general agreement with the notion of an Environmental Kuznets Curve, assuming our analysis missed the initial increases in these metals. Our analysis certainly missed initial low levels of Pb and Cr (circa 1820) (Iskandar and Keeney 1974) and also likely missed the increases due to early industrial activities (1850–1900, S. King, *unpublished data*) by only analyzing sediment core data after 1900 when RWPC data were available. This lends somewhat more support for an EKC relationship for Cd, Cr, and Pb. The potential for missing part of the curve is a shortcoming of nearly any EKC study that examines long-term data instead of panel data; and the work presented here is one of the longest in duration. Although highly speculative, our data for Cd, Cr, and Pb weakly suggest that an EKC is more likely for variables that severely impact human health.

The metal that deviated from the general EKC prediction was copper, where we saw a definitive (statistically significant) increase with RWPC. Copper is often the major aquatic toxic metal in stormwater (Makepeace et al. 1995) and is a component of algacides still in use in lakes and golf courses in Dane county; however, less algacide has been used in Lake Mendota than in other nearby lakes (Lathrop and Johnson 1979, Lathrop et al. 1992). Other sources of copper include building materials and automobile components (Bannerman et al. 1993, Makepeace et al. 1995). Bannerman et al. (1993) collected stormwater runoff from residential, commercial, and industrial areas around Lake Mendota (including roofs, parking lots, and streets). Their data generally showed Cu levels to be greater than or equal to Pb, and greater than Cr and Cd (Bannerman et al. 1993). Thus, the trend for copper (opposite to that of the other metals we examined) may be because copper is more abundant in stormwater than the other metals. In addition, trends for copper may differ from the other metals due to its greater solubility as $CuCO_3$ in calcareous, aqueous systems (relative to the other metals). Also, as a micronutrient for algal growth, Cu is taken up and recycled in the lake through algal growth and decay (Stumm and Morgan 1995). Therefore, it may not deposit in lake sediments as quickly as the other metals we examined, although that is still its major longterm fate.

Lastly, while the results for sulfur are extremely problematic (low sample size, Fig. 6i and j), levels of sulfur also increased with increasing wealth across most of the range in wealth. Only Model 1 is statistically significant, which would suggest a general linear increase in S with increasing wealth. However, an eventual downturn in S (and thus, an EKC) may be hinted at by the AIC_c scores. The results for sulfur are certainly problematic and not definitive. Like P, S dynamics are influenced by recycling of S within the water column that may create time lags in the response to any reductions in inputs from the watershed.

Our results provide an opportunity to examine some of the conditions under which we might expect an ecological variable to be related to wealth and to examine the validity of some of the assumptions underlying the EKC hypothesis. Our most statistically rigorous results (Secchi, P, and Cu) showed pollution *increasing* with wealth in Dane County, Wisconsin, contrary to the general notion of an EKC. Our findings suggest support for the idea that an EKC is less likely for variables whose impacts are not immediately reversible, as it is likely that the two indicators of nutrient status (DRP and water clarity) are influenced by the time lags in P biogeochemistry, rendering them less immediately reversible problems. Our results may suggest limited support for the idea that the EKC is more likely for some, but not all, pollutants linked directly to severe human health impacts. Plots of several trace metals (Cd, Cr, and Pb) vs. wealth all showed weak decreasing trends with increasing wealth (albeit not statistically significant), and chromium showed very tentative evidence of an EKC. Copper, on the other hand, increased with RWPC, countering our hypothesis for heavy metals.

Finally, we only considered seven ecological variables in one particular watershed, and thus, these patterns must be evaluated further elsewhere to determine their broader applicability. However, our results do not show unequivocal support for an EKC, and suggest that economic prosperity cannot always be expected to lead to improvements in environmental quality. Furthermore, the low r^2 values and lack of statistical significance in many of our models suggest that, in general, wealth is not a strong predictor of environmental quality as defined here. Other factors are likely to explain more of the variability in our ecological indicators than wealth.

The presumed existence of an Environmental Kuznets Curve has been used to justify managing the environment by creating opportunities for growth. While some of the economic literature suggests that EKCs exist for some pollutants, the difficulty lies in understanding under what conditions they would or would not exist. We and other authors have found that the EKC exists only in very special conditions, with certain economic indicators and certain ecological indicators (Rothman and de Bruyn 1998). A growing literature also suggests specific causes of pollution reduction not related to economic growth (Arrow et al. 1995): the 1973 oil crisis, energy prices, trade, and population density (Moomaw and Unruh 1997, Panayotou 1997, Argas and Chapman 1999). Our results also suggest that factors other than wealth are needed to explain the dynamics of several pollutants. Identifying factors other than wealth that influence environmental quality may be complex but is important in understanding pollution reduction efforts. Policy-makers should be careful making assumptions that economic growth will necessarily lead to environmental improvements.

Acknowledgments

This work was funded by National Science Foundation IGERT Grant Number DGE-9870703–Interdisciplinary Graduate Education and Research (IGERT) program at University of Wisconsin–Madison in Social and Aquatic interactions as well as the National Center for Ecological Analysis and Synthesis, at the University of California, Santa Barbara. Additional support for E. M. Bennett and B. K. Greenfield was provided by the Center for Limnology, University of Wisconsin-Madison, and for S. E. Gergel and B. K. Greenfield by the Environmental Protection Agency–STAR Fellowship. Steve Carpenter and Pete Nowak deserve special thanks for their support of and advice on this project, as does Dick Lathrop for generously providing both phosphorus data and useful reviews of this work. Initial discussions with Debbie Davidson (and other IGERT participants too numerous to name) were essential in the development of this project. We also thank Beth Harper for her library assistance and Arik Levinson for his perspective on earlier drafts of this paper. We also thank Jim Hurley and Paul Garrison from the Wisconsin Department of Natural Resources for help obtaining lake sediment cores. The comments of two anonymous reviewers helped improve this manuscript greatly.

Literature Cited

Appleby, P. G., and F. Oldfield. 1984. Empirical testing of Pb-210 dating models for lake sediments. Pages 93–24 *in* E. Y. Haworth and J. W. G. Lund, editors. Lake sediments and environmental history: studies in palaeolimnology and palaeoecology in honour of Winifred Tutin. University of Minnesota Press, Minneapolis, Minnesota, USA.

Argas, J., and D. Chapman. 1999. A dynamic approach to the environmental Kuznets curve hypothesis. Ecological Economics **28**:267–277.

Arrow, K., B. Bolin, R. Constanza, P. Dasgupta, C. Folke, C. S. Holling, B. O. Jansson, S. Levin, K. G. Maler, C. Perrings, and D. Pimentel. 1995. Economic growth, carrying capacity, and the environment. Science **268**:520–521.

Bannerman, R. T., D. W. Owens, R. B. Dodds, and N. J. Hornewer. 1993. Sources of pollutants in Wisconsin stormwater. Water Science and Technology **28**(3–5):241–259.

Barbier, E. B. 1997. Introduction to the environmental Kuznets curve special issue. Environment and Development Economics **2**:357–367.

Beckerman, W. 1992. Economic growth and the environment. Whose growth? Whose environment? World Development **20**:481–496.

Benjamin, M. M. and B. D. Honeyman. 1992. Trace Metals. Pages 317–352 *in* S. S. Butcher, R. J. Charlson, G. H. Orians, and G. V. Wolfe, editors. Global Biogeochemical Cycles. Academic Press, London, UK.

Bennett, E. M., T. Reed-Andersen, J. N. Houser, J. G. Gabriel, and S. R. Carpenter. 1999. A phosphorus budget for the Lake Mendota watershed. Ecosystems **2**:69–75.

Bhattarai, M., and M. Hammig. 2001. Institutions and the environmental Kuznets curve for deforestation: a cross-country analysis for Latin America, Africa, and Asia. World Development **26**(6):995–1010.

Bohara, A. K., R. P. Betrrens, K. Gawande, and P. G. Wang. 2001. One size doesn't fit all: estimating environmental Kuznets curves for hazardous waste site counts. Environmetrics **12**(4):383–394.

Box, G. E. P., and G. M. Jenkins. 1970. Time series analysis: forecasting and control. Holden-Day, London, UK.

Brock, T. D. 1985. Chemistry and nutrient loading. Pages 43–83 *in* A eutrophic lake: Lake Mendota, WI. Springer-Verlag, New York, New York, USA.

Burnham, K. P., and D. R. Anderson. 1998. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, New York, USA.

Carpenter, S. R., D. Ludwig, and W. A. Brock. 1999. Management of eutrophication for lakes subject to potentially irreversible change. Ecological Applications **9**:751–771.

Charles, M. J., and R. A. Hites. 1987. Sediments as archives of environmental pollution trends. *In* R. A. Hites and S. J. Eisenreich, editors. Sources and fates of aquatic pollutants. American Chemical Society, Washington, D.C. USA.

Cline, D. R. 1965. Geology and groundwater resources of Dane County, Wisconsin. Water Supply Paper 1779-U, U.S. Geological Survey, Washington, D.C., USA.

Cole, M. A., A. J. Rayner, and J. M. Bates. 1997. The environmental Kuznets curve: an empirical analysis. Environment and Development Economics **2**:401–416.

Dane County Regional Planning Commission. 1998. 1997 regional trends: Dane County, Wisconsin. [Online: ⟨http://www.co.dane.wi.us/rpc/97trends/97trends.htm⟩.]

De Bruyn, S. 1997. Explaining the environmental Kuznets curve: structural change and international agreements in reducing sulfur emissions. Environment and Development Economics **2**:485–503.

De Bruyn, S. M., J. C. J. M. van den Bergh, and J. B. Opschoor. 1998. Economic growth and emissions: reconsidering the empirical bases of environmental Kuznets curves. Ecological Economics **25**:161–175.

De Luca, S. J., L. B. Milano, and C. N. Ide. 1991. Rain and urban stormwater quality. Water Science and Research **23**(Kyoto):133–140.

Eggiman, D. W., and P. R. Betzer. 1976. Decomposition and analysis of refractory oceanic suspended materials. Analytical Chemistry **48**(6):886–890.

Ekins, P. 1997. The Kuznets curve for the environment and economic growth: examining the evidence. Environmental Planning A **29**:805–830.

Gawande, K., R. P. Berrens, and A. K. Bohara. 2001. A consumption-based theory of the environmental Kuznets curve. Ecological Economics **37**(1):101–112.

Grossman, G. M., and A. B. Krueger. 1993. Environmental impacts of a North American Free Trade Agreement. *In* P. Garber, editor. The U.S.–Mexico Free Trade Agreement. MIT Press, Cambridge, Massachusetts, USA.

Hilborn, R., and M. Mangel. 1997. The ecological detective: confronting models with data. Princeton University Press. Princeton, New Jersey, USA.

Hurley, J. P., M. M. Shafer, S. E. Cowell, D. E. Armstrong, J. T. Overdier, and P. E. Hughes. 1996. Trace metal assessment of Lake Michigan tributaries using low-level techniques. Environmental Science and Technology **30**:2093–2098.

Iskandar, I. K., and D. R. Keeney. 1974. Concentration of heavy metals in sediment cores from selected Wisconsin lakes. Environmental Science and Technology **8**(2):165–170.

Kuznets, S. 1955. Economic growth and income inequality. American Economic Review **45**:1–28.

Lathrop, R. C. 1992*a*. Nutrient loadings, lake nutrients, and water clarity. Pages 97–126 *in* J. F. Kitchell, editor. Food web management: a case study of Lake Mendota, Wisconsin. Springer-Verlag, New York, New York, USA.

Lathrop, R. C. 1992*b*. Lake Mendota and the Yahara River chain. Pages 17–30 *in* J. F. Kitchell, editor. Food web management: a case study of Lake Mendota, Wisconsin. Springer-Verlag, New York, New York, USA.

Lathrop, R. C., S. R. Carpenter, and D. M. Robertson. 1999. Summer water clarity responses to phosphorus, *Daphnia* grazing, and internal mixing in Lake Mendota. Limnology Oceanogr. **44**:137–146.

Lathrop, R. C., S. R. Carpenter, and L. Rudstam. 1996. Water clarity in Lake Mendota since 1900: responses to differing levels of nutrients and herbivory. Canadian Journal of Fisheries and Aquatic Sciences **53**:2250–2261.

Lathrop, R. C., S. R. Carpenter, C. A. Stow, P. A. Soranno, and J. C. Panuska. 1998. Phosphorus loading reductions needed to control blue-green algal blooms in Lake Mendota. Can. J. Fish. Aquat. Sci. **55**:1169–1178.

Lathrop, R. C., B. M. Johnson, T. B. Johnson, M. T. Vogelsang, S. R. Carpenter, T. R. Hrabik, J. F. Kitchell, J. J. Magnuson, L. G. Rudstam, and R. S. Stewart. 2002. Stocking piscivores to improve fishing and water clarity: a synthesis of the Lake Mendota biomanipulation project. Freshwater Biology **47**:2410–2424.

Lathrop, R. C., and C. D. Johnson. 1979. Appendix B: water quality conditions. *In* Dane County water quality plan. Dane County Regional Planning Committee, Madison, Wisconsin, USA.

Lathrop, R. C., S. B. Nehls, C. L. Brynildson, and K. R. Plass. 1992. The fishery of the Yahara lakes. Technical Bulletin Number **181**. Wisconsin Department of Natural Resources, Madison, Wisconsin, USA.

Makepeace, D. K., D. W. Smith, and S. J. Stanley. 1995. Urban stormwater quality: summary of contaminant data. Critical Reviews in Environmental Science and Technology **25**(2):93–139.

McConnell, K. E. 1997. Income and the demand for environmental quality. Environment and Development Economics **2**:383–399.

Michael, H. J., K. J. Boyle, and R. Bouchard. 1996. Water quality affects property prices: a case study of selected Maine lakes. Maine Agricultural and Forest Experiment Station Report Number **398**. University of Maine, Orono, Maine, USA.

Moomaw, W. R., and G. C. Unruh. 1997. Are environmental Kuznets curves misleading us? The case of CO_2 emissions. Environment and Development Economics **2**:451–463.

Naidoo, R., and W. L. Adamowicz. 2001. Effects of economic prosperity on numbers of threatened species. Conservation Biology **15**(4):1021–1029.

National Research Council. 1992. Restoration of aquatic ecosystems: science technology and public policy. National Academy Press, Washington, D.C., USA.

Novotny, V., and H. Olem. 1994. Water quality: prevention, identification, and management of diffuse pollution. Van Nostrand Reinhold, New York, New York, USA.

NTL-LTER. 2000. North Temperate Lakes–Long-term Ecological Research Program, National Science Foundation, chemistry database. John J. Magnuson Center for Limnology, University of Wisconsin, Madison, Wisconsin, USA.

Panayotou, T. 1993. Empirical tests and policy analysis of environmental degradation at different stages of economic development. *In* Working paper: technology and environment programme. International Labour Office, Geneva, Switzerland.

Panayotou, T. 1997. Demystifying the environmental Kuznets curve: turning a black box into a policy tool. Environment and Development Economics **2**:465–484.

Patterson, C. C., and D. M. Settle. 1976. The reduction of orders of magnitude errors in lead analysis of biological materials and natural waters by evaluating and controlling the extent and sources of industrial lead contamination introduced during sample collection, handling and analysis. Pages 321–351 *in* D. La Fleur, editor. Accuracy in trace analysis: sampling, sample handling and analysis. U.S. National Bureau of Standards Special Publication **422**.

Postel, S., and S. R. Carpenter. 1997. Freshwater ecosystem services. Pages 195–214 *in* G. Daily, editor. Nature's services: societal dependence on natural ecosystems, Island Press, Washington, D.C., USA.

Reed-Andersen, T. 1999. Flows of phosphorus on an agricultural landscape: implications for eutrophication and restoration of Lake Mendota. Dissertation. University of Wisconsin, Madison, Wisconsin, USA.

Rothman, D. S., and S. M. de Bruyn. 1998. Probing into the environmental Kuznets curve hypothesis. Ecological Economics **25**:143–145.

SAS. 2000. SAS/ETS user's guide. Version 8. SAS Institute, Cary, North Carolina, USA.

Selden, T. M., and D. Song. 1994. Environmental quality and development: is there a Kuznets curve for air pollution estimates? Journal of Environmental Economic Management **27**:147–162.

Shafik, N., and S. Bandyopadhyay. 1992. Economic growth and environmental quality: time series and cross-country evidence. Background Paper for World Development Report. World Bank, Washington, D.C., USA.

Skonhoft, A., and H. Solem. 2001. Economic growth and land-use changes: the declining amount of wilderness land in Norway. Ecological Economics **37**(2):289–301.

Soranno, P. A., S. L. Hubler, S. R. Carpenter, and R. C. Lathrop. 1996. Phosphorus loads to surface waters: a simple model to account for spatial pattern of land use. Ecological Applications **6**:865–878.

Stahr, R. C. 2001. Political Science Department, Oregon State University, Corvallis, Oregon. [Online: ⟨http://www.orst.edu/Dept/pol_sci/fac/sahr/cv2000.pdf⟩.]

Steinnes, D. N. 1992. Measuring the economic value of water quality: the case of lakeshore land. Annals of Regional Science **26**:171–176.

Stern, D. I. 1998. Progress on the environmental Kuznets curve? Environment and Development Economics **3**:173–196.

Stumm, W., and J. J. Morgan. 1995. Aquatic chemistry: chemical equilibria and rates in natural waters. Third edition. John Wiley and Sons, New York, New York, USA.

U.S. Census Bureau. 2001. Decadal census data. [Available online: ⟨http://www.census.gov/population/cencounts/wi190090.txt⟩ and ⟨http://govinfo.library.orst.edu/cgi-bin/usaco-list?24–025.wic⟩.]

Vincent, J. R. 1997. Testing for environmental Kuznets curves within a developing country. Environment and Development Economics. **2**:417–431.

Walker, M. T. 1994. Forms of trace metals in Wisconsin streambeds. Thesis. University of Wisconsin, Water Chemistry Program, Madison, Wisconsin, USA.

Wilson, M. A., and S. R. Carpenter. 1999. Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecological Applications **9**:772–783.

Wisconsin Department of Taxation. 1900–2000. Wisconsin Department of Taxation Bulletin (as available) 1900–2000. Wisconsin Department of Taxation, Madison, Wisconsin, USA.

World Commission on Environment and Development. 1987. Our common future. Oxford University Press, Oxford, UK.

World Health Organization. 1977. Lead. International Program on Health and Safety. Environmental Health Criteria Monographs Number **3**.

World Health Organization. 1988. Chromium. International Program on Health and Safety. Environmental Health Criteria Monographs Number **61**.

World Health Organization. 1992. Cadmium. International Program on Health and Safety. Environmental Health Criteria Monographs Number **134**.

World Health Organization. 1998. Copper. International Program on Health and Safety. Environmental Health Criteria Monographs Number **200**.

Zaim, O., and F. Taskin. 2000. A Kuznets curve in environmental efficiency: an application on OECD countries. Environmental and Resource Economics **17**(1):21–36.