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Economic growth, biodiversity loss and conservation effort

Simon Dietz[a], W. Neil Adger[b,*]

[a]Department of Geography and Environment, London School of Economics, Houghton Street, London WC2A 2AE, UK
[b]Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, UK

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Abstract

This paper investigates the relationship between economic growth, biodiversity loss and efforts to conserve biodiversity using a combination of panel and cross section data. If economic growth is a cause of biodiversity loss through habitat transformation and other means, then we would expect an inverse relationship. But if higher levels of income are associated with increasing real demand for biodiversity conservation, then investment to protect remaining diversity should grow and the rate of biodiversity loss should slow with growth. Initially, economic growth and biodiversity loss are examined within the framework of the environmental Kuznets hypothesis. Biodiversity is represented by predicted species richness, generated for tropical terrestrial biodiversity using a species-area relationship. The environmental Kuznets hypothesis is investigated with reference to comparison of fixed and random effects models to allow the relationship to vary for each country. It is concluded that an environmental Kuznets curve between income and rates of loss of habitat and species does not exist in this case. The role of conservation effort in addressing environmental problems is examined through state protection of land and the regulation of trade in endangered species, two important means of biodiversity conservation. This analysis shows that the extent of government environmental policy increases with economic development. We argue that, although the data are problematic, the implications of these models is that conservation effort can only ever result in a partial deceleration of biodiversity decline partly because protected areas serve multiple functions and are not necessarily designated to protect biodiversity. Nevertheless institutional and policy response components of the income biodiversity relationship are important but are not well captured through cross-country regression analysis.

Keywords: Environmental Kuznets curve; Biodiversity; Species-area relationship; Conservation effort

1. Introduction

The relationship between economic growth and various indicators of environmental quality has come under increasing scrutiny because of widely observed manifestations of the consequences of unsustainable resource use at local and global scales. For certain indicators and certain statistical designs, the reduced-form relationship between income per capita and environmental quality is an inverted-U. Environmental degradation first increases with rising income, reaches a turning point, and then decreases. This has been termed the environmental Kuznets curve, though, as with Kuznets' (1955) own hypothesis on the relationship between income and inequality, the evidence that an environmental Kuznets curve applies generically is equivocal. Some environmental indicators do not yield an inverted-U relationship and researchers have often drawn contradictory conclusions for those that may (Ekins, 1997, Stern et al., 1996; Stern, 1998).

Global environmental parameters for which major irreversibilities are thought to exist, such as carbon dioxide and its role in climate change, have tended not to follow the environmental Kuznets curve (World Bank, 2000, Ravallion et al., 2000). There is reason to expect, as with other global problems, that it is not possible to 'grow out of' the problem of biodiversity decline. Indeed, current rates of species loss are many times the background rates of both extinction and speciation (Ehrlich and Wilson, 1991; Wilson, 1986; Reid, 1992; Smith et al., 1993). Biodiversity loss is also the product of a complex interaction of factors such as agricultural expansion and its protection, the product of equally complex effective demands for wilderness, recreation space and maintenance of genetic diversity for their use in agriculture and pharmaceuticals. The services and functions provided by biodiversity are ultimately undervalued by society within the context of the distribution of

* Corresponding author.
E-mail address: n.adger@uea.ac.uk (W.N. Adger).

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those benefits and the institutional context in which individual and collective decisions are made (Adger et al., 1995; Costanza et al., 1997; Turner et al., 1998; Pritchard et al., 2000). Developing a single indicator of biodiversity loss to test the reduced-form relationship with income over time is difficult.

We regress an output measure of the relationship—predicted species richness at the national level—and two input measures of conservation effort: protected area designation and implementation of international trade regulations. If predicted species richness were to yield an environmental Kuznets curve with per capita income, we would expect to see a U-shape. However, we hypothesise speciation cannot match the current rate of extinction, such that net biodiversity loss may slow at some level of income but will never be reversed. Conservation effort is hypothesised to increase with per capita income as governments respond to an increasing public demand for biodiversity, to a large extent the result of a positive and high income elasticity of demand for its aesthetic benefits (see for example the discussion in Kristrom and Reira, 1996). In the final section we discuss the implications of these findings in terms of the scale at which conservation effort takes place, and the ultimate causal factors in human overexploitation of the world's biological resources.

2. Biodiversity measures and development processes

2.1. Generating data for biodiversity

Quantifying species richness within the framework of the environmental Kuznets hypothesis places particular requirements on the data. The environmental Kuznets curve has been tested using either cross section data or panel data. From an econometric point of view, panel data are favoured because they allow a restrictive assumption manifest in cross-sectional analysis to be relaxed; the effect on the environment of income changes is the same for all countries. In statistical terms, this means the regression coefficients are common to all groups in the cross-section; one curve fits all. In this study, local conditions are likely to generate significant differences in the income environment relationship between countries such as Brazil and India. In particular, national species richness (aggregated across all taxa) in any given year will be in large part a function of natural habitat conditions, independent of anthropogenic impacts. This has the potential to confound the detection of human impacts on species richness. Therefore, commonality should not be assumed and only panel data will be considered.

But no direct panel data exist for species richness. The only data sets explicitly dealing with species give present day numbers of species or threatened species.[1] The disadvantage of the former has just been explained. The disadvantage of the latter is that it is an indicator of pressure on biodiversity, but not loss of biodiversity. More importantly, ecologists agree that only a fraction of all species have been taxonomically classified. Wilson (1986) suggests we do not even know the true number to the nearest order of magnitude. Thus, if the focus is on the loss of species from all taxa due to human impacts, the above indicators are inherently limited.

Attention must thus be focused on an indirect indicator of species richness. Estimates of biodiversity change have been made using the species-area relationship.[2] This relationship relates the number of species in a given area to the size of the area. An oft forgotten point is that, like the environmental Kuznets hypothesis, it is empirical and its biological significance must be inferred.[3] This study does not, however, demand a biological justification. A specific form of the equation has become established:

$$S = cA^z \tag{1}$$

where S is the number of species, A is area, c is a constant reflecting the density of species per unit and z is the slope of the relationship between S and A when S and A are expressed as logarithms. S is particularly sensitive to the magnitude of z, which varies by region, taxa and between island and (subsets of) continental flora and fauna. Research has generated a range of values of Z between 0.15 and 0.35 (MacArthur and Wilson, 1967). A median value of 0.25 is taken here, although it should be noted that Z may vary within ecosystem types because, for example, of edge or fragmentation effects that can determine different rates of extinction with marginal change in habitat area. Little, on the other hand, is known about c. Studies estimating the change in biodiversity eliminate c through the ratio of species in a given year (t) relative to a base year (0):

$$S_0 = cA_0^z \tag{2}$$

$$S_t = cA_t^z$$

$$S_t/S_0 = A_t/A_0^z$$

The simplicity of the relationship makes it attractive but introduces perhaps spurious certainty concerning the nature of change in this area. In addition, results vary by region, taxa and between continents and islands. Ultimately, the predictions of any one species-area curve will be accompanied by wide confidence intervals. Empirical examinations of the species-area relationship using the increasing levels of location-mapped species data, demonstrate that it is a reasonable approximation at broad scales (though other simple power relations have also been suggested), while all rule of thumb relations break down at finer resolution (Plotkin et al., 2000).

[1] The World Conservation Monitoring Centre (2000) and the World Resources Institute (1999) are two examples.

[2] Refer to Reid (1992) for a review of applications of the species-area relationship up to 1992.

[3] For a detailed discussion of the species-area relationship see Connor and McCoy (1979).

The next step is defining habitat area (A). Since global estimates of ecosystem types would involve increased uncertainty due to variations in z value by region (notably latitude), we restrict the study to one region and, specifically, the most biologically diverse. Species diversity is recognised to be highest in tropical rainforests (Myers, 1980; Wilson, 1986) and, in particular, in primary or undisturbed tracts. We focus on tropical forests yet the availability of a suitable data set even in this restricted area entails limitations.

The need for a time series confines this study to one data source; the FAO's Production Yearbook (FAO, various years). Unfortunately, this source adopts a broad definition of forests as 'all woody vegetations' (Koop and Tole, 1999) and thus prevents us from disaggregating between forest types. To mitigate for this, only 34 tropical countries identified as having 'tracts of tropical moist forest that are appreciable in size or are significant for their ecological and biotic values' (Myers, 1980) are included[4] (see Appendix A). Nevertheless, significant uncertainty still exists, because of inconsistencies in defining what constitutes a tropical moist forest but also in what constitutes deforestation and in how to measure it (Brown and Pearce, 1994). While confidence bands around the data are large, Allen and Barnes (1985) have demonstrated by rank correlation that the FAO data are sufficiently similar to other studies that only include tropical forests to permit their use in assessing tropical deforestation.

2.2. Preventing biodiversity loss

Biodiversity loss is the result of a complex set of proximate and underlying causes. The nature of land tenure and property rights and overexploitation of particular species for domestic and international trade have been the focus of conservation efforts in many countries and are of central importance in any discussion of global conservation. Although overexploitation of resources and biodiversity takes place under all types of property rights regime, clearly the security of ownership for private resources and the legitimacy of institutions for collective decision-making are key determinants of sustainability (Bohn and Deacon 2000; Adger, 2003; Agrawal, 2001). The state often takes tracts of land under its own jurisdiction to protect biodiversity on behalf of society. State protected land can take different forms: nature reserves protected for scientific interest, natural monuments protecting natural features and national parks that have some emphasis on recreation and access.

The state also changes the property rights in goods and services through regulation. The regulation of trade in endangered species is designed to prevent certain species, valued for their products, from being driven to extinction. The Convention on International Trade in Endangered Species (CITES) was first signed in 1973 by 21 states and presently has been adopted by 127 states. During this time, CITES has been viewed as the 'flagship of the flora and fauna preservation treaties' (Lanchberry, 1998, p. 69). Species covered by the convention are listed in one of three Appendices A, B and C. Trade in Appendix A species is essentially banned. Trade in Appendices B and C species is permitted but regulated through a system of permits. In principle, trade restrictions help to drive prices up and quantities down, thus reducing the threat of extinction. However, much depends on how effectively illegal trade is controlled. Furthermore, controlling trade can cut off valuable sources of revenue for many societies. Moran and Pearce (1997) identify the same problems for those cut off from the resources they depend on by state protected land. For them, these two strategies represent a 'moral view', which disinvests value in biodiversity, taking away its economic value.

Governments may be ineffective at protecting land if they lack the knowledge of how to use a resource properly or the funding to enforce policy (Bromley, 1997). Examples of conflicts between protected areas and local development priorities abound, where bureaucratic land protection is not matched by enforcement in the relevant areas, particularly, but not exclusively in developing countries (see reviews in Stoll-Kleemann, 2001; Smith et al., 1997). It is important to note neither international trade regulations nor state land protection necessarily represent the best means of protecting biodiversity. Nevertheless, they are widely practised and thus of great importance to biodiversity conservation globally.

3. Data and models

3.1. Predicted species richness and income

The environmental Kuznets hypothesis predicts environmental damage first increases and then decreases with rising income. The relationship normally resembles an inverted-U shape. However, referring to the nature of the environmental indicator used in this study (Eq. (2)), the relationship will be reversed and should resemble a U shape (Fig. 1). Yet can species richness be expected to exhibit this behaviour? The environmental Kuznets curve can be divided into two halves, the 'falling limb' and the 'rising limb' in Fig. 1. The 'falling limb' indicates decreasing numbers of species. The accelerated anthropogenic impact on species is already well documented (Ehrlich and Wilson, 1991; Wilson, 1986; Reid, 1992; Smith et al., 1993). However, species are not replenished at the same rate: background speciation would not yield a 'rising limb' of the same magnitude.

Instead, species richness might be expected to decrease and then *level off* with increasing income. As discussed above, theory suggests economic forces fuel the drive for environmental improvement, but species diversity cannot replenish itself at an equivalent rate. This suggests a hyperbolic curve, which combines the 'falling limb' of

[4] Myers classifies 40 countries in this group, but missing data for certain independent variables reduces the data set to 34 countries.

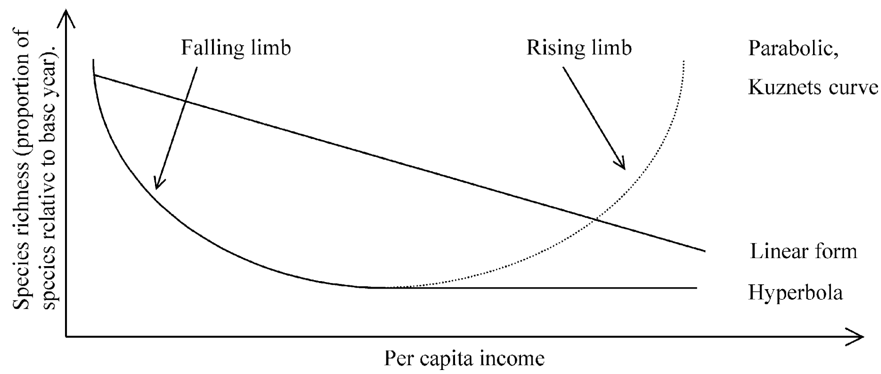

Fig. 1. Possible forms of the income biodiversity relationship.

the environmental Kuznets curve and a slowing of biodiversity loss, may be a more realistic representation of the relationship, as shown in Fig. 1. Inspection of the data for species richness also indicates a close clustering of observations around the base level, independent of income level (Fig. 2). In view of this, a linear equation is also tested.

Various techniques exist to estimate the curves. The three examined in this study are; ordinary least squares (OLS), fixed effects and random effects. Two methods are of particular interest to this study: fixed and random effects. Both build on the regression equation for OLS, which is the same for all countries. Fixed effect loosens up the assumption of commonality across countries by estimating a separate constant for each country (Koop and Tole, 1999). Random effects works in a similar way but assumes that international heterogeneity is randomly (and normally) distributed.

The difference between the two models, whether the vertical displacement of the regression equation should be parametric or random—has been debated in the literature (Greene, 1997). A fixed effects model implies international differences are generated by country-specific factors not covered by the regressors. The random effects specification, on the other hand, implies national peculiarities are unimportant and differences should be assumed random. This is a responsible approach if the sample is part of a much larger population (Greene, 1997). Yet this data set represents an almost complete set of countries containing significant tropical rainforests (Myers, 1980). Furthermore, intuition suggests environmental and economic factors should play a rather important role. We can discriminate between the three using LM statistics[5] and the Hausman test,[6] but, at this interjection, fixed effects are expected to be favoured.

To summarise, the variables and equations are as follows:

$S(i,t)$, the dependent variable, is the predicted species richness in any year compared to the reference year 1970 (Eq. (2)), multiplied by a factor of 1000 to prevent clustering: 1000 (A_t/A_0^z).

$G(i,t)$ is income per capita in log form.[7] Data comes from the Penn World Table.[8]

$C(i,t)$ is population change, expressed as a percentage of the previous year. Population is thought to be a determinant of the rate of tropical deforestation (e.g. Rudel, 1994). Data comes from the Penn World Table.

$P(i,t)$ is population density, expressed as people per hectare. The variable is a combination of FAO Production Yearbook data on land area and Penn World Table population statistics.

$T(t)$ is a linear time trend (T is simply the relevant year). This captures the time dependency of both income and biodiversity (current levels depend on previous levels).

$F(i,t)$ is forest area in hectares. The relative impact of deforestation in any country depends on absolute forest area and feeds back into future trends.

$D(i,t)$ is democracy, the sum of political rights and civil liberty indices taken from Freedom House data.[9] Both indices are based on a ranking system of 1–7, with 1 equalling most democracy. Here, the two indices are added and the ranking scale reversed, so that 2 corresponds to least democracy and 14 to most democracy (after Bhattarai and Hammig, 2001). Several researchers argue that democracy, or a vector of related institutional variables, are an important determinant of the rate of deforestation (Bhattarai and Hammig, 2001; Mather and Needle, 1999; Torras and Boyce, 1998; Barbier, 2001). The thinking behind this proxy is that as democracy grows so institutional forms promoting

[5] 'LM statistics' is the standard Lagrange Multiplier test for fixed/random effects over the basic model. It analyses whether the variance of $\mu(i)$ is equal to zero—i.e. whether fixed/random effects are constant, in which case simple OLS is valid. High values favour fixed/random effects.

[6] The Hausman test is an empirical test used to differentiate between random effects and fixed effects. It assesses whether individual effects are correlated with the regressors. If so, random effects are inconsistent. If not, both random and fixed effects are consistent but random effects are more efficient. High values favour fixed effects.

[7] A semi-log form like this is recommended when y is increasing slower or faster than x.

[8] Summers and Heston (1991) explain the methodology and layout of mark 5 of the table. The most up-to-date version, 5.6, is available at http://datacentre2.chass.utoronto.ca/pwt/

[9] www.freedomhouse.org

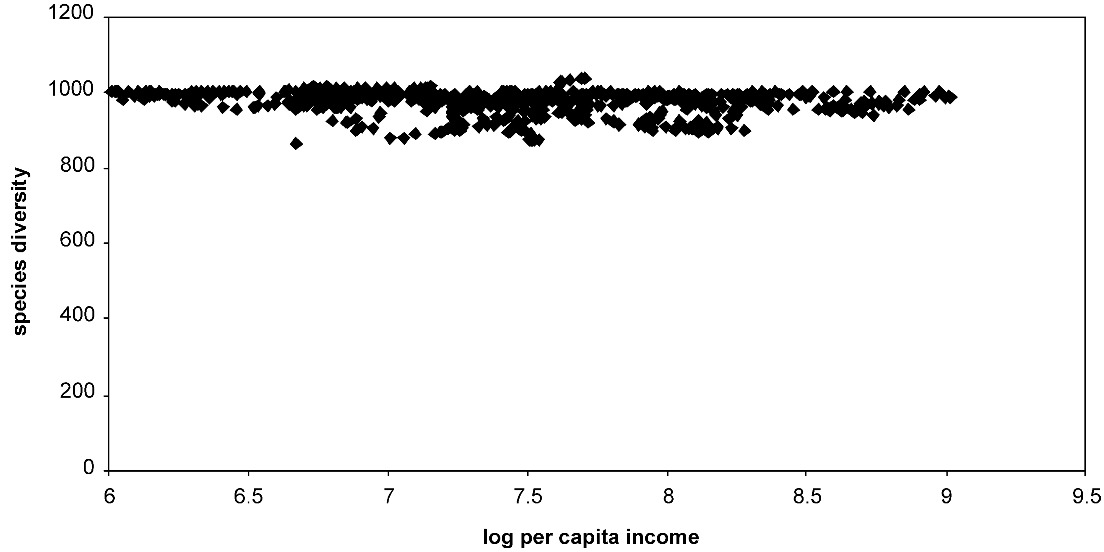

Fig. 2. Scatter plot of species richness versus per capita income.

positive environmental change such as secure property rights and environmental activism flourish.

The basic regression model for the hyperbolic equation is

$$S(i,t) = \alpha + \beta_1 1/\ln G(i,t) + \beta_2 C(i,t) + \beta_3 P(i,t) + \beta_4 T(t) + \beta_5 F(i,t) + \beta_6 D(i,t) + \varepsilon(i,t). \quad (3)$$

The linear equation is

$$S(i,t) = \alpha + \beta_1 \ln G(i,t) + \beta_2 P(i,t) + \beta_3 D(i,t) + \beta_4 T(t) + \beta_5 F(i,t) + \beta_6 D(i,t) + \varepsilon(i,t). \quad (4)$$

3.2. Protected areas and income

Government intervention to protect biodiversity is often seen as the primary mechanism to constrain the tendency for economic growth to cause habitat and species loss. We may expect, therefore, a positive relationship between economic development and the level of government intervention, for example in designating protected areas. Lightfoot (1994) explores this hypothesis using 11 different indicators of development and tests the correlation between development and area of designated conservation land in countries around the world in four discrete time periods: 1950–60, 1960–70, 1970–80 and 1980–90 using data from the 1990 United Nations List of National Parks and Protected Areas.

This study uses the updated 1993 United Nations List of National Parks and Protected Areas (IUCN, 1994), which uses the general term 'wildland management area' to embrace a range of protected area categories with different management objectives. Broadly, the UN list only includes those areas 'especially dedicated to the protection and maintenance of biological diversity' (WCMC, 2000), yet the extent to which species diversity is prioritised varies between categories. All categories are included and the sum for each country is normalised as a percentage of national land territory (see Appendix B for countries and time series[10]). This is the dependent variable in the regression analysis. Development is measured through per capita GDP only, again using the Penn World Table. This allows the relationship tested by Lightfoot for four decades to be disaggregated into a single continuous time series. Population density is included to account for the likelihood that in densely populated countries, significant tracts of ecosystem may have been lost prior to the advent of protection. In other words, there is little land protectable. A linear time trend (as before) also accounts for the positive time dependency of GDP and percentage protected area. Population change is further present. The basic regression model is

$$A(i,t) = \alpha + \beta_1 \ln G(i,t) + \beta_2 P(i,t) + \beta_3 D(i,t) + \beta_4 T(t) + \varepsilon(i,t). \quad (5)$$

OLS, fixed effects and random effects are tested as before. Again, theory in this case would suggest socioeconomic and natural factors varying by country and not captured by the regressors in (Eq. 5) should exert a significant influence. Therefore, fixed effects should be favoured. An inverted-U shape should not be produced, as countries are expected to supplement the area of protection as development proceeds. Thus, only a linear model is tested. The direction of the relationship is expected to be positive, if the area of land protected by the state increases with national economic development.

[10] 141 of the 144 countries with IUCN categorised 'wildland management areas' are included. Canada, Mexico and the USA are omitted due to time constraints.

Table 1
Summary statistics for selected variables in the species richness analysis

	Mean	Standard deviation
Proportion of predicted species lost ($\times$ 1000)	975.3	30.3
GDP (PPP$)	2132.2	1525.0
Time trend (year)	1981.7	6.0
Forest area (000 ha)	40,952.7	89,269.5
Percentage change in population	0.03	0.01
People per hectare of land	0.8	1.3
Democracy	7.3	3.1

3.3. CITES reporting and income

The Implementation of the Convention on International Trade in Endangered Species (CITES) is reviewed through a reporting process. Importantly, almost all parties are required to report on trade in species listed in Appendices A and B annually (see Appendix C) and there are strict deadlines for their submission. In the latest edition of World Resources (World Resources Institute, 2001), the percentage of reports submitted relative to those expected is listed for all parties. Reporting does not necessarily reflect actual implementation but it does reflect the only systematic means the convention has of monitoring how strictly trade is regulated (Lanchberry, 1998). Furthermore, failure to meet deadlines is identified as a particular problem among developing countries (Ong, 1998), where resources and expertise may be lacking.

On this premise, this study tests the relationship between development, as measured by per capita GNP for 1999 (World Bank, 2000),[11] and the percentage of expected reports actually submitted in 1999. GNP does not represent a direct causal factor, but it is correlated with the quality of national bureaucracy (Rausch and Evans, 2000).[12] The number of reports required is also included, as the dependent variable is sensitive to changes in this factor. For example, a country that fails to submit its one report meets 0% of its requirement, whereas a country failing to submit one of its 20 reports meets 95% of its requirement.

The regression model is a simple cross-section of the form

$$R(i) = \alpha + \beta_1 G(i) + \beta_2 T(i) + \varepsilon(i). \quad (6)$$

where R is the percentage of reports submitted for country i, G is GNP per capita and T is the total number of reports expected. The model is estimated by OLS.

4. Results

Tables 1–3 report empirical results for the species diversity analysis. Table 1 reports summary statistics. Table 2 displays the outcome of tests between OLS, fixed and random effects for Eqs. (3) and (4). Table 3 reports results for Eqs. (3) and (4), using the model selected on the basis of Table 2. Of particular note are the low variability in species (even after being multiplied by 1000) and GDP, as illustrated in Fig. 2. LM statistics clearly favour both fixed and random effects over simple OLS. The results of the Hausman test favour fixed effects over random effects in both cases. Thus, the hypothesis that national characteristics play a significant role in determining change in biodiversity is borne out. The hyperbolic and linear equations both fit the data significantly (see F-statistics) and in both equations, the income terms are significant at the 1% level. Therefore, it is not possible to determine which relationship best represents the data.

Table 2
Comparing OLS, fixed and random effects for the linear and hyperbolic equations linking species richness with income

	Hyperbolic equation	Linear equation
LM statistics	2480.9**	2536.7**
Hausman test	34.2**	33.7**

** shows statistical significance at the 1% level.

Furthermore, the signs on the income terms are the opposite of what we would expect. There is no theoretical reason to explain this, because it is well known that species are currently being lost at rates significantly greater than they are being created. The relationship is therefore assumed to be a product of low variation in both the species diversity and income data (Fig. 2). Time and forest area are both significant at the 1% level. Democracy is significant at the 5% level. Population density is significant at the 5% level for the hyperbolic equation but insignificant for the linear equation. Population change is statistically insignificant in both equations.

The strength of fit of the regression model is difficult to measure where OLS, fixed and random effects are compared. One paper compares R^2 values for OLS and fixed effects (Selden and Song, 1994), yet this ignores the point that R^2 is artificially higher for fixed effects. In addition, R^2 cannot be calculated for random effects. It is of interest, however, to compare R^2 for fixed effects with other studies. In both equations here, $R^2 = 0.76$. Cropper and Griffiths (1994) returned values of 0.13–0.64 for deforestation across Africa, Asia and Latin America. Shafik (1994) found no correlation ($R^2 = 0$) for annual and total deforestation yet values of between 0.96 and 1 for various measures of water and air pollution. Selden and Song (1994) also found values of above 0.95 for air pollution.[13] Results thus far cannot identify the better equation. The linear

[11] In Purchasing Power Parity (PPP) dollars.
[12] High income countries tend to have high bureaucratic performance ratings (Rausch and Evans, 2000).

[13] This list is by no means comprehensive. Only results for fixed effects are shown. Deforestation is strongly related to species diversity, as defined in this study. Water and air pollution are included to show that very strong correlations can result. The conclusions drawn should be put in this context.

Table 3
Estimates of the linear and hyperbolic equations linking species richness with income with fixed effects. Standard errors are in parenthesis

	Linear	Hyperbolic
ln GDP	16.3 (3.3)**	–
1/ln GDP	–	−2102176 (4295.1)**
Time trend	−2.2 (0.1)**	−2.1 (0.1)**
Forest area	1×10^{-4} (2×10^{-3})**	9×10^{-4} (2×10^{-3})**
Population change	−141.0 (139.5)	−62.6 (141.0)
Population density	−4.3 (3.4)	−5.8 (3.5)*
Democracy	−0.7 (0.3)*	−0.6 (0.3)*
F-statistics	53.7**	53.6**
Adjusted R^2	0.76	0.76

* shows statistical significance at the 5% level, **shows statistical significance at the 1% level.

equation is likely an artefact of the data distribution, which is strongly concentrated around the dependent variable's reference value.

Eq. (5) was estimated by OLS, fixed and random effects. An LM statistics value of 16,871.5 and a Hausman test value of 85.3 indicate fixed effects are most suitable. This confirms the posited relationship between government conservation effort and economic development outlined in the previous section. Table 4 outlines the coefficients and *F*-statistic. The model is significant at the 1% level and, of the regressors, all but population change are significant at the 1% level. Thus there appears to be a relationship between developments as measured by GDP per capita, and the percentage of land protected by governments to preserve biological diversity. The relationship is not, however, strong (R^2 for fixed effects = 0.36). So, although there may be a tendency towards increased conservation efforts (in terms of designated protected areas) with increasing income, other geographical and cultural determinants may be important in this relationship and not captured in such analysis. Some countries, such as New Zealand and Japan and others, for example, put much effort into marine conservation often without designating protected areas (see Brown et al., 2002). Historical legacies of land tenure arrangements are also crucial and mean that countries such as Scotland are only now implementing their first national parks.

Table 4
The linear relationship between percentage protected area and income, based on panel data for 141 countries (1950–1991) (see Appendix B)

	Result for fixed effects
ln GDP	12.3 (1.9)**
Time trend	0.2 (2×10^{-2})**
Population density	1.8 (0.2)**
Population change	2.7 (1.2)*
F-statistic	18.8**
Adjusted R^2	0.36

* shows statistical significance at the 5% level, ** shows statistical significance at the 1% level.

Eq. (6) on the relationship between conservation effort in the area of trade in endangered species and economic development, a cross-section, can only be estimated by OLS. The results are shown in Table 5. Eq. (6) is significant at the 1% level, as are the individual coefficients GNP per capita and number of reports expected. However, R^2 is only 0.3. Thus, GNP per capita, as a measure of development, is weakly related to the implementation of CITES. To put this in context, using the coefficient value for GNP per capita, an increase in GNP from \$500 per capita (corresponding roughly to the world's poorest nation) to \$20,000 (for developed countries) would lead to an increase in reporting requirement met of 20%. Other factors exert a greater influence on the percentage of expected reports submitted.

5. Discussion

The results enable three broad conclusions to be drawn. First, we conclude that an environmental Kuznets curve does not exist for species richness, as measured in this study. In other words it is unlikely that countries have ever been able to increase the species richness within their borders even if they have invested in conservation effort as incomes rise. Second, in such a model where no turning point is apparent, the fixed effects description is favoured over a random effects model. Third, economic development is related to the area of state protected land but it is not the overriding determinant of the rate of designation.

An environmental Kuznets curve does not exist for our measure of species richness. We find no reason to expect the presence of an environmental Kuznets curve, since species cannot be replenished at the rate at which they are being lost. Therefore, we have tested for the 'falling limb' of an environmental Kuznets curve, followed by stabilisation (but non-recovery) of species richness. However, we cannot discriminate between this hyperbolic curve and a linear decline in species richness, and the income coefficients are in any case inverted, certainly due to poor data.

How does this conclusion compare with previous research? A number of studies have investigated environmental Kuznets curves for deforestation, the basis for estimates of forest cover here. Most researchers, and particularly those using less restrictive assumptions of

Table 5
The relationship between percentage reports for CITES expected in 1999, actually submitted and income

	Results for OLS
GNP	1×10^{-3} (3×10^{-4})**
Reports expected	1.3 (0.3)**
Constant	43.3 (5.6)**
F-statistics	20.3**
Adjusted R^2	0.3

** shows statistical significance at the 1% level.

cross-country commonality, reject the environmental Kuznets curve hypothesis for deforestation (Cropper and Griffiths, 1994; Koop and Tole, 1999; Shafik 1994). For those models where the curve appears to exist with institutional and other variables taken into account, turning points may be well above current levels of income in the regions (e.g. Barbier, 2001, on the EKC for agricultural expansion). Bhattarai and Hammig (2001) find an environmental Kuznets curve for their sample of 66 countries, separated into three continents: Latin America, Africa and Asia. However, their large group of countries includes those Myers (1980) determines not to have significant tracts of tropical rainforest. The danger inherent in this larger list is that reforestation through plantations is mistaken for the recovery of primary rainforest, with its accompanying species. Therefore, caution must be exercised in transferring those results to biodiversity. Naidoo and Adamowicz (2001) estimate environmental Kuznets curves for species classified as threatened by the IUCN. They use seven taxonomic groups across 137 countries, and generate an environmental Kuznets curve for birds but not for other taxa such as reptiles and invertebrates. Birds are a charismatic taxonomic group, whose preservation is in demand relative to other groups. Thus their protection may be prioritised. Naidoo and Adamowicz's results provide a challenge to our aggregate approach, by showing conservation may well vary by taxonomic group. However, they are generating an environmental Kuznets curve for pressure on species, which is not the same as the measure of loss estimated on this paper.

Fixed effects are empirically favoured over random effects. For all equations estimated, results of the Hausman test favour fixed effects over random effects confirming our observations in the sections above. If the sample of groups (in this case countries) is comprehensive and not a small subset of a larger population, then fixed effects are favoured. From another viewpoint, international socioeconomic and environmental differences not captured by the regressors play a role in the relationship between species diversity and per capita income. This result should be compared with previous research, which has often overlooked the significance of the estimation technique. Selden and Song (1994); Koop and Tole (1999) compare fixed and random effects as part of their studies. Selden and Song find fixed effects are favoured for measures of urban air pollution, whereas Koop and Tole find random effects are favoured for deforestation. The most appropriate method must be confirmed by empirical test for the environmental issue in question given the discrepancy in models between studies.

The possibility of generating accurate data to test the environmental Kuznets hypothesis is limited at present. The nature of the study tends to exclude any cross-sectional datasheets that specifically list species numbers. However, a single 'snapshot' of species numbers would not be revealing since there are numerous environmental and geographical determinants of underlying species richness. In the absence of time series data on species loss, numbers must be derived indirectly. The species-area relationship has been frequently used to predict rates of species loss over coming years, but using a single value for the z constant belies wide confidence bands. Finally, there is substantial uncertainty surrounding the forest cover data. Species diversity is higher in primary rainforest but the only data set providing a sufficiently long time series, the FAO Production Yearbook (FAO, various years), includes 'all woody vegetations' (see discussion in Koop and Tole, 1999; Barbier, 2001). Additionally, the data itself has been compiled using different definitions and methods of measurement. Therefore, one of the important points to come out of this study is the need for a global biodiversity time series or, at least, accurate forest data on which biodiversity calculations can be based.

The relationships between per capita income and state protected land and with the implementation of the CITES convention reveal a number of insights. Economic development is related to the area of state protected land, although it is not a strong determinant on its own. Eq. (5) is significant at the 1% level, as is income per capita. Fixed effects are once more favoured over random effects, again because the data represent an almost complete sample and because international differences are important, resonating with the results of Lightfoot (1994). Eleven indicators of socio-economic development were correlated with protected land (also using UN/IUCN data), but the highest coefficient was 0.53. Lightfoot (1994) concludes that socio-economic development probably does affect government land protection, but that it cannot be confirmed empirically at the global scale. The reason is that 'a more complex set of interrelated social, economic, cultural and natural phenomena all work together simultaneously and cannot be isolated and examined independent from the whole system of variables[14], (p. 121). The result for fixed effects corroborates this previous analysis.

Economic development is related to the percentage of reports required by CITES actually submitted, although it is not a strong determinant. Eq. (6) is significant at the 1% level. Income, GNP per capita, is also significant at the 1% level. Thus, reporting for the CITES convention is only weakly dependent on income. Alternative explanations have been proposed. Ong (1998) states that 'many developing country parties in particular have failed to (submit their reports on time)' (p. 294), yet Lanchberry (1998) argues that 'reporting and implementation problems occur more often with developed countries than with developing countries' (p. 70). Neither study offers empirical evidence. The empirical results presented here suggest that low levels of income in a country may be correlated with restrictions on government enforcement of CITES and other environmental legislation.

[14] Lightfoot also pointed to data omissions: the UN/IUCN list does not include protected land under 1000 hectares, nor some public and private reserves.

Table A1
Panel data for protected areas

Africa:	
Cameroon	1972–1992
Congo	1972–1992
Gabon	1972–1992
Ghana	1972–1992
Ivory Coast	1972–1992
Kenya	1972–1992
Liberia	1972–1986
Madagascar	1972–1992
Nigeria	1972–1992
Sierra Leone	1972–1992
Tanzania	1972–1988
Uganda	1972–1992
Zaire	1972–1989
Central America:	
Costa Rica	1972–1992
El Salvador	1972–1992
Guatemala	1972–1992
Honduras	1972–1992
Mexico	1972–1992
Nicaragua	1972–1990
Panama	1972–1992
South America:	
Bolivia	1972–1992
Brazil	1972–1992
Colombia	1972–1992
Ecuador	1972–1992
Guyana	1972–1990
Peru	1972–1992
Venezuela	1972–1992
Asia:	
Bangladesh	1972–1992
India	1972–1992
Indonesia	1972–1992
Malaysia	1972–1992
Myanmar	1972–1989
Philippines	1972–1992
Sri Lanka	1972–1992
Thailand	1972–1992

6. Conclusion

These analyses of protected area designation and regulation do not lead to the conclusion that states are powerless to preserve biodiversity within their borders. Because of the interaction of habitat loss with agricultural expansion, land use practices and urban sprawl, there are many options to promote and protect locally and globally important biodiversity. The costs of such interactions may even be relatively modest and the benefits potentially huge. James et al. (2001) suggest that current global expenditure on protected areas is approximately \$6bn and that to expand this area to meet stated conservation goals may be achieved at a cost of \$12–21bn even accounting for opportunity costs of land use. Balmford et al. (2002) demonstrate that such modest investment has a high rate of economic return when the benefits of ecosystem services are brought into

Table B1
Panel data for protected areas

Africa:	
Angola	1960–1989
Benin	1959–1991
Botswana	1960–1989
Burkina Faso	1959–1991
Burundi	1960–1991
Cameroon	1960–1991
Central African Republic	1960–1991
Chad	1960–1991
Comoros	1960–1991
Congo	1960–1991
Djibuoti	1970–1987
Ethiopia	1950–1986
Gabon	1960–1991
Guinea	1959–1991
Guinea-Bissau	1960–1991
Kenya	1950–1991
Lesotho	1960–1991
Liberia	1960–1986
Madagascar	1960–1991
Malawi	1954–1991
Mali	1960–1991
Mauritania	1960–1991
Mauritius	1950–1991
Mozambique	1960–1991
Namibia	1960–1991
Niger	1950–1991
Reunion	1960–1989
Rwanda	1960–1991
Senegal	1960–1991
Seychelles	1960–1990
Sierra Leone	1961–1991
Somalia	1960–1989
South Africa	1950–1991
Sudan	1970–1991
Swaziland	1960–1989
Tanzania	1960–1988
Togo	1960–1991
Uganda	1950–1991
Zaire	1950–1989
Zambia	1955–1991
Zimbabwe	1954–1991
Central America:	
Belize	1980–1991
Costa Rica	1950–1991
El Salvador	1950–1991
Guatemala	1950–1991
Honduras	1950–1991
Nicaragua	1950–1990
Panama	1950–1991
South America:	
Argentina	1950–1990
Bolivia	1950–1991
Brazil	1950–1991
Chile	1950–1991
Colombia	1950–1991
Ecuador	1950–1991
Guyana	1950–1991
Paraguay	1950–1991
Peru	1950–1991
Suriname	1960–1989
Uruguay	1950–1991

(*continued on next page*)

Table B1 (*continued*)

Venezuela	1950–1991
Carribean:	
Bahamas	1977–1987
Barbados	1960–1989
Dominican Republic	1950–1991
Grenada	1984–1990
Haiti	1960–1989
Jamaica	1953–1991
Puerto Rico	1955–1989
St Kitts and Nevis	1983–1991
Trinidad and Tobago	1950–1989
Asia:	
Bangladesh	1959–1991
China	1960–1991
Hong Kong	1960–1991
India	1950–1991
Indonesia	1960–1991
Japan	1950–1991
Laos	1984–1991
Malaysia	1955–1991
Mongolia	1984–1990
Myanmar	1950–1989
Nepal	1960–1986
Pakistan	1950–1991
Philippines	1950–1991
Singapore	1960–1991
South Korea	1953–1991
Sri Lanka	1950–1991
Taiwan	1951–1990
Thailand	1950–1991
Oceania:	
Australia	1950–1990
Fiji	1960–1990
New Zealand	1950–1991
Papua New Guinea	1960–1991
Solomon Islands	1980–1988
Vanuatu	1983–1990
Western Samoa	1979–1990
Europe:	
Austria	1950–1991
Belgium	1950–1991
Bulgaria	1980–1991
Czechoslovakia	1960–1990
Denmark	1950–1991
Finland	1950–1991
France	1950–1991
Germany, Federation Republic	1950–1991
Greece	1950–1991
Hungary	1970–1991
Iceland	1950–1991
Ireland	1950–1991
Italy	1950–1991
Luxembourg	1950–1991
Malta	1954–1989
Netherlands	1950–1991
Norway	1950–1991
Poland	1970–1991
Portugal	1950–1990
Romania	1960–1989
Spain	1950–1991
Sweden	1950–1991
Switzerland	1950–1991

Table B1 (*continued*)

UK	1950–1991
USSR	1960–1989
Yugoslavia	1960–1990
Middle East:	
Algeria	1960–1991
Bahrain	1975–1988
Cyprus	1950–1991
Egypt	1950–1991
Iran	1955–1991
Iraq	1953–1987
Israel	1953–1991
Jordan	1954–1990
Kuwait	1980–1989
Morocco	1950–1991
Oman	1967–1987
Qatar	1980–1989
Saudi Arabia	1960–1989
Syria	1960–1991
Tunisia	1960–1991
Turkey	1960–1991
UAE	1980–1989
Yemen	1969–1989

the account. Clearly a change in societal demand for both biodiversity and protected areas would be required to realise such actions. But it would also require protected areas to actually fulfil their conservation aims—an area on which there remains much controversy (e.g. see Bruner et al., 2001).

The results presented in this paper paint a broad picture of the relationship between level of economic activity, terrestrial biodiversity loss and conservation effort. The cross-country nature of the data and analysis provides general lessons for conservation policy and practice that economic growth is ultimately as much part of the problem as the solution. But of course conservation of biological diversity is a multi-faceted issue that is spatially and culturally differentiated. The evidence in Naidoo and Adamowicz (2001) shows that the relationship even with income is complex when patterns between taxa are examined.

A more complete understanding is required of the benefits of conservation, for example, of the role of diversity in underpinning economic activity, particularly for resource-dependent economies. Ecosystem resilience and keystone processes within ecological systems may, in fact, represent what Folke et al. (1996) term the natural insurance capital for a society. Yet conservation policy focussed on protected areas may not deliver the resilience required or may not avoid the potential for irreversible habitat change. An understanding of the cultural and institutional landscape of conservation in society is, therefore, a further necessary element in implementing policy such that economic growth can be realised without the inevitability of biodiversity loss. This paper has shown that biodiversity conservation is, indeed, one of the class of environmental problems where economic growth on its own

Table C1
Parties to the CITES Convention

Reports expected in 1999, GNP data available:
Africa:
Algeria
Benin
Botswana
Burkina Faso
Burundi
Cameroon
Central African Republic
Chad
Congo, Democratic Republic
Congo, Republic
Egypt
Ethiopia
Gabon
Gambia
Ghana
Guinea
Guinea-Bissau
Ivory Coast
Kenya
Madagascar
Malawi
Mali
Mauritius
Morocco
Mozambique
Namibia
Niger
Nigeria
Senegal
Sierra Leone
South Africa
Sudan
Tanzania
Togo
Tunisia
Uganda
Zambia
Zimbabwe
North America:
Canada
USA
Central America:
Costa Rica
Dominican Republic
El Salvador
Guatemala
Honduras
Mexico
Nicaragua
Panama
South America:
Argentina
Bolivia
Brazil
Chile
Colombia
Equador
Paraguay
Peru
Uruguay
Venezuela

Table C1 (*continued*)

Carribean:
Trinidad and Tobago
Asia:
Bangladesh
China
India
Indonesia
Japan
Malaysia
Mongolia
Nepal
Pakistan
Philippines
Singapore
Sri Lanka
Thailand
Vietnam
Oceania:
Australia
New Zealand
Papua New Guinea
Europe:
Austria
Belarus
Belgium
Bulgaria
Czech Republic
Denmark
Estonia
Finland
France
Germany
Greece
Hungary
Italy
Netherlands
Norway
Poland
Portugal
Romania
Russia
Slovakia
Spain
Sweden
Switzerland
UK
Middle East:
Georgia
Iran
Israel
Jordan
Saudi Arabia
Turkey
UAE
Reports expected in 1999, no GNP data available:
Equatorial Guinea
Liberia
Rwanda
Somalia

(*continued on next page*)

Table C1 (*continued*)

Belize
Cuba
Guyana
Suriname
Afghanistan
No reports expected:
Cambodia
Fiji
Jamaica
Latvia
Myanmar
Swaziland
Uzbekistan
Yemen

is unlikely to ever result in a turning point towards a more sustainable and secure environmental future.

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Appendix A. Panel data for species diversity

Table A1.

Appendix B. Panel data for protected areas

Table B1.

Appendix C. Parties to the CITES convention

Table C1.

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