An open access e-Journal for sustainable solutions

Sustainability: Science, Practice, & Policy

Volume 4 • Issue 1
Spring 2008

ISSN: 1548-7733

Editorial

Articles

Community Essays

Book Review Perspectives

Published online • http://ejournal.nbii.org

email: ejournal@csa.com

nbii & ProQuest CSA

Sustainability: Science, Practice, & Policy
http://ejournal.nbii.org

EDITORIAL

Friedrich Schmidt-Bleek
Factor 10 Institute, Carnoules, France

Factor 10: The future of stuff

Old and New Policies

Given current economic and environmental policies, nature's life-sustaining services will continue to decline at a rapid pace. "Business as usual" may put human life on Earth eventually into question. Meanwhile, economic options will become limited and world peace more fragile.

Traditional *environmental* policies focus on dealing with specific problems. In certain respects, this approach has been quite successful. For instance, it has cleaned up water pollution, taken dangerous goods off the market, recycled certain products, and slowed the acceleration of climatic change.

However, since traditional problem solving begins *after* recognizing a problem's existence, such policies are neither helpful on a systems level, nor are they preventive in a general sense. Solving individual problems can even exacerbate other problems, in particular those as yet undiscovered.

Internalization of the environmental costs of individual known problems among millions of possible destructive interactions between hundreds of thousands of different pollutants and the highly complex ecosphere *cannot* be relied upon when seeking sustainable solutions.

Lately, *recycling* appears to be experiencing a policy revival—except that its administrators and practitioners now call it "resources policy." While recycling can contribute toward saving natural resources, there is no evidence that this "end-of-the-pipe" approach could ever lead to sustainable conditions. Much of the damage to the services of nature has been done *before* waste treatment can begin. Typically, national recycling policies can cover only a few percent of total materials flows. In Germany, about 1% of the total resources flow is recycled today—at a yearly expense of several billion Euros.

At present, more than 95% of the resources lifted from nature are wasted before the finished goods reach the market. And many industrial products—such as cars—demand additional natural resources while being used.

It is high time to eliminate the systemic root cause for the incompatibility between today's economic activities and the continued functioning of the life-sustaining services of nature, without which humans cannot survive. For survival on planet Earth, the time has come to implement truly damage-preventing strategies.

Today, the fundamental physical flaw of human activities is the enormous consumption of natural resources per unit output of value or service. This observation applies to all renewable and nonrenewable materials, domestic animals, water, soil, and land use.

The key for sustainability is to radically increase the resource productivity of all economic activities, including energy generation.

While it may seem obvious, it is nevertheless worth repeating that climatic change, too, is the consequence of enormous flows of human-induced carbonaceous material and of large quantities of N_2O emissions originating from the technical fixation of millions of tons of nitrogen from the air in the production of fertilizer**.**

It has been widely accepted that to be successful in approaching sustainability an average minimum *tenfold dematerialization* of the Western style of life in absolute terms has to be achieved.

Today, the environmental safety threshold has already been surpassed, as is evident from such developments as climatic change, widespread water shortages, desertification, disease proliferation, massive erosions, and increasing natural catastrophes such as hurricanes and floods. And yet, only some 20% of humankind enjoys the full benefits of our economic model, while *all* human beings—and in particular the poor—have begun to suffer the consequences of its flaws.

But even if one were to ignore the ecological problems caused by the overuse of nature, globalizing the western lifestyle is *not* possible, because it would require more than two planets as a resource basis. Rapidly rising raw material prices testify to this fact.

Technologies for Tomorrow

To translate the findings just outlined into a general guideline for policy development, the European Union's Panel on Eco-Innovation has recently concluded:

> Eco-Innovation means the creation of novel and competitively priced goods, processes, systems, services, and procedures that can satisfy human needs and bring quality of life to all people with a life-cycle-wide minimal use of natural resources (material including energy carriers, and surface area) per unit output, and a minimal release of toxic substances (Schmidt-Bleek, 2007).

This observation suggests that continued reliance on traditional "environmental technologies" is no longer enough. Many examples exist where incremental improvement of existing technologies has increased resource productivity two to four times. However, sufficiently decoupling production and consumption from nature requires new systems, goods, services, processes, and procedures for meeting human needs. One such novel solution is to propel ships by "sky sails," potentially saving up to 60% of fuel for 50,000 freighters at competitive costs. To such solutions, the markets of the future will belong.

The development of as-yet-not industrialized countries is impossible without dematerialized solutions. Entrepreneurial success on all economic levels—including exporting goods, blueprints, and services—will also depend on striving for maximum resource productivity, as will gaining independence from those possessing raw materials—including energy carriers—and preventing armed conflicts over access to natural resources.

While increasing material productivity, reducing erosion, and using land optimally are necessary for moving toward sustainability, they are not the only conditions. Welfare is more than material wealth and consumption. Welfare includes factors such as employment with adequate income, equity, education, health, safety (freedom from violence), environmental aesthetics, social security, and leisure.

Goals for Sustainability and Suitable Indicators

Creating new values for civil society will require the casting of goals with a definite time frame. Wherever possible, these goals should be encapsulated into measurable physical terms so that development can be managed. To the extent that value creation requires natural resources, the goals have to respect the laws of nature.

Specifics, including policy instruments, for protecting nature's services may vary for differing geographic and geological conditions. However, since humankind has only one planet, the fruits of the commons and its protection must be shared fairly.

Scholarly literature has suggested the following global goals for the target year 2050:

- The ecological footprint per person should not exceed 1.2 hectares.
- The worldwide per capita consumption of nonrenewable resources should be less than five to six tons per year. (This goal implies a tremendous increase in resource efficiency in industrialized countries. In Germany, for instance, it means a Factor 10 increase, requiring a yearly absolute improvement in resource productivity of almost 5%, starting now. In the United States, the reduction of resource use would have to amount to about a factor of 15, and in Finland close to a factor of 20).

These goals must be discussed further. If the dematerializations indicated above for industrialized countries were achieved, this would allow developing countries to increase their use of natural resources without jeopardizing the overall goal of global sustainability.

Because it is impossible to manage a system without metrics, we must agree on appropriate indicators. These must satisfy six criteria: 1) they must be based on measurable quantities; 2) they must be generally applicable on a "cradle to grave" basis; 3) they must be directionally true; 4) they must be cost efficient in their application; 5) they must be based on scientific evidence and on broadly accepted guidelines such as the above definition for ecoinnovation; and 6) they must respect and relate to the laws of nature (for instance, economic indicators must go beyond conventional measures of gross domestic product (GDP)).

As to the ecological dimensions of sustainability, calculations of total material requirements (TMR), material input per service-unit (MIPS), and ecological rucksack measurements satisfy these criteria.[1] In addition, the value/weight and labor input/weight of industrial goods have been suggested as initial indicators. Furthermore, great need remains for indices that reflect the resource implications of progress in the institutional, social, and economic dimensions of sustainability.

[1] "Ecological rucksack" refers to the total material input for manufacturing a product, from cradle to the point of sale in kilograms (kg), minus the mass of the product, itself (in kg).

Economic Policies

No incentives or policies currently exist for a sufficiently resource-efficient economy. Adjusting the economic and fiscal framework is therefore the most fundamental and urgent prerequisite for moving toward sustainability.

For this shift, a strong preference seems to be emerging for economic instruments, such as environmental tax reform and market-creation policies, including tradable permits. Instead of value-added taxation, for instance, it may be more efficient to tax natural resource use before goods for final use have been produced, while lowering taxation of labor accordingly. But, because of market failures, economic instruments may not work in all cases; therefore other instruments and measures should be considered, such as information and coordination instruments and command-and-control mechanisms, for instance, adjusting norms and standards.

The choice of policy options should depend on their efficiency in dematerializing goods and services at the least possible cost to civil society.

Today, the public procurement of goods and services amounts to some 15–20% of final consumption. Preference for dematerialized goods, infrastructures, and services could give the manufacturing sector a powerful incentive to increase resource productivity. In Germany, this may be a particularly attractive option as it has been shown that some 20% of resource input production costs could be saved on average without negatively affecting outputs.

Agreement has also emerged in civil society that improving education and training on all levels, as well as enhancing the public availability of relevant information, will play a central role as part of a progressive strategy.

Basics for Approaching Sustainability

1. The key flaw of the present mainstream economic model is its lack of incentives for increasing the productivity of natural resources.
2. This flaw creates a dangerous situation because the present rate of resource use:

    - Cannot be globalized since at least two planets would be needed as a resource basis
    - Does not permit the fair development of poorer countries
    - Increases the potential for worldwide conflict
    - Increases the dependence of many countries on others that are more blessed with natural resources
    - Can deplete or exhaust nature's services without which humankind cannot survive.

3. Among the policies that governments can institute to improve the situation, preference is emerging for economic instruments, *inter alia*, aiming simultaneously at dematerialization as well as at job creation by shifting taxes and overheads from labor to natural resources.
4. During the next few decades, the productivity of natural material resources has to be improved by at least a factor of ten compared to current resource consumption in western countries.
5. The use of fossil-energy carriers must be abandoned as rapidly as possible through a switch to inexhaustible sources of energy with the help of dematerialized technology.
6. Goals for sustainable value generation, expressed in measurable terms, are required for monitoring and managing progress toward a future with a future.
7. Indicators related to resource saving have to be set for monitoring ecological, economic, social, and institutional developments.
8. As new technical and societal developments tend to require ten to twenty years to take hold, dematerialization must commence immediately.
9. A single country cannot bring about the needed changes, but Europe with its historic experiences, economic power, and technical skills has a realistic chance to lead humankind to a more promising future.

Author's Note

The first World Resources Forum will be held in Davos, Switzerland on September 16, 2009. Consult also "Future: Beyond Climate Change," position paper 08/01 at http://www.factor10-institute.org/files/FUTURE_2008.pdf.

References

Schmidt-Bleek, F. 2007. *Nutzen wir die Erde richtig? (Let us use the earth?)*. Frankfurt: Fischer Verlag.

About the Author

Friedrich Schmidt-Bleek is a physicochemist who has been responsible for administering the federal environmental research program in Germany and for developing the German Chemical Act. He advanced the concept of Factor 10 dematerialization of the economy for reaching economic and ecological sustainability. Schmidt-Bleek also advises the European Commission on sustainable development and has published some twenty books on eco-economic issues.

He is founding president of the Factor 10 Institute (http://www.factor10-institute.org) and can be reached at Factor 10 Institute, La Rabassière Carrère des Bravengues, F 83660 Carnoules, France.

ARTICLE

A modest proposal: global rationalization of ecological footprint to eliminate ecological debt

Brian Ohl[1], Steven Wolf[2*], & William Anderson[3]

[1] Cornell Institute for Public Affairs (CIPA), Cornell University, 294 Caldwell Hall, Ithaca, NY 14853 USA (email: brian_ohl@yahoo.com)
[2] Department of Natural Resources, Cornell University, 124 Fernow Hall, Ithaca, NY 14853 USA (email: saw44@cornell.edu)
[3] School of Operations Research and Information Engineering, Cornell University, 414A Rhodes Hall, Ithaca, NY 14853 USA (email:wanders@gmail.com)

In the context of ecological overshoot, extreme poverty, and profligate consumption, we propose using ecological footprint analysis (EFA) to regulate and rationalize material consumption worldwide. EFA quantifies human-consumption flows relative to renewable natural capital stocks given specified levels of technology. Worldwide, 1.8 global hectares (gha) of bioproductive land exist per person, yet the human population is currently consuming 2.2 gha per person. Given global overshoot and the radically uneven distribution of consumption, we propose a global regime of cap-and-trade of ecological footprint. Under the terms of our modest proposal, all nations would be allocated population-based ecological footprints of an "earthshare" of 1.8 gha per person. Nations with large per capita footprints would be obligated to make reductions through some combination of reduced consumption, resource-productivity gains, population decreases, ecological restoration, and purchase of footprint credits. In contrast, countries with small per capita footprints could sell footprint credits to finance modernization along ecological lines. Mathematical simulation of our proposal indicates global convergence of nations' ecological footprints in 136 years. In our view, the obscenity of contemporary ecological degradation and human suffering is perhaps rivaled by the audacity of our proposal to commodify biocapacity worldwide. We leave it to the reader to compare our response to institutional failure and the problem of distributive justice to the remedy Swift offered in 1729.

KEYWORDS: geopolitics, international trade, environmental equity, incentives, sustainable development, economics, resource conservation

Introduction

Over the past fifty years, human alteration of earth's ecosystems has accelerated and diversified markedly, degrading the natural capital and related ecosystem functions on which we depend. The Millennium Ecosystem Assessment (MEA) undertaken by the United Nations from 2001-2005 inventoried the state of the world's ecosystems and quantified the effects of human activities on them. The analysis was designed to "establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being" (MEA, 2005). According to its findings, approximately 60% of the ecosystem services evaluated are being degraded or used unsustainably (MEA, 2005). This statistic is, of course, a measure of the "overshoot" that environmentalists have long identified as a failure to recognize and heed the limits to growth (Catton, 1980).

Linked to this ecological crisis, we face a poverty crisis and a related ethical dilemma rooted in concern for distributive justice. Without sufficient access to natural resources, humans cannot survive and certainly cannot thrive. Meeting our basic human needs requires water, food, shelter, clothing, and other goods derived from nature. With incomes of less than one dollar a day, close to a billion people currently live in "extreme economic poverty" and, by definition, lack access to essential natural resources to meet basic needs (World Bank, 2008). The profligate use of resources by the rich within and across nations degrades natural resources and limits access, exacerbating problems of the poor who often rely directly on such resources for income generation and sustenance. Impoverished people, with scant alternatives and a foreshortened planning horizon, further degrade local environments, deepening their hardships and further eroding local potential for socioeconomic development. Roberts (2003) has referred to the ecological degradation that accompanies a lack of economic opportunity as the "pollution of poverty."

From an ecological perspective, poverty remediation is not unproblematic. As is increasingly

*Corresponding Author.

well understood, improvement of the economic status of the world's poor could, over time, result in unintended and devastating side effects. If developing nations adopt first-world consumption patterns and dominant technical designs, emissions—including greenhouse gases from private automobiles and nitrogen from inorganic fertilizers—would overwhelm ecosystem-buffering capacity at local and global scales (Tilman, 1999; Galloway, 2000). In this context, people are seeking to expand what is now an unacceptable choice set: to take no action to alleviate systemic desperate poverty and the accompanying resource degradation or, alternatively, to cope with the pollution, ecological disorganization and overshoot that would result from adding billions of people to today's global middle class.

In the contemporary political economic context much of the energy and optimism directed at finding an alternative to this pair of poor choices focuses on so-called "leap-frog technologies," radical innovations that will allow rapidly developing nations to avoid the negative socioeconomic and ecological consequences of modernization while imparting competitive advantages (e.g., hydrogen-based transportation, regenerative agriculture). While we share a certain measure of respect for innovation dynamics as a central element of progressive strategies, in our analysis we emphasize institutional processes regulating the pace and nature of technical change (Berkhout, 2002). We regard technologies as responses to scarcity and changes in factor prices, and factor prices are subject in part to social controls (i.e., politics). Our commitment to social controls (i.e., a retreat from voluntarism and an embrace of regulations and externally imposed limits) is a principle element of the immodesty suggested by the title of this article.

Our proposal is a policy instrument to address the social and ecological challenges that humanity confronts. We propose to regulate and rationalize material consumption at the global level through establishment of a system of cap-and-trade of consumption rights. Like Jonathan Swift in his essay of 1729, our interest lies in advancing a public dialogue focused around our shared problems. We leave it to the reader to compare our proposal to Swift's remedy to the dual problem of social inequity and how to live within our means.

An Institutional Analysis of Ecological Overshoot

As identified in the MEA, resource degradation is largely a consequence of a failure to develop institutions that account for the world's natural capital in supporting our welfare. Ecosystem services—valuable functions performed by natural systems, including purification of air and water, pollination of crops, stabilization of climate and habitat (NRC, 2005)—are often not valued or are undervalued in the marketplace, in policy-making processes, in organizations, and at the household level (Daly & Cobb, 1989; Costanza et al. 1997; Balmford et al. 2002). As a consequence of institutional failure, ecosystems' contributions to socioeconomic systems are targets of chronic under-investment (Wackernagel & Rees, 1997).

Ecological modernization theorists argue that the core contemporary political, social, and economic institutions of late modernity have the potential to address the current ecological crisis (Crowley, 1999; Marx, 2000; Mol & Sonnenfeld, 2000; Mol, 2000, 2001, 2003). Specifically, they argue that hyperindustrialization—radical resource productivity gains—will limit throughput, reducing inputs, waste, and attendant ecological degradation (Hawken et al. 1999). As part of this material transformation, ecological rationality is incorporated into economic logic, social organization, and politics. In this framework, socioeconomic development and environmental protection are seen as complementary, rather than contradictory, goals. As an integrated social-scientific analysis and policy prescription, ecological modernization points to institutional change—defined to include formal coordination mechanisms such as rules and prices as well as cognitive structures such as conceptions of fairness, justice, and personal identity—as capable of transforming circuits of production and consumption and attendant ecological disorganization. In keeping with this generic prescription of multi-scaled institutional innovation, we introduce a policy proposal in which ecological footprint analysis (EFA) would be used to measure, monitor, and manage consumption of natural resources in pursuit of sustainable development.

EFA seeks to provide a unified, comparable measure of human ecological impact known as the "ecological footprint" (Loh & Wackernagel, 2004). This footprint can serve as an indicator of sustainability (York et al. 2003; Loh & Wackernagel, 2004). Human consumption greater than available biocapacity—i.e., productive capacity of earth's renewable natural resources to produce goods and services on a sustainable basis—constitutes ecological overshoot (Venetoulis et al. 2004). We outline a globally scaled environmental agreement to reduce and redistribute consumption as represented by ecological footprint. Our policy is designed to ensure against global ecological overshoot and to raise the material standard of living of the world's poor to an equitable level.

In keeping with general tendencies in governance (e.g., Mazmanian & Kraft, 1999), we employ a property rights or market-based approach to protect natural capital, including a cap-and-trade scheme. This proposal conforms to the general logic of a system of tradable environmental property rights, as outlined by Carley & Spapens (1998). In its global scope, its reliance on nation states as units of analysis, the centrality of cap-and-trade logic, and its focus on inducement of technological change, our proposal mimics key aspects of the Kyoto Protocol.[1]

This agreement will serve the following functions.

1. Limit the extent of nations' ecological footprints and provide incentives to reduce footprint by phasing in increasingly stringent controls over time.
2. Provide incentives for product and process innovations and system redesign resulting in heightened eco-efficiency and ecological restoration (i.e., increased biocapacity).
3. Transfer resources to relieve poverty and to support sustainable development of poor nations.
4. Provide incentives to rapidly industrializing nations such as China and India to pursue sustainable development paths.

The following section justifies the regulatory controls we propose through historical and ethical arguments. We then review the mechanics of EFA and our policy proposal in detail. We mathematically simulate how our proposition would affect ecological footprints across the globe as we move toward convergence of ecological requirements of all nations on earth. The final section reflects on the implications of our analysis.

Critique of the Unilateral Appropriation of Natural Resources

In seeking to rationalize consumption and progress toward sustainability, we must simultaneously address both excessive consumption and insufficient access to natural resources and investment capital. While it is likely inaccurate to argue that lifestyles of the rich explain the misery of the poor, these problems are connected; more importantly, they imply a unified strategy to promote sustainability.

Severe disparities in income and consumption exist between developed and less developed regions, and this stratification is increasing.

> The income gap between the fifth of the world's people in the richest countries and the fifth of the world's people in the poorest was 74 to 1 in 1997, up from 60 to 1 in 1990 and 30 to 1 in 1960. By 1997, the richest 20% captured 86% of world income, with the poorest 20% capturing a mere 1% (Wallach & Woodall, 2004).

Throughout his or her lifetime, the average American "accounts for the consumption of 540 tons of construction materials, 18 tons of paper, 23 tons of wood, 16 tons of metal and 32 tons of organic chemicals" (Carley & Spapens, 1998). Examinations of aggregate consumption estimate that people in high-income countries consume, on average, six times as much as people in low-income countries (Loh, 2002). Rich nations are largely responsible for ecological overshoot and the consequent drawdown of natural capital. While seldom contemplated and nowhere made explicit, drawdown of natural capital is an expression of a property claim. The consumption of a disproportionate share of the world's natural capital constitutes *unilateral appropriation of natural resources* (Pogge, 1998).

> Although accepting that all inhabitants of the earth ultimately have equal claims to its resources, defenders of capitalist institutions have developed conceptions of justice that support rights to unilateral appropriation and discretionary disposal of a disproportionate share of resources. They argue that a practice permitting unilateral appropriation of disproportionate shares is justified if all are better off under this practice than they would be if such appropriation were limited to proportional shares (Pogge, 1998).

According to the "Lockean Proviso," in a state of nature people are subject to a moral constraint in that unilateral resource appropriation is justifiable only if "enough, and as good" remains for others (Pogge, 1998). That is to say, shares must be proportional for all. This provision can be lifted "only if everyone will be better off under the new rules than under the old, that is, only if everyone can rationally consent to the alteration" (Pogge, 1998). Pogge argues that this conditional requirement is not fulfilled today because hundreds of millions of people are born into and remain in extreme poverty. While it is possible for these people to rent out their labor, the compensation they receive is often not enough to meet their basic needs.

In this context, it is not true that "all strata of humankind, and the poorest in particular, are better

[1] In some respects, our proposal is an extension of climate change mitigation schemes such as Aubrey Myer's proposed program of global contraction and convergence of greenhouse gas emissions. Thanks to Maurie Cohen for this reference.

off with universal rights to unilateral appropriation and pollution than without the same" (Pogge, 1998). Thus, the Lockean proviso remains in force (Pogge, 1998). Unilateral appropriation of natural resources would only be justified if there were some form of compensation from the benefit of this appropriation. Following this logic, Pogge (1998) calls for a "Global Resources Dividend" to compensate the world's poor for unjust unilateral appropriation of natural resources.[2] In recent years, the idea of "ecological debt" has been used to describe debt accumulated through unilateral appropriation of natural resources and by extension the global commons (Sachs, 2004). Simms (2005) argues that this cumulative process, which began hundreds of years ago under Western colonialism, is largely responsible for the economic successes of the world's rich nations. In the context of ecological overshoot and extreme poverty, Simms (2005) contends that rich nations have accumulated an ecological debt far greater than the questionable financial debt of poor nations.

While reparation payments have not been widely contemplated, existing international environmental agreements have incorporated notions of equity and distributive justice. Precedents recognize differences in rights and responsibilities across nations. For example, the 1987 Montreal Protocol "differentiated between rich and poor countries" and the 1992 Convention on Biological Diversity refers to the need for "equitable sharing" of the agreement's costs and benefits (Beckerman & Pasek, 2001). The United Nations Framework Convention on Climate Change (UNFCCC), signed at the Río Conference in 1992, states, "the parties should protect the climate for the benefit of present and future generations of humankind, on the basis of equity and in accordance with their common but differentiated responsibilities and respective capabilities" (Beckerman & Pasek, 2001). Further, the Kyoto Protocol differentiates responsibility in that "countries with higher per capita emission rates are expected to accept bigger cuts from the levels of their emissions in 1990" (Attfield, 1999). Kyoto proponents justify this provision with the argument that "while everyone has a right to develop, only some nations have in fact developed sufficiently. Those that are not yet sufficiently developed are therefore entitled to continue with their own development and not expected to divert resources in the mitigation of climate change" (Shue, 1995). Recognizing these precedents and the moral requirements underlying them, our proposal allocates responsibility to adapt (and presumably cost) in proportion to national consumption.

Ecological Footprint Analysis

As part of an analysis of the ecological implications of economic throughput, in the early 1970s Paul Ehrlich and John Holdren developed the IPAT model (I = P x A x T) in which human environmental *Impact* equals *Population* multiplied by *Affluence* (i.e., quantity and quality of consumption) multiplied by *Technology* (i.e., efficiency of production and waste assimilation) (Rees, 2000). This framing of the material problem draws our attention to a set of relationships that define opportunities for intervention at multiple scales.

In the tradition of critical analysis of natural resource consumption and a failure to "tread lightly on the earth," EFA estimates the areal extent of biologically productive land and water ecosystems required to support some specified individual, activity, organization, territory, or even the planet as a whole (Chambers et al. 2000). For a given individual, the ecological footprint is the amount of bioproductive land and water area, of average quality, needed to support his or her particular lifestyle as proxied by types and amounts of consumption (i.e., A = affluence) and the efficiency of local processes of material transformation and waste assimilation (i.e., T = technology). Multiplying average values of these factors for a given country by the number of citizens in that country (i.e., P = population) yields an estimate of the nation's ecological footprint.

Worldwide, the average ecological footprint of a human being is estimated to be 2.2 global hectares (gha). While calculations suggest that average per capita ecological footprint has been slightly declining since 1980, this decrease has been overshadowed by the aggregate increase stemming from population growth (Venetoulis et al. 2004). As Loh & Wackernagel (2004) explain, "global Ecological Footprint changes with population size, average consumption per person, and resource efficiency. Earth's biocapacity changes with the amount of biologically productive area and its average productivity." In this framework, ecological footprints are tallied as debits, while biocapacity constitutes assets (thus, ecosystem services are analogous to interest). EFA is thus a means of evaluating our balance sheet and conducting an ongoing critical accounting of the world's renewable natural capital.

Global biocapacity is calculated by placing the world's available land and water into a set of ecological categories (Wackernagel et al. 2005). Six categories are recognized as productive: arable land,

[2] Hancock (2003) goes further and argues that access to natural resources is a basic human right guaranteed under international human rights declarations such as the Universal Declaration of Human Rights (UNDHR) and International Covenant on Economic, Social, and Cultural Rights (ICSECR) (United Nations, 1948, 1976).

pastureland, forested land, productive sea space, built land, and carbon land.[3] These general categories are further differentiated into subcategories for purposes of compiling a resource inventory of bioproductive area. Bioproductivity coefficients are assigned to each land and sea type based on a synthesis of scientific information. These coefficients are predicated upon "equivalence factors (capturing the productivity difference among land-use categories) and yield factors (capturing the difference between local and global average productivity within a given land-use category)" (Wackernagel et al. 2005). Application of these coefficients to resource-inventory data generates an area-weighted average to represent global renewable ecological capacity.

After subtracting 12% of available bioproductive area from the global total for the sustenance of non-human species, 11.3 billion gha of biocapacity remain (Holmberg et al. 1999; Loh & Wackernagel, 2004).[4] For a population of 6.302 billion people (2003 estimate), approximately 1.8 gha are available per capita (GFN, 2006). Wackernagel and Rees refer to the amount of bioproductive land and sea of average quality available per capita worldwide as an "earthshare" (Chambers et al. 2000). The relationship between an earthshare and an individual's footprint is a measure of sustainability. Ratios greater than 1.0 indicate living within means. Ratios less than 1.0 signal overshoot. Similar estimates can be produced at the level of individual nations or the globe as a whole.

Two prominent methods are used to estimate ecological footprints: national footprint accounting (NFA) and input-output analysis. For a given nation, NFAs are calculated for a series of categories of material goods that capture core dependence on natural resources. National imports are added to domestic production and exports are subtracted (Wackernagel et al. 2005; GFN, 2006). For each category of goods, this measure of net material consumption is translated into a spatial measure. In conducting this translation, commodities consumed from croplands, pasturelands, and forests, for example, are differentiated into primary and secondary products. Primary products are unprocessed, for example corn. Secondary products are the goods derived from primary products, for example high fructose corn syrup. For primary products, ecological footprint calculations are based on global yield estimates. For processed secondary products, calculations are based on national conversion coefficients representing the efficiency of national production and waste-assimilation processes (Wackernagel et al. 2005).

Aggregating the spatial measures derived for each category, as described above, yields a nation's ecological footprint. Per capita footprint is calculated by dividing national footprint by national population. Obviously, focusing on average levels of consumption obscures vitally important variance within a population (Sachs & Santarius, 2007). The NFA approach also fails to accurately reflect all resource uses associated with international trade (i.e., services), and does not inform whether impacts occur within or outside a country due to the aggregation of imports and domestic production (Lenzen & Murray, 2001, 2003; Turner et al. 2007; Wiedmann et al. 2007a, 2007b).

In recent years, input-output analysis has been applied to obtain more robust ecological footprint estimates at national, regional, and local levels (Bicknell et al. 1998; Lenzen & Murray, 2001, 2003; Wiedmann et al. 2006). Environmental input-output analysis captures resource flows through interindustrial monetary transaction data (Lenzen & Murray, 2003). Multiple approaches, including a basis in land condition (i.e., accounting for losses in ecological functionality of land that is altered from a pristine state), have been introduced for calculating input-output based ecological footprints (Lenzen & Murray, 2001, 2003). There is, however, no standardized method of ecological footprint accounting based on input-output analysis, making different methods incompatible (Wiedmann et al. 2006). Given current objectives, we will rely on NFA data for development and assessment of our policy proposal.

A Property Rights Approach to Managing Ecological Footprint

Past international environmental agreements have focused on specific ecological risks stemming from discrete pollution streams. The Montreal Protocol, for example, confines itself to the reduction and elimination of ozone-depleting substances such as chlorofluorocarbons (CFCs). The Kyoto Protocol focuses on the reduction of greenhouse-gas emissions such as carbon dioxide (CO_2) to prevent or slow climate change. While these agreements suggest that international cooperation is possible (Speth & Haas, 2006), they are limited in their ability to engage with the overarching problems of ecological overshoot and inequitable resource distribution.

[3] Carbon land is the area required to sequester greenhouse gas emissions. This footprint component now accounts for approximately one half of the global ecological footprint, a nine fold increase from 1961 (GFN, 2007).

[4] Allocating 12% of biocapacity for biodiversity conservation is derived from rather informal estimates of conservation biologists as to habitat required to avert catastrophic acceleration of extinction rates (Chambers et al. 2000).

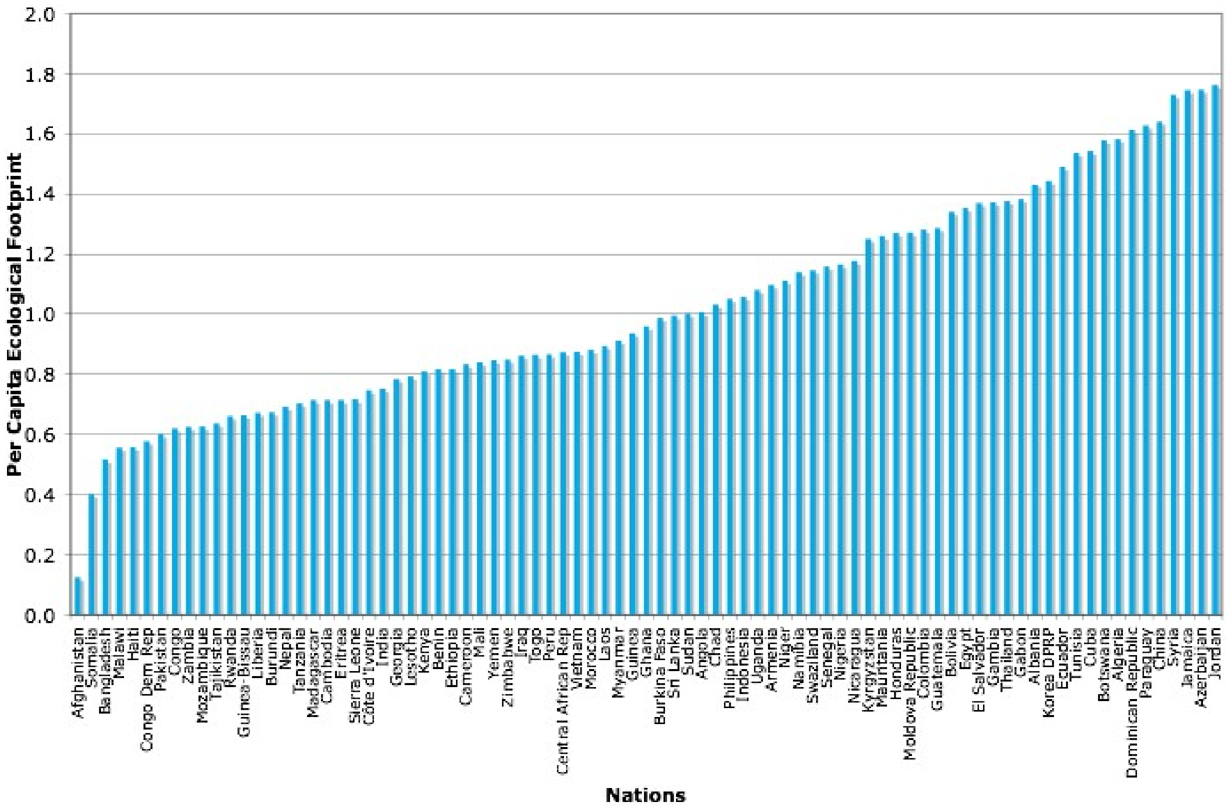

Figure 1 Creditor or Surplus Nations (GFN, 2006).

In seeking ways to address our current situation, we are drawn to a property rights-based approach. We propose to allocate ecological footprint to nations (and by extension, their citizens) and then place controls on national footprints while simultaneously creating exchange mechanisms. National footprints are to be allocated on the basis of population (national population x earthshare).[5]

As stated earlier, current estimates define an earthshare as 1.8 gha per capita (Loh & Wackernagel, 2004). Currently, of the 147 nations included in the Global Footprint Network database, 83 have per capita ecological footprints below or equal to 1.8 gha (GFN, 2006; Hails, 2006) (Figure 1). These nations are consuming, in terms of per capita averages, at levels below their allocation of earthshare. We refer to this group of countries as "surplus" or creditor nations, as under our proposed agreement they are positioned to sell consumption rights (i.e., extend ecological footprint credit) to "debtor" nations.

The remaining 64 nations have ecological footprints greater than 1.8 gha per capita (Figure 2). In the present term, these are "debtor" nations. Under our proposed agreement, they are responsible for reducing consumption and/or purchasing ecological footprint credits to meet their global obligations. To progress toward global sustainability and equity, our proposition aims to ensure that, over time, debtor nations reduce their per capita footprints to levels below or equal to an earthshare. Creditor nations would be obligated to maintain per capita footprints at or below an earthshare.

[5] Alternative approaches for the initial allocation of consumption rights could be contemplated. For example, it is easy to imagine the grandfathering of high consumption nations, awarding them additional consumption allowances according to the logic that they should be permitted to retain privileges gained under the old standard of "first come, first served." Alternatively, national consumption allowances could be tied to national biocapacity according to the logic that this will result in spatial rationalization of population and investment relative to ecological endowments. In defending our decision to allocate rights irrespective of development status and biocapacity, we reject the first argument categorically. We also dismiss the second argument because such a strategy would ignore global interdependencies and counteract efforts to protect existing biocapacity from over-exploitation. Critics of our approach may contend that a population-based allocation rewards nations with large populations and high population growth. While population pressures are clearly part of the IPAT framework, we argue that as nations reach consumption convergence through the international agreement outlined here, population growth in developing nations will likely decline.

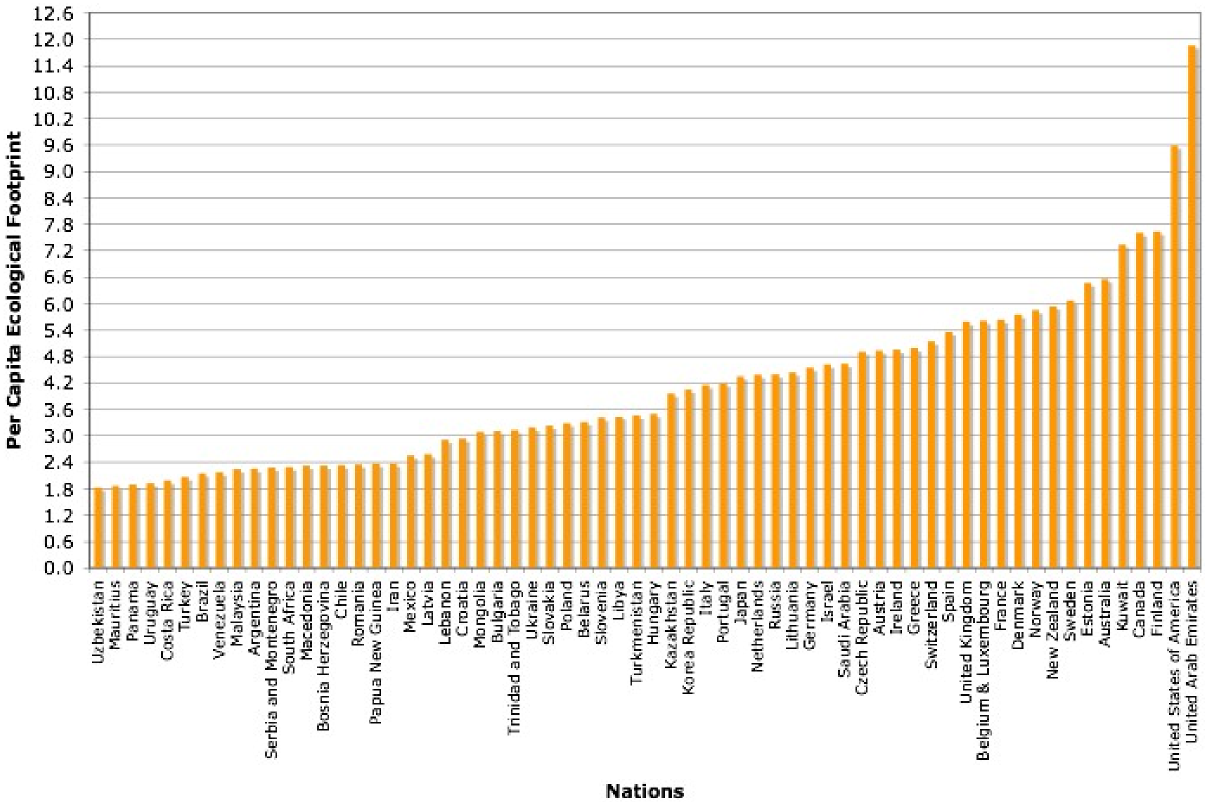

Figure 2 Debtor or Deficit Nations (GFN, 2006).

Cap-and-Trade of Ecological Footprint

Such controls and immodest aims would need to adhere to several policy principles. Nations with large per capita ecological footprints would not be able to adapt overnight due to technical, economic, social, and political constraints. Thus, implementation of this proposal would need to be phased in over some substantial time period. In the spirit of relying on incentives and incorporating flexibility into regulatory structures, we propose to implement a cap-and-trade strategy. Nations consuming beyond their footprint allocation would be free to meet their obligations through a combination of reductions in ecological footprint and purchase of footprint from creditor nations. Creditor nations would be free to sell "consumption credits" to debtor nations. The dynamics of this market would, of course, structure the price of credits and the nature and rate of investment in footprint reduction.

Such an agreement would need to incorporate substantial flexibility to allow countries to pursue implementation consistent with their values, strategies, and assets. While significant opportunities for low-cost improvements in resource efficiency and biocapacity (i.e., ecological restoration) are available in industrialized societies (Hawkin et al. 1999), improvements are often cheaper in developing nations (Grubb et al. 1999; Oberthur & Ott, 1999; Illum & Meyer, 2004). Consistent with the Kyoto Protocol, we allow for Joint Implementation (JI), a mechanism that permits a recipient country to "receive additional funds, modern technology and know-how, whereas the investing country would acquire (CO_2) credits at a lower cost than taking action at home" (Oberthur & Ott, 1999). Presumably, JI would support reductions in consumption at the global scale at lower aggregate cost and would accelerate technology transfer to developing nations.[6]

Our proposal includes a mechanism for annual update of each nation's rights and responsibilities based on changes in their 1) consumption, 2) technology, 3) population, and 4) biocapacity (i.e., ecological restoration that results in expanded carrying capacity). These updates represent feedback loops such that unsustainable investments and patterns of behavior are costly, while progress toward sustainability yields rewards.

[6] JI is further supported by the uneven global distribution of biocapacity. For example, biodiversity "hotspots" are essential resources and should be maintained.

We propose that controls on footprint be phased in over time. Deficit nations will be required to reduce their deficit footprints by 5% per year until they reach a sustainable level of consumption. Creditor nations, in other words nations with a per capita footprint of less than an earthshare, will be required to stay within the limit of an earthshare per person. Such nations could expand resource usage up to that threshold, but not exceed it without incurring debtor-nation responsibilities.

Simulation

We developed a mathematical model to simulate convergence of ecological footprints of debtor and creditor nations under the proposed agreement. The simulation is structured as follows.

1. All nations' footprints are normalized in terms of deviation from the allotted 1.8 gha earthshare per capita.[7] For example, the United States has an average per capita ecological footprint of 9.6 gha, which results in a +7.8 gha deviation from the allotted 1.8 gha per capita. Similarly, Cambodia with a per capita footprint of 0.7 gha has a deviation from a per capita earthshare of -1.1 gha.
2. Average per capita consumption converges across the globe at the level of nation states on the basis of annual incremental changes. All nations eventually reach and maintain consumption at the rate of an earthshare per capita, defined as a per capita national average ecological footprint of 1.8 gha. This expectation applies to both deficit nations (that are consuming above 1.8 gha per capita) and surplus nations (that are consuming below 1.8 gha per capita).
3. Debtor nations are required to reduce deficit ecological footprint by 5% annually. In each year, 2.769% of this 5% obligation will be achieved through real, material reductions and 2.231% will be achieved through purchase of consumption credits. These specific values derive from our mathematical simulation (see below), but our intention is to require rich nations to meet roughly half of their annual obligations in the initial years of the agreement through real internal reductions rather than by simply purchasing credits.[8] Real reductions can occur in consumption, population, and/or via higher productivity (i.e., efficiency of production and consumption).
4. JI is not considered in this simulation and we treat population and biocapacity as constants.
5. The sale of an ecological footprint credit provides an offset to the purchasing nation and income to the selling nation for a period of 20 years. This 20-year contract provides flexibility for deficit nations to reduce consumption (through long-term internal reduction strategies) and creates an annual income stream to surplus nations to support sustained investment in security, infrastructure, human capital, and other resources required for economic development and ultimately increased consumption. While the real reduction or increase of ecological footprint following the sale of credits will be realized over the 20-year contract period, as an accounting convention we record footprint increases and reductions in the year following the sale of the credit.

Rich nations' commitment to annual 5% footprint reductions drives the dynamic model that we simulate. As stated above, the percentage of reduction that debtor countries (i.e., those with per capita ecological footprints > 1.8 gha) must meet by real, material reductions is fixed, as is the percentage they purchase from surplus nations (i.e., those with per capita ecological footprints < 1.8 gha). These values are determined via *post optimization*, an iterative feedback method for improving parameter convergence. Note that because rich nations reduce their footprint deficit every year, the number of credits that they purchase declines over time. The percentage remains fixed, but the volume of credits changes.

We find that pegging real footprint reduction requirements at 0.02769 every year and allowing "debtor" nations to purchase 0.02231 of their earthshare debt in the form of credits results in convergence. Specifically, all nations achieve average per capita ecological footprints of an earthshare (1.8 gha) in 136 years (Figures 3 & 4). Changes in nations' consumption over the 136-year period are not linear. For example, in the first 26 years of the agreement, deviations from an earthshare for both deficit and surplus nations are reduced by 69%. The curves in the figures derive from plotting every nation's footprints at 25-year intervals. For purposes of clarifica-

[7] For further details, refer to http://www.footprintnetwork.org.

[8] Note that a policy that did not require real reductions in ecological footprint could simply result in a transfer of wealth from rich to poor nations and thus fail to significantly mitigate overshoot. In contrast, a scheme that rested solely on mandatory real reductions and did not allow for purchase of credits would fail to fuel sustainable development in poor nations. Allowing rich nations to meet their obligations through variable combinations of real reductions and credit purchases would increase flexibility. Under a modified policy proposal, it is easy to imagine allowing a liberalized trading regime under which rich nations could make real reductions in ecological footprint beyond what is required in a given year and then sell their rights to purchase credits. For present purposes, we hold the purchase and real reduction fractions constant.

tion, the x-axis explicitly identifies the positions of a set of nations consuming at very different levels. Figure 4 zooms in on the later years of policy implementation, as these details cannot be seen at the scale of Figure 3.

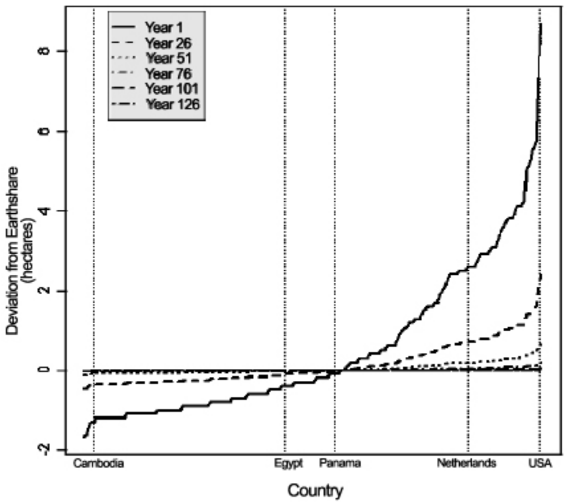

Figure 3 Graphic simulation of convergence of global ecological footprint as demonstrated by plots at 25-year intervals.

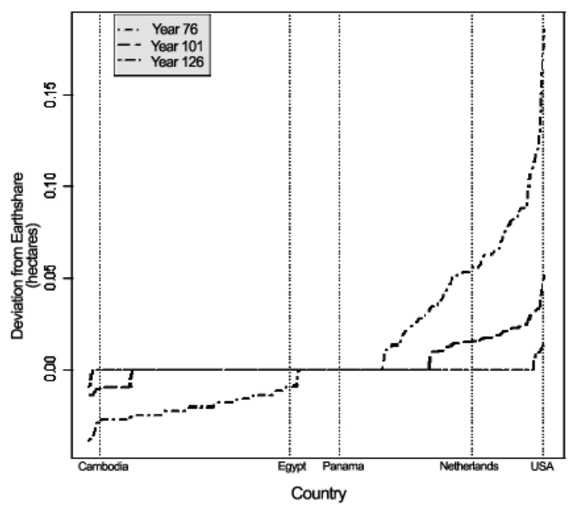

Figure 4 Close up of graphic simulation of global ecological footprint convergence in latter years under proposed agreement as demonstrated by plots at 25-year intervals.

Figure 5 illustrates the dynamic rate of footprint credits for sale by surplus nations over time. In the initial years under the policy, approximately up until year 60, these nations would sell close to 5% of their stock of credits each year, almost certainly resulting in a very significant income stream. Over the following 65-year period, the rate of sales drops off rapidly. In the final years of implementation, the stock of credits approaches zero as surplus nations advance toward consumption at rates approximating an earthshare per capita. Simultaneously, as debit nations approach 1.8 gha per person footprints, they eventually come to a point of purchasing zero credits to meet their annual obligations, yielding a zero rate of sale.[9]

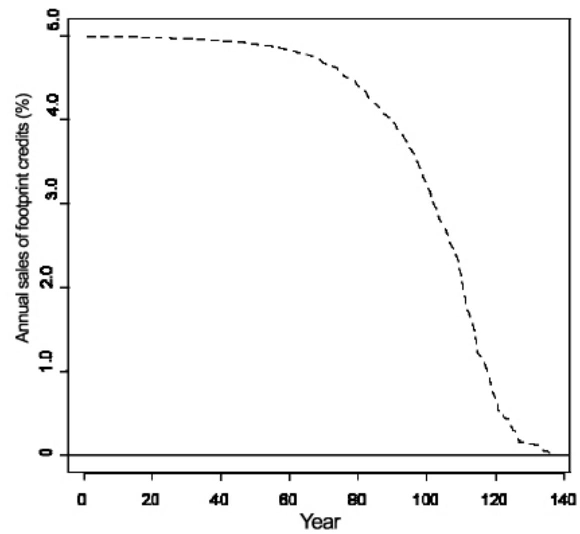

Figure 5 Annual rate of sales of ecological footprint credits as a percentage of total stock of credits initially held by surplus nations.

Domestic Implementation

Up to this point, our discussion has only addressed obligations and opportunities at the nation-state level. Within individual nations, it is possible to imagine allocating rights to an earthshare to each citizen and then implementing a cap-and-trade system in parallel to the global system we have just described. Citizens who exceed an earthshare would be required to conserve and/or purchase surplus footprints from poorer or more frugal fellow citizens. Thus, the distribution of property rights corresponding to an earthshare could result in large monetary transfers between rich and poor individuals within nations. Such transfers could promote domestic poverty alleviation and possibly a significant redistribution of wealth within countries. As every nation would have the ability to determine how it was going

[9] In computing, sequences that converge to 0 require special treatment. In most cases, dividing small numbers by small numbers iteratively will not yield zero. The values do become infinitesimally small. For our purposes we assigned a value of zero to numbers that drop below 0.009. Our iterative simulation is stopped, and zero values are assigned, when all countries have a per capita footprint deviation from an earthshare of 0.009 or less.

to meet the agreement requirements, elites would likely push the costs of compliance on the poor. A significant element of our proposal's immodesty rests precisely in our not addressing the inequities, repression, and violence that would accompany implementation.

Conclusion

The proposal we outline here would rationalize and equalize global ecological footprint at the level of individual nation states. This proposed international treaty would conserve natural capital by limiting the freedom of rich nations to degrade the environment and amplify global risks. With respect to current and future generations, our proposal advances distributive justice and reduction of ecological debt. Significant infusions of funding into developing nations would reduce the pressure on impoverished people to further degrade their natural capital while providing investment capital to support development. Additionally, the agreement would create incentives for innovations that enhance resource productivity. Such expanded efficiency could fuel sustainable economic development and material wellbeing.

Beyond self-interest in mitigating risks that stem from global environmental degradation, we have argued that concentrated appropriation of natural resources cannot be sustained on ethical grounds. There is no right to withdrawals in cases where they exceed a proportionate share and leave others worse off. On the basis of these arguments, nations consuming more than an "earthshare" per person, as determined by the ecological footprinting methodology, are to bear the costs of the proposed policy.

Our proposal allows us to contemplate how contemporary perspectives on environmental policy play out when applied to overarching environmental issues. It has been suggested that environmental policies capable of addressing such problems must be flexible, efficient and accountable, information-rich, innovation spurring, incentive-based, transmedia (i.e., integrate management across soil, air, and water), and applicable at multiple levels of social organization (i.e., nested) (Kettl, 1998). In keeping with these criteria, our proposal features tradable permits and JI. We reward innovation and provide flexibility to actors to structure their own investment schedule within a regulatory framework premised on clear metrics. Annual updating of rights and responsibilities constitutes critical information signals. In sum, our proposal very much accords with current policy-design principles.

This analysis is obviously limited in many ways; for example, ecological footprinting methodologies do not encompass nonrenewable resource consumption. Further, our policy would create perverse incentives. For example, because conversion of native forestland to pastureland would increase biocapacity and allowable footprint, this approach would sustain the economic-benefit stream from the environment, but it would fail to address all environmental values (Lenzen et al. 2007). Lastly, it is important to note that the NFA methodology that we employ cannot be directly translated into material-consumption rates. While nations' per capita ecological footprints would converge under our policy agreement, differences in the technical efficiency of production and consumption across nations would remain, resulting in varying levels of health and welfare. Further, equalizing the ecological footprints of nation states does not, of course, ensure individuals' equal access to resources. As every country would have the responsibility to determine how to meet the requirements of the agreement, in the absence of controls, there is a significant risk that elites would push the costs and burdens of compliance onto poor, politically marginalized people.

Despite these shortcomings, as a thought exercise, the analysis confronts us with the enormity of the challenge ahead if we take sustainability seriously. At the same time, we are able to begin to explore the material meaning of commitment to sustainability (Blühdorn & Welsh, 2007) and social coordination strategies that would allow us to live within our collective means (Cohen, 2006). Phrased this way, it does not sound like such an immodest proposal.

By making an outrageous suggestion, Swift shocked people of all stripes, thereby raising the visibility of poverty in Ireland and its institutional roots. He gave interest groups with little in common a shared target of mockery and he bounded the solution set, accelerating political processes of negotiation and collective action. We would be pleased if our modest proposal accomplishes any or all of these outcomes. In direct contrast with the intervention Swift proposed, we note that the policy instrument we advocate—establishment of new enclosures and reliance on markets—is increasingly perceived as an efficient and fair solution to problems of social and material coordination and not at all outrageous. In this sense, our proposal could be interpreted to be pragmatic rather than immodest. But, we want to make clear that we are not at all convinced of the wisdom of global commoditization of biocapacity. We are committed to the "ends" we outline, but are quite conflicted with respect to the "means." In our view, the obscenity of contemporary ecological degradation and human suffering is perhaps rivaled by the audacity of this institutional innovation.

Acknowledgement

We gratefully acknowledge constructive critique and suggestions of anonymous referees and Maurie Cohen on earlier versions of this paper.

References

Attfield, R. 1999. *The Ethics of the Global Environment*. Edinburgh: Edinburgh University Press.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., & Turner, R. 2002. Economic reasons for conserving wild nature. *Science* 297(5583):950–953.

Beckerman, W. & Pasek, J. 2001. *Justice, Posterity, and the Environment*. New York: Oxford University Press.

Berkhout, F. 2002. Technological regimes, path dependency, and the environment. *Global Environmental Change* 12(1):1–4.

Bicknell, K., Ball, R., Cullen, R., & Bigsby, H. 1998. New methodology for the ecological footprint with an application to the New Zealand economy. *Ecological Economics* 27(2):149–160.

Blühdorn, I. & Welsh, I. 2007. Eco-politics beyond the paradigm of sustainability: A conceptual framework and research agenda. *Environmental Politics* 16(2):185–205.

Carley, M. & Spapens, P. 1998. *Sharing the World: Sustainable Living & Global Equity in the 21st Century*. New York: St. Martin's Press.

Catton, W. 1980. *Overshoot: The Ecological Basis of Revolutionary Change*. Urbana: University of Illinois Press.

Chambers, N., Simmons, C., & Wackernagel, M. 2000. *Sharing Nature's Interest: Ecological Footprints as an Indicator of Sustainability*. London: Earthscan.

Cohen, M. 2006. Sustainable consumption research as democratic expertise. *Journal of Consumer Policy* 29(1):67–77.

Costanza, R., d'Arge, R., de Groot, R., Farberi, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R., Paruelo, J., Raskins, R., Sutton, P., & van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387(6230):253–260.

Crowley, K. 1999. Jobs and environment: the "double dividend" of ecological modernization? *International Journal of Social Economics* 26(7–9):101–102.

Daly, H. & Cobb, J. 1989. *For The Common Good: Redirecting the Economy Toward Community, the Environment, and a Sustainable Future*. Boston: Beacon Press.

Galloway, J. 2000. Nitrogen mobilization in Asia. *Nutrient Cycling in Agroecosystems* 57(1):1–12.

Global Footprint Network (GFN). 2006. National Footprints: Data and Methods. http://www.footprintnetwork.org/gfn_sub.php?content=datamethods. April 12, 2007.

Global Footprint Network (GFN). 2007. Carbon Footprint: http://www.footprintnetwork.org/gfn_sub.php?content=app_carbon_footprint. February 6, 2008.

Grubb, M., Vrolijk, C., & Brack, D. 1999. *The Kyoto Protocol: A Guide and Assessment*. London: The Royal Institute of International Affairs, Energy and Environment Programme.

Hails, C. (Ed.). 2006. *Living Planet Report 2006*. Gland, Switzerland: WWF International. http://assets.panda.org/downloads/living_planet_report.pdf.

Hancock, J. 2003. *Environmental Human Rights: Power, Ethics, and Law*. Burlington, VT: Ashgate.

Hawken, P., Lovins, A., & Lovins, L. 1999. *Natural Capitalism: Creating the Next Industrial Revolution.* New York: Little, Brown and Company.

Holmberg, J., Lundqvist, U., Robert, K.-H., & Wackernagel, M. 1999. The ecological footprint from a systems perspective of sustainability. *International Journal of Sustainable Development and World Ecology* 6(1):17–33.

Illum, K. & Meyer, N. 2004. Joint implementation: methodology and policy considerations. *Energy Policy* 32(8):1013–1023.

Kettl, D. (Ed.). 1998. *Environmental Governance: A Report on the Next Generation of Environmental Policy*. Washington, DC: Brookings Institution Press.

Lenzen, M. & Murray, S. 2001. A modified ecological footprint method and its application to Australia. *Ecological Economics* 37(2):229–255.

Lenzen, M. & Murray, S. 2003. *The Ecological Footprint: Issues and Trends*. ISA Research Paper 01-03. Australia: Centre for Integrated Sustainability Analysis, The University of Sydney. http://www.isa.org.usyd.edu.au/publications/documents/Ecological_Footprint_Issues_and_Trends.pdf.

Lenzen, M., Hansson, C., & Bond, S. 2007. On the bioproductivity and land-disturbance metrics of the Ecological Footprint. *Ecological Economics* 61(1):6–10.

Loh, J. (Ed.). 2002. *Living Planet Report 2002*. Gland, Switzerland: WWF International. http://assets.panda.org/downloads/lpr2002.pdf.

Loh, J. & M. Wackernagel (Eds.). 2004. *Living Planet Report 2004*. Gland, Switzerland: WWF International. http://assets.panda.org/downloads/lpr2004.pdf.

Marx, A. 2000. Ecological modernization, environmental policy and employment. Can environmental protection and employment be reconciled. *Innovation* 13(3):311–325.

Mazmanian, D. & M. Kraft (Eds.). 1999. *Toward Sustainable Communities: Transition and Transformations in Environmental Policy*. Cambridge, MA: MIT Press.

Millennium Ecosystem Assessment (MEA). 2005. *Ecosystems and Human Well-Being: Synthesis.* Washington, DC: Island Press. http://www.millenniumassessment.org/documents/document.356.aspx.pdf.

Mol, A. 2000. Globalization and environment: between apocalypse-blindness and ecological modernization. In G. Spaargaren, A. Mol, & F. Buttel (Eds.), *Environment and Global Modernity*. pp. 121–149. Thousand Oaks, CA: Sage.

Mol, A. & D. Sonnenfeld (Eds.). 2000. *Ecological Modernisation Around the World: Perspectives and Critical Debates*. Portland, OR: Frank Cass.

Mol. A. 2001. *Globalization and Environmental Reform*. Cambridge, MA: MIT Press.

Mol, A. 2003. Ecological modernization: industrial transformations and environmental reform. In C. Humphrey, T. Lewis, & F. Buttel (Eds.), *Environment, Energy, and Society: Exemplary Works*. pp. 401–412. Belmont, CA: Wadsworth.

National Research Council (NRC). 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, DC: National Academies Press.

Oberthur, S. & Ott, H. 1999. *The Kyoto Protocol: International Climate Policy for the 21st Century*. New York: Springer.

Pogge, T. 1998. A global resources dividend. In D. Crocker & T. Linden (Eds.), *Ethics of Consumption: The Good Life, Justice, and Global Stewardship*. pp. 501–536. Lanham, MD: Rowman & Littlefield.

Rees, W. 2000. Eco-footprint analysis: merits and brickbats. *Ecological Economics* 32(3):371–374.

Renascence Editions. 1999. Jonathan Swift, A Modest Proposal (1729). http://www.uoregon.edu/~rbear/modest.html. June 26, 2007.

Roberts, J. 2003. Global inequality and climate change. In C. Humphrey, T. Lewis, & F. Buttel (Eds.), *Environment, Energy, and Society: Exemplary Works*. pp. 254–262. Belmont, CA: Wadsworth.

Sachs, W. 2004. Sustainable development. In M. Redclift & G. Woodgate (Eds.), *The International Handbook of Environ-*

mental Sociology. pp. 71–82. Northampton, MA: Edgar Elgar.

Sachs, W. & T. Santarius (Eds.). 2007. *Fair Future: Resource Conflicts, Security, and Global Justice.* London: Zed.

Shue, H. 1995. Ethics, the environment and the changing international order. *International Affairs* 71(3):453–461.

Simms, A. 2005. *Ecological Debt: The Health of the Planet and the Wealth of Nations*. London: Pluto Press.

Speth, J. & P. Haas (Eds.). 2006. *Global Environmental Governance*. Washington, DC: Island Press.

Tilman, D. 1999. Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proceedings of the National Academy of Sciences* 96(11):5995–6000.

Turner, K., Lenzen, M., Wiedmann, T., & Barrett, J. 2007. Examining the global environmental impact of regional consumption activities–Part 1: A technical note on combining input-output and ecological footprint analysis. *Ecological Economics* 62(1):7–44.

United Nations. 1948. Universal Declaration of Human Rights. http://www.un.org/Overview/rights.html. June 9, 2007.

United Nations. 1976. International Covenant on Economic, Social and Cultural Rights. http://www.unhchr.ch/html/menu3/b/a_cescr.htm. June 9, 2007.

Venetoulis, J., Chazan, D., & Gaudet, C. 2004. *Ecological Footprint of Nations*. Oakland, CA: Redefining Progress. http://www.rprogress.org/publications/2004/footprintnations2004.pdf.

Wallach, L. & Woodall, P. 2004. *Whose Trade Organization? A Comprehensive Guide to the WTO*. New York: New Press.

Wackernagel, M., Monfreda, C., Moran, D., Wermer, P., Goldfinger, S., Deumling, D., & Murray, M. 2005. *National Footprint and Biocapacity Accounts 2005: The Underlying Calculation Method*. Oakland, CA: Global Footprint Network. http://www.footprintnetwork.org/download.php?id=5.

Wackernagel, M. & Rees, W. 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. *Ecological Economics* 20(1):3–24.

Wiedmann, T., Minx, J., Barrett, J., & Wackernagel, M. 2006. Allocating ecological footprints to final consumption categories with input-output analysis. *Ecological Economics* 56(1):28–48.

Wiedmann, T., Lenzen, M., Turner, K., & Barrett, J. 2007a. Examining the global environmental impact of regional consumption activities–Part 2: Review of input-output models for the assessment of environmental impacts embodied in trade. *Ecological Economics* 61(1):15–26.

Wiedmann, T., Lenzen, M., Turner, K., Minx, J., & Barrett, J. 2007b. *Multiregional Input-Output Modeling Opens New Opportunities for the Estimation of Ecological Footprints Embedded in International Trade*. International Ecological Footprint Conference. May 8–10, Cardiff.

World Bank. 2008. Understanding Poverty. http://go.worldbank.org/RQBDCTUXW0. February 2, 2008.

York, R., Rosa, Eugene A., & Dietz, T. 2003. Footprints on the earth: the environmental consequences of modernity. *American Sociological Review* 68(2):279–300.

Sustainability: Science, Practice, & Policy
http://ejournal.nbii.org

ARTICLE

Rethinking the economist's evaluation toolkit in light of sustainability policy

Stefan Hajkowicz

CSIRO Sustainable Ecosystems, 306 Carmody Road, St. Lucia, QLD 4067 Australia (email: Stefan.Hajkowicz@csiro.au)

The dominant economic evaluation technique is benefit-cost analysis (BCA). However, sustainability policy must handle outcomes that cannot easily be quantified in monetary units. Multiple criteria analysis (MCA) is emerging as an alternative, and/or complementary, economic evaluation tool. The economics profession has been slow to adopt MCA. This paper first explores the role of MCA within the economist's evaluation toolkit alongside BCA, cost-effectiveness analysis (CEA), and cost-utility analysis (CUA) and then proposes a process for selecting an appropriate evaluation method. The choice of technique will depend on the extent to which environmental goods can be valued in monetary units. The paper argues that MCA has an expanded role to play alongside BCA (and the other methods) to ensure that sustainability policies are realized.

KEYWORDS: cost analysis, economic analysis, environmental economics, resource management, management tools, policy, evaluation

Introduction

Multiple criteria analysis (MCA) is an evaluation framework that ranks or scores the performance of decision options (e.g., policies, projects, locations) against multiple objectives measured in different units. Typically, the criteria are weighted by decision makers to reflect their relative importance.[1] The MCA approach emerged within the field of operations research during World War II, with early applications in military planning (e.g., Eckenrode, 1965). The MCA's theoretical foundations can be traced back to multiattribute utility theory (MAUT) developed by Keeney & Raiffa (1993) and axioms of utility measurement first supplied by von Neumann & Morgenstern (1944).

In environmental and resource economics, MCA has mostly received a positive reception. Many researchers find it a useful supplement to conventional benefit-cost analysis (BCA) when intangible nonmarket goods are important (Eder et al. 1997; Heilman et al. 1997; Joubert et al. 1997; Prato, 1999; Fernandes et al. 1999; Dunning et al. 2000). MCA has hundreds of applications in natural resource management (for reviews see Romero & Rehman, 1987; Hayashi, 2000). However, not all resource economists are convinced. For example, Bennett (2005) refers to MCA as an "avoidance strategy" to sidestep a rigorous and complete BCA.

Given that MCA application is becoming increasingly common, such criticisms are worth exploring. This paper examines MCA's role in the economic appraisal of policy options in light of sustainability requirements.[2] It argues that MCA is a valid and useful evaluation tool for sustainability appraisal when nonmarket impacts are important.

What is Multiple Criteria Analysis (MCA)?

The use of MCA to support public and private sector policy decisions has steadily grown since the 1970s. Two decades ago, Romero & Rehman (1987) reviewed 150 MCA applications in fisheries, forestry, water, and land resource applications. More recently, Hayashi (2000) reviewed over 80 published studies in agriculture. In energy planning, Pohekar & Ramachandran (2004) identify more than 90 published MCA applications. Steuer & Na (2003) examine 265 applications of MCA in the field of financial decision making. Today there are hundreds of MCA techniques (for a recent review see Figueira et al. 2005a) and Weistroffer et al. (2005) identify 81 MCA software packages, many of which are commercially available.

[1] Criteria are defined here as the attributes (or indicators) used to measure performance against the decision makers' objectives.

[2] Options are defined here as the items (alternatives) being chosen, ranked, or scored by the decision maker.

An MCA model can be represented with an evaluation matrix (X) of n options and m criteria. The evaluation matrix contains performance measures where $x_{i,j}$ is the raw performance score assigned to option i against criterion j. Typically, though not always, the relative importance of criteria is measured with a weights vector W where w_j represents the importance of the j^{th} criterion. Both W and X may contain qualitative (ordinal) or quantitative (cardinal) data. An evaluation matrix is often structured as follows:

	Option i=1	*Option i=2*	*Option i=n*
Criterion j=1	$x_{i=1,j=1}$	$x_{i=2,j=1}$	$x_{i=n,j=1}$
Criterion j=2	$x_{i=1,j=2}$	$x_{i=2,j=2}$	$x_{i=n,j=2}$
Criterion j=m	$x_{i=1,j=m}$	$x_{i=2,j=m}$	$x_{i=n,j=m}$

An MCA model always has at least two criteria and two options. If the purpose of the MCA is discrete choice, i.e., to select one or more options, an initial check can be made for strict dominance, that is, for options that are outperformed by another option on all criteria. If $v_{i,j}$ is the transformed performance score (where a higher value is better) of $x_{i,j}$, option i can be considered strictly dominated by i' if:

$$v_{i',j} \geq v_{i,j} \text{ for all } j = 1,\ldots,m \text{ and}$$
$$v_{i',j} > v_{i,j} \text{ for some } j=1,\ldots,m.$$

The pretest for strict dominance can sometimes make the decision analysis elementary, negating the requirement for more advanced MCA models.

The stages of MCA (Figure 1) include:

1. *Problem structuring:* This crucial stage of MCA, typically requiring the bulk of the effort, involves the identification of criteria and decision options and obtaining performance measures (Janssen, 2001).
2. *Criteria weighting:* This involves obtaining information from decision makers about the relative importance of criteria. Weights may be expressed at either an ordinal or cardinal measurement level.
3. *Criteria transforming:* As the criteria are in different units they need to be transformed into commensurate units prior to aggregation in the ranking or scoring function.
4. *Option ranking and/or scoring:* The weights and transformed performance measures are combined to determine the overall performance of each option, relative to other options.
5. *Sensitivity analysis and decision making:* Variation of MCA methods, performance measures, and weights test the sensitivity of the result. The decision maker(s) can then make a final choice.

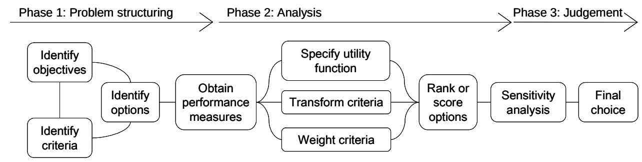

Figure 1 The multiple criteria analysis decision making process (adapted from Hajkowicz, 2003).

Among a wide variety of MCA algorithms available to attain a final ranking or scoring of the decision options, some of the more common are the Analytic Hierarchy Process (AHP) (Saaty, 1987), weighted summation (Figueira et al. 2005b), ELECTRE (Roy, 1968; Figueira et al. 2005b), PROMETHEE (Brans et al. 1986), and Compromise Programming (Zeleny, 1973; Abrishamchi et al. 2005).[3] These are a few of the many different methods to "solve" an MCA problem. It has been shown that changing the method can alter the result, although the differences are typically minor (Gershon & Duckstein, 1983; Ozelkan & Duckstein, 1996; Eder et al. 1997; Raju et al. 2000). Choosing the best MCA method for a given task is a considerable challenge (Tecle, 1992). Consideration needs to be given to the measurement scale of evaluation data (ordinal or cardinal), the nature of criteria transformations, the presence of uncertain input data, the existence of intercriterion dependencies, the number of decision makers (individual, group, or society), and how decision makers would like to interact with the decision model.

Arguably, the most commonly applied MCA technique, possibly by virtue of its relative ease of computation, is linear-weighted summation (Howard, 1991; Zanakis et al. 1998). This approach determines overall performance scores for decision options (u_i) by:

$$u_i = \sum_{j=1}^{m} v_{i,j} w_j \qquad (1)$$

where:

$$\sum_{j=1}^{m} w_j = 1;$$
$$0 < w_j \leq 1;$$

[3] See Figueira (2005a) for a more detailed review of MCA algorithms.

$$v_{i,j} = \frac{x_{i,j} - \min\limits_{i=1}^{n}(x_{i,j})}{\max\limits_{i=1}^{n}(x_{i,j}) - \min\limits_{i=1}^{n}(x_{i,j})} \qquad (2)$$

$\max\limits_{i=1}^{n}(x_{i,j})$ = the maximum value of $x_{i,j}$ for $i = 1,\ldots n$;

and

$\min\limits_{i=1}^{n}(x_{i,j})$ = the minimum value of $x_{i,j}$ for $i = 1,\ldots n$.

Alternative Economic Evaluation Frameworks

The appropriateness of MCA depends upon the suitability of other economic evaluation frameworks. Four main economic evaluation frameworks are available:

- Benefit-cost analysis (BCA)
- Cost-effectiveness analysis (CEA)
- Cost-utility analysis (CUA)
- Multiple criteria analysis (MCA)

A CEA can be performed when the benefits of the decision options are adequately measured by a single unit, e.g., tons of soil. Costs in CEA are still computed with standard discounted cash flow (DCF) analysis. The aim is to identify the option which (a) achieves a target outcome at least cost; or (b) maximizes the outcome measure subject to cost constraint.

In CUA, the costs are still computed via standard DCF, but the benefits are measured by multiple attributes in different units. CUA emerged in the early 1980s in healthcare economics (Drummond et al. 1997). Today, Quality Adjusted Life Years (QALYs) are routinely calculated to measure the nonmarket benefits of patient treatment or healthcare programs. The attributes used to determine a QALY score are sensation, mobility, emotion, cognition, self care, pain, and fertility. These are weighted and collapsed into a single numeraire (unit of value) using multiattribute utility theory. The CUA approach is now well established in healthcare economics and has emerging application in environmental and resource economics (Cullen et al. 2001).

Although the term "CUA" is not used by Ribaudo et al. (2001), they describe how such an approach was applied to select conservation contracts under the United States Conservation Reserve Program (CRP). The benefits of contracts were measured with a multiattributed environmental benefits index (EBI). Combining the cost of each option with the EBI enabled purchasing decisions. The BushTender program in Victoria (Australia) is predicated on a similar concept and makes purchasing decisions on the basis of a biodiversity benefits index (BBI) and contract cost (Stoneham et al. 2003).

The process for choosing which of BCA, CEA, CUA, and MCA to apply depends largely on the valuation of benefits (Figure 2). If benefits are adequately measured in monetary units, then BCA provides an appropriate framework. If this is not the case, the analyst will need to contemplate nonmarket valuation (NMV), which will require attention to both reliability and cost effectiveness. If it is decided that NMV is not feasible or worthwhile, then CUA may be appropriate. If there is no monetary cost data, e.g., the options are strategic policy directions, then MCA can be used. It is also noted that MCA can be used with "cost" as one of the criteria.

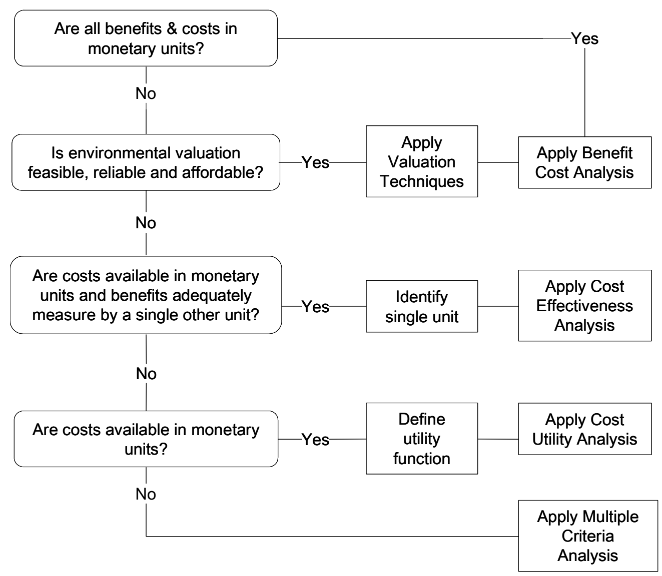

Figure 2 Process for choosing whether to use BCA, CEA, CUA or MCA.

This article argues that all four frameworks are solidly founded, able to measure benefits adequately, and potentially applicable in different situations. None is inherently better or more robust and all are based on solid theoretical foundations. The key determinant of which to use relates to valuation.

The Limits to Valuation?

The valuation of environmental resources has attracted considerable attention over the past several decades (Adamowicz, 2004). The appropriateness of different valuation techniques, and the suitability of valuation itself, has been heavily debated. There are three main approaches to valuing environmental resources.

First, cost savings and avoidance (CSA) is a set of valuation techniques that is limited to market impacts. It includes measures of preventative and mitigatory expenditure (e.g., Spurgeon, 1998), lost production (e.g., Hajkowicz & Young, 2005), ameliorative expenditure (e.g., Abdalla et al. 1992) and asset damage repair costs (e.g., Tol, 1996) as a consequence of an environmental problem. These analyses typically ask: "How much is the environmental problem costing?" Or, conversely, "How much is being saved because of the presence of a well functioning environment?"

Second, revealed preference techniques estimate the price of a nonmarket good from a closely related proxy market good. Hedonic pricing (e.g., Pearson et al. 2002) and the travel cost method (e.g., Chen et al. 2004) are types of revealed preference techniques. They assess the premium being paid in a real market (e.g., property market) to access a nonmarket environmental good (e.g., scenic views).

Third, stated preference techniques are based on hypothetical questions that are posed to survey respondents. Contingent valuation asks survey respondents about their willingness to pay (WTP) for environmental goods or willingness to accept (WTA) compensation for the loss of environmental goods (e.g., Carson et al. 2003). Choice modelling asks respondents to select bundles of environmental goods at different costs and infers prices from their choices (e.g., van Bueren & Bennet, 2004).

If the analyst considers these methods feasible, accurate, and comprehensive, then the flowchart recommends using BCA (Figure 2). While all three approaches provide effective tools for policy analysis, this paper argues that valuation has limitations.

Both CSA and revealed pricing are methodologically strong, but are limited in scope. In contrast, stated preference has practically limitless scope, but is methodologically weaker (Strijker et al. 2000). This means that not all outcomes can be valued in all cases.

CSA and revealed preferences source data from real markets and thereby avoid the methodological difficulties associated with surveys. The drawback with these techniques lies not in their methodology, but in their scope. CSA limits valuation to market impacts, excluding nonmarket goods such as landscape aesthetics and biodiversity preservation. For revealed pricing, scope may be limited due to unavailability of a proxy market for many environmental goods.

Stated preference techniques can broaden the scope of valuation. However, this introduces significant methodological difficulties associated with consumer-preference surveys (Sagoff, 1988; Diamond & Hausman, 1994; McFadden, 1999; Ludwig 2000; Whittington, 2002). Despite advances over time, two problems persist in stated preference survey designs: (a) the marketplace is hypothetical, which creates uncertainties about real consumer behavior; and (b) the respondent is often unfamiliar with, or unaware of, the environmental good under question. Ludwig (2000) observes:

> [P]eople are asked to place prices on things that are not ordinarily priced. For some commodities, we form an opinion about a suitable price from long experience in a market. If there is no such experience and no such market, there may be little consistency among responses and little validity in inferences drawn from the responses.

These methodological issues have hindered the use of stated preference valuations by policy makers. Adamowicz's (2004) comprehensive review of hundreds of valuation studies conducted since 1975 found that NMV results are seldom used in real policy decisions despite a vast number of academic studies. Greater use is made of valuations based on market prices, i.e., CSA approaches. This can be attributed to either a failure by policy makers to grasp the relevance of NMVs or to fundamental methodological problems of valuing highly intangible nonmarket goods.

Rather than attempting to express such intangibles in monetary units, the pathway to improved resource allocation may lie in alternative decision-making frameworks. In a review of valuation studies, Adamowicz (2004) concludes that:

> The most significant advance in environmental valuation may be to move away from a focus on value and focus instead on choice behavior and data that generate information on choices. Advances in resource allocation are most likely to arise from better understanding of preferences and choice, rather than the generation of more value estimates and catalogues of these measures.

The fields of decision theory and MCA place the focus on choice behavior. They aim to provide tools and processes to help decision makers resolve trade-offs in a transparent, auditable, and analytically robust manner. In MCA the emphasis is on decision making and value measurement is a means to that end.

Potential Pitfalls (and Solutions) in Using MCA

While nonmarket valuation involving stated preferences has methodological problems, MCA is not a panacea. Some of the common sources of error associated with MCA are:

1. *Incorrect problem structure*: The selection of criteria and options to guide the MCA process (i.e., problem structuring) is typically the most crucial analytical task (Janssen, 2001). MCA failures can usually be traced back to poor problem definition. New research into MCA is developing improved means of selecting options and criteria. Mingers & Rosenhead (2004) review several "problem structuring methods" (PSMs). Scheubrien & Zionts (2006) developed an interactive computer model to assist with problem-structuring tasks.
2. *Poor performance data:* If the performance measures populating the MCA matrix are inaccurate, the results will also be inaccurate. Sensitivity analysis can help determine the extent to which performance-data uncertainties actually matter (i.e., change the overall ranking) (Kangas et al. 2000; Hyde et al. 2004). Sometimes there is over reliance on qualitative performance measures such as expert-judgment scores. These can be used where there is no other data source, but Keeney & Raiffa (1993) consider quantitative performance criteria preferable. Saaty's (1987) AHP is an MCA technique designed to elicit expert judgments when quantitative data are unavailable.
3. *Inappropriate capturing of decision-maker preferences:* The weighting task is complex and can be misunderstood by decision makers. A wide range of weighting methods are available to MCA analysts (Hajkowicz et al. 2000; Roberts & Goodwin, 2002). Edwards & Barron (1994) argue for the use of "swing weights" where decision makers take criterion ranges (minimum and maximum performance scores) into account when assigning weights.
4. *Incorrect application of additive utility:* Often a linear additive model provides a reasonable utility function. However, there are some cases where criteria are noncompensatory, for instance when strong performance on one criterion (e.g., in stream zinc concentrations) does not compensate for poor performance on another (e.g., in stream arsenic concentrations). Debate about additive utility models is illustrated by the United Nation's Human Development Index (HDI).[4] Sagar & Najam (1998) argue the HDI should be computed by a multiplicative utility function as opposed to the current additive form.
5. *Duplicate or overlapping criteria*: This difficulty occurs when two or more criteria measure the same underlying attribute. Duplicates can sometimes be detected by searching for high intercriterion correlations. There are MCA methods designed to overcome these problems (Brauers, 2004). One approach is to search for unusually high correlations between the criteria; this could suggest they are measuring the same underlying trend.

Some of these issues were highlighted in the Netherlands when the nation's highest administrative court (The Council of State) overruled an MCA that government agencies had used to select a hazardous waste site. The judge's verdict centered on inappropriate MCA methodology, including a failure to set appropriate weights, poor quality data in the evaluation matrix, and inappropriate transformation functions (Janssen, 2001). None of these represented a problem with the MCA technique itself, but rather how it was applied. As with any analytical tool, imperfect MCAs will result from real world constraints like limited data and time. The field of MCA is evolving rapidly, with many new tools and software packages to help analysts and decision makers avoid methodological pitfalls (Brauers, 2004).

Comparing MCA and BCA

Several researchers have applied both MCA and BCA to the same natural resource management problem and then compared the results (Joubert et al. 1997; Strijker et al. 2000; Brauers, 2004; Brouwer & van Ek, 2004). These studies show no clear conclusion that either approach is "better," rather, both have strengths and weaknesses. Strijker et al. (2000) argue that alternatives to BCA are "next-best solutions," but are, nevertheless, required due to practical and methodological drawbacks with environmental valuation. They propose "minimizing the disadvantages of both methods" by using BCA results within the MCA. In a water-planning problem in Cape Town, South Africa, Joubert et al. (1997) take a similar position, suggesting that BCA and MCA are complementary tools.

[4] The Human Development Index (HDI) has been published by the United Nations Development Program since 1990 for each country. Providing an alternative measure of whole-of-country performance to gross domestic product, it is defined by indicators of educational status, life expectancy, and a logarithmically adjusted measure of income.

The differences between BCA and MCA are summarized in Table 1. Arguably, the main difference is how the criteria weights are set and whose preferences are used. In BCA, weights are derived from the marketplace while in MCA weights are specified by decision makers. Also, when an MCA contains a cost criterion, it may be possible to compute the marginal rate of substitution for monetary (versus nonmonetary) outcomes. Some decision-support methods attempt to make this tradeoff explicit (Hammond et al. 1998).

Table 1 The Differences Between MCA and BCA

MCA Task	BCA Comparison
1. Problem structuring	In BCA the criteria are already identified, they are the benefit and cost items, but identification of decision options and obtaining performance measures are part of the process.
2. Criteria weighting	For BCA, weighting is by market prices, or surrogates. Hence, the weights are determined by aggregate consumer preferences, and not by decision-maker preferences, which may not be representative of societal preferences.
3. Transforming criteria	In BCA all outcomes are measured in dollar units, so the transformation is already completed via the valuation process.
4. Ranking and/or scoring options	The scoring in BCA is on the basis of net present value (NPV), benefit-cost ratio (BCR), or internal rate of return (IRR).
5. Conducting sensitivity analysis and making a decision.	The same is done in BCA. Input parameters such as the discount rate are systematically varied, and the impact on the result is assessed.

Where nonmarket values prevent application of BCA, MCA has been shown to help decision makers learn and make transparent, auditable choices in what would otherwise be unstructured decisions (Prato, 1999; Hajkowicz et al. 2000; Hayashi, 2000; Robinson, 2000). MCA conforms to formal axioms of multiattribute utility theory (Keeney & Raiffa, 1993), which Schultz (2001) argues would have improved the rigor and internal consistency of the United States Environmental Protection Agency's Index of Watershed Indicators were it applied.

Conclusion

As with any evaluation tool, MCA has bounded scope for application and introduces methodological challenges of its own. The common obstacles, and potential sources of error, in MCA applications are choosing the criteria and options; avoiding redundant (duplicate) criteria; weighting criteria; transforming criteria; selecting decision makers; and obtaining reliable performance measures. If sufficient time, effort, and skill are devoted to these tasks, MCA provides a robust and informative evaluation of decision options.

The choice of whether to apply MCA or an alternative economic evaluation framework hinges upon the question of valuation. There are strong arguments, both practical and methodological, that valuation has limited scope. Many intangible nonmarket environmental goods are beyond the realm of monetary quantification. In these cases, the adoption of CEA, CUA, or MCA can provide a more robust and methodologically sound analysis.

The argument is not for MCA to replace BCA or environmental valuation. BCA and some valuation techniques have an established place in the economist's toolkit and will continue to inform resource-allocation decisions. Rather, the toolkit needs diversification to handle the complexities of evaluation when intangible outcomes are important. Policy makers will be in a better position to achieve sustainability outcomes if MCA is made available alongside more conventional methods.

Acknowledgement

Gratitude is expressed to three anonymous reviewers who provided detailed, thoughtful, and helpful comments on the draft manuscript.

References

Abdalla, C., Roach, B., & Epp, D. 1992. Valuing environmental quality changes using averting expenditures: an application to groundwater contamination. *Land Economics* 68(2):163–169.

Abrishamchi, A., Ebrahimian, A., Tajrishi, M., & Marino, M. 2005. Case study: application of multicriteria decision making to urban water supply. *Journal of Water Resources Planning and Management* 131(4):326–335.

Adamowicz, W. 2004. What's it worth? An examination of historical trends and future directions in environmental valuation. *The Australian Journal of Agricultural and Resource Economics* 48(3):419–444.

Bennet, J. 2005. Australasian environmental economics: contributions, conflicts and cop-outs. *The Australian Journal of Agricultural and Resource Economics* 49(3):243–261.

Brans, J., Vincke, P., & Marshal B. 1986. How to select and how to rank projects: The PROMETHEE method. *European Journal of Operational Research* 24(2):228–238.

Brauers, W. 2004. *Optimization Methods for a Stakeholder Society: A Revolution in Economic Thinking by Multi-Objective Optimization.* Boston, MA: Kluwer.

Brouwer, R. & van Ek, R. 2004. Integrated ecological, economic and social impact assessment of alternative flood control policies in the Netherlands. *Ecological Economics* 50(1–2):1–21.

Carson, R., Mitchell, R., Hanemann, M., Kopp, R., Presser, S., & Ruud, P. 2003. Contingent valuation and lost passive use: damages from the *Exxon Valdez* oil spill. *Environmental and Resource Economics* 25(3):257–286.

Chen W., Hong, H., Liu, Y., Zhang, L., Hou, X., & Raymond, M. 2004. Recreation demand and economic value: An application of travel cost method for Xiamen Island. *China Economic Review* 15(4):398–406.

Cullen. R., Fairburn, G., & Hughey, K. 2001. Measuring the productivity of threatened species programs. *Ecological Economics* 39(1):53–66.

Diamond, P. & Hausman, J. 1994. Contingent valuation: is some number better than no number? *Journal of Economic Perspectives* 8(4):45–64.

Drummond, M., O'Brien, B., Stoddart, G., & Torrance, G. 1997. *Methods for the Economic Evaluation of Healthcare Programmes*. Oxford: Oxford Medical Publications.

Dunning, D., Ross, Q., & Merkhofer, M. 2000. Multiattribute utility analysis for addressing Section 316(b) of the Clean Water Act. *Environmental Science and Policy* 3(S1):7–14.

Eckenrode, R. 1965. Weighting multiple criteria. *Management Science* 12(3):180–192.

Eder, G., Duckstein, L., & Nachtnebel, H. 1997. Ranking water resource projects and evaluating criteria by multicriterion Q-analysis: an Austrian case study. *Journal of Multi-Criteria Decision Analysis* 6(5):259–271.

Edwards, W. & Barron, F. 1994. SMARTS and SMARTER: Improved simple methods for multiattribute utility measurement. *Organisational Behaviour and Human Decision Processes* 60(3):306–325.

Fernandes, L., Ridgley, M., & van't Hof, T. 1999. Multiple criteria analysis integrates economic, ecological and social objectives for coral reef managers. *Coral Reefs* 18(4):393–402.

Figueira, J., G. Salvatore, & M. Ehrgott (Eds.). 2005a. *Multiple Criteria Decision Analysis: State of the Art Surveys.* New York: Springer.

Figueira, J., Mousseau, V., & Roy, B. 2005b. ELECTRE methods. In J. Figueira, G. Salvatore, & M. Ehrgott (Eds.), *Multiple Criteria Decision Analysis: State of the Art Surveys.* pp. 133–162. New York: Springer.

Gershon, M. & Duckstein, L. 1983. Multiobjective approaches to river basin planning. *Journal of the Water Resources Planning and Management* 109(1):13–28.

Hajkowicz, S., McDonald, G., & Smith, P. 2000. An evaluation of multiple objective decision support weighting techniques in natural resource management. *Journal of Environmental Planning and Management* 43(4):505–518.

Hajkowicz, S. 2003. Allocating environmental funds amongst regions: Fairness, efficiency and transparency. In E. Beriatos, C. Brebbia, H. Coccossis, & A. Kungolos (Eds.), *Sustainable Planning and Development*. pp. 169–187. Wessex: WIT Press.

Hajkowicz, S. & Young, M. 2005. Costing yield loss from acidity, sodicity and dryland salinity to Australian agriculture. *Land Degradation and Development* 16(5):417–433.

Hammond, J., Keeney, R., & Raiffa, H. 1998. *Smart Choices: A Practical Guide to Making Better Decisions.* Boston, MA: Harvard Business School Press.

Hayashi, K. 2000. Multicriteria analysis for agricultural resource management: A critical survey and future perspectives. *European Journal of Operational Research* 122(2):486–500.

Heilman, P., Yakowitz, D., & Lane, L. 1997. Targeting farms to improve water quality. *Applied Mathematics and Computation* 83(2–3):173–194.

Howard, A. 1991. A critical look at multiple criteria decision making techniques with reference to forestry applications. *Canadian Journal of Forest Research* 21(4):1649–1659.

Hyde, K., Maier, H., & Colby, C. 2004. Reliability-based approach to multicriteria decision analysis for water resources. *Journal of Water Resources Planning and Management* 130(6):429–438.

Janssen, R. 2001. On the use of multi-criteria analysis in environmental impact assessment in The Netherlands. *Journal of Multi-Criteria Decision Analysis* 10(2):101–109.

Joubert, A., Leiman, A., de Klerk, H., Katau, S., & Aggenbach, J. 1997. Fynbos vegetation and the supply of water: a comparison of multi-criteria decision analysis and cost benefit analysis. *Ecological Economics* 22(2):123–140.

Kangas, J., Store, R., Leskinen, P., & Mehtätalo, L. 2000. Improving the quality of landscape ecological forest planning by utilizing advanced decision-support tools. *Forest Ecology and Management* 132(2–3):157–171.

Keeney, R. & Raiffa, H. 1993. *Decisions with Multiple Objectives: Preferences and Value Tradeoffs.* New York: Cambridge University Press.

Ludwig, D. 2000. Limitations of economic valuation of ecosystems. *Ecosystems* 3(1):31–35.

McFadden, D. 1999. Rationality for economists? *Journal of Risk and Uncertainty* 19(1-3):73–105.

Mingers, R. & Rosenhead, J. 2004. Problem structuring methods in action. *European Journal of Operational Research* 152(3): 530–554.

Ozelkan, E. & Duckstein, L. 1996. Analysing water resources alternatives and handling criteria by multi criterion decision techniques. *Journal of Environmental Management* 48(1):69–96.

Pearson, L., Tisdell, C., & Lisle, A. 2002. The impact of Noosa National Park on surrounding property values: An application of the hedonic price method. *Economic Analysis and Policy* 32(2):155–171.

Pohekar, S. & Ramachandran, M. 2004. Application of multi-criteria decision making to sustainable energy planning: A review. *Renewable and Sustainable Energy Reviews* 8:365–381.

Prato, T. 1999. Multiple attribute decision analysis for ecosystem management. *Ecological Economics* 30(2):207–222.

Raju, S., Duckstein, L., & Arondel, C. 2000. Multicriterion analysis for sustainable water resources planning: a case study in Spain. *Water Resources Management* 14(6):435–456.

Ribaudo, M., Hoag, D., Smith, M., & Heimlich, R. 2001. Environmental indices and the politics of the conservation reserve program. *Ecological Indicators* 1(1):11–20.

Roberts, R. & Goodwin, P. 2002. Weight approximations in multi-attribute decision models. *Journal of Multi-Criteria Decision Analysis* 11(6):291–303.

Robinson, J. 2000. Does MODSS offer an alternative to traditional approaches to natural resource management decision making? *Australasian Journal of Environmental Management* 7(3):170–180.

Romero, C. & Rehman, T. 1987. Natural resource management and the use of multiple criteria decision making techniques: a review. *European Review of Agricultural Economics* 14(1): 61–89.

Roy, B. 1968. Classement et choix en présence de points de vue multiples (la méthode ELECTRE). *Revue d'information et de Recherche Opérationnelle* 8:57–75.

Saaty, T. 1987. The analytic hierarchy process: What it is and how it is used. *Mathematical Modelling* 9(3–5):161–176.

Sagar, A. & Najam, A, 1998. The human development index: A critical review. *Ecological Economics* 25(3):249–264.

Sagoff, M. 1988. *The Economy of the Earth: Philosophy, Law, and the Environment.* New York: Cambridge University Press.

Scheubrein, R. & Zionts, S. 2006. A problem structuring front end for a multiple criteria decision support system. *Computers & Operations Research* 33(1):18–31.

Schultz, M. 2001. A critique of EPA's index of watershed indicators. *Journal of Environmental Management* 62(4):429–442.

Spurgeon, J. 1998. The socio-economic costs and benefits of coastal habitat rehabilitation and creation. *Marine Pollution Bulletin* 37(8–12):373–382.

Steuer, R. & Na, P. 2003. Multiple criteria decision making combined with finance: A categorized bibliographic study. *European Journal of Operational Research* 150(3):496–515.

Stoneham, G., Chaudhri, V., Ha, A., & Strappazzon, L. 2003. Auctions for contracts: An empirical examination of Victrorias BushTender trial. *The Australian Journal of Agricultural and Resource Economics* 47(4):477–500.

Strijker D., Sijtsma, F., & Wiersma, D. 2000. Evaluation of nature conservation: An application to the Dutch Ecological Network. *Environmental and Resource Economics* 16(4):363–378.

Tecle, A. 1992. Selecting a multicriterion decision making technique for watershed resources management. *Water Resources Bulletin* 28(1):129–140.

Tol, R. 1996. The damage costs of climate change towards a dynamic representation. *Ecological Economics* 19(1):67–90.

van Bueren, M. & Bennett, J. 2004. Towards the development of a transferable set of value estimates for environmental attributes. *The Australian Journal of Agricultural and Resource Economics* 48(1):1–32.

von Neumann, J. & Morgenstern, O. 1944. *Theory of Games and Economic Behaviour*. Princeton, NJ: Princeton University Press.

Weistroffer, H., Smith, C., & Narula, S. 2005. MCDM software. In J. Figueira, G. Salvatore, & M. Ehrgott (Eds.), *Multiple Criteria Decision Analysis: State of the Art Surveys.* pp. 989–1018. New York: Springer.

Whittington, D. 2002. Improving the performance of contingent valuation studies in developing countries. *Environmental and Resource Economics* 22(1–2):323–367.

Zanakis, S., Solomon, A., Wishart, N., & Dublish, S. 1998. Multiattribute decision making: A simulation comparison of selected methods. *European Journal of Operational Research* 107(3):507–529.

Zeleny, M. 1973. Compromise programming. In J. Cocharane & M. Zeleny (Eds.), *Multiple Criteria Decision Making*. pp. 262–301. Columbia, SC: University of Southern Carolina Press.

ARTICLE

Protecting paradise: a cross-national analysis of biome-protection policies

Candace Archer & Shannon Orr
Bowling Green State University, Williams Hall, Bowling Green, OH 43402 USA (email: skorr@bgsu.edu)

Land protection policies such as creating and preserving national parks have been promoted to counter global threats to the environment and to conserve biodiversity. We know little, however, about the country characteristics that might be good predictors of whether states will choose to protect land or not. What factors within a state need to be the focus of global attention or need to be encouraged to promote land-protection policies? Using the global standard of 10% ecoregion protection, we test four categories of predictors–biodiversity, environmental threats, politics (such as treaty participation and NGO activity), and economics (such as GDP and trade measures)–as well as a multidimensional model in a multivariate analysis of 129 countries. Our findings suggest that the multidimensional model best predicts when it is likely that a country will protect land. While a number of key variables such as economic are not supported, the environmental threats model presents us with the strongest individual reason for land protection.

KEYWORDS: natural areas protection, biodiversity, geopolitics, economic factors, environmental impact sources, environmental management

Introduction

Land-protection policies have been promoted to counter global threats to the environment such as timber harvesting, land overuse, and population growth. In 1987, the United Nations Commission on Environment and Development (UNCED) recommended in the Brundtland Report, *Our Common Future*, that adequate conservation of the earth's ecosystems required at minimum a tripling of the total expanse of protected areas (Brundtland Commission, 1987). Building on this report goal, many governments and conservation organizations have interpreted this recommendation to mean protecting between 10 and 12% of a region's land area (O'Neill, 1996).[1] In 1992, the targeted goals were further specified at the Fourth World Conservation Union (IUCN) World Parks Congress. Refining the UNCED targets to ensure protection of varied ecosystems and landscapes, the IUCN called for at least 10% of each of the fifteen global biomes to be protected (IUCN, 2007).[2]

[1] It should be noted that this 10–12% target has been criticized as inadequate, reflecting what was deemed politically viable at the time rather than what was optimal from an environmental protection standpoint (see, for example, Cox et al. 1994).

[2] Biomes are defined as "the world's major communities, classified according to the predominant vegetation and characterized by adaptations of organisms to that particular environment" (Campbell, 1996).

Together the UNCED and IUCN targets express the continuing development of a global norm to protect land through policy mechanisms such as national parks. The World Conservation Union estimates that there are 44,000 protected areas in the world that cover over 13.6 million square kilometers, reflecting a dramatic increase since the formation of the world's first national park at Yellowstone in the United States in 1872 (IUCN, 2007), yet still insufficient from an environmental policy perspective (Rodrigues et al. 2004; Parris, 2005; Deguise & Kerr, 2006)

While the extent of protected land has grown over the last century and the importance of protecting land has been widely cited as critical to sustainable development and environmental protection, there has been little scholarly work on why countries might choose to protect land (Gutman, 2002; Abuzinada, 2003; Parrish et al. 2003; Stoll-Kleemann, 2005). Specifically, do such countries share any characteristics? This presents an interesting scholarly puzzle, and from an applied policy perspective it is imperative to understand the factors that influence decisions to protect land. What country characteristics might be good predictors of whether states will choose to protect land or not and more explicitly attain the 10% biome protection goal? What factors within a state need to be the focus of global attention or need to be encouraged to promote land protection policies? Based on answers to these questions, can we predict what areas may or may not be protected in the future and perhaps change the policy process to protect valuable land and meet related sustainable development goals?

Two issues merit further discussion. First, actual and effective land protection ultimately depends on several factors including state capacity (financial and administrative) to carry out the policies in place. Although many states appear on the list as meeting the 10% target, this designation might in actual practice be exaggerated because governments are unable to effectively enforce the stated policies. This important issue has drawn the attention of several scholars who argue that legally protecting land does not easily translate into actually protecting land. Moreover, it has been proven that lands with a "protected" designation differ widely in actual protection (see Zimmerer et al. 2004; Naughton-Treves et al. 2005; 2006). We acknowledge the lack of homogenization among lands under protective status and agree that the process is more complicated than simply legislating protection. In this article, however, we focus on understanding state similarities and differences that would predict when the 10% threshold is met or approached. Even if the actual protection applied to a particular piece of land that helps a state reach the threshold is less than ideal, we assert that carrying out or enforcing protection is ultimately dependent on the policies being constructed and land being designated for protection. While the 10% threshold might be an imperfect measure of what is actually occurring in terms of conservation on the ground, protected status is a necessary initial condition for committing state financial and administrative resources and for actual protection to begin. The difference between how land is actually protected from state to state and how state capacity influences the success of protection is beyond the scope of this paper, but clearly we need to better understand those issues as well.[3]

Second, specific state contexts and anecdotal accounts of specific pieces of land are compelling, but our goal is to assess the practice more systematically. Thus, the present study seeks to provide a better understanding of why countries might choose to implement policies to protect land. Using the global standard of 10% biome protection we test four categories of predictors—biodiversity, environmental threats, politics, and economics—which are discussed below.

To better assess the role of these indicators, we present a multivariate analysis of 129 countries.[4] We develop five models to test the relevance of sets of indicators, and include a multidimensional model incorporating all categories. Our dependent variable is *protected ecoregions*, as calculated by the Center for International Earth Science Information Network (CIESIN) and derived from the World Database of Protected Areas and the World Wildlife Federation's mapping of ecoregions. Our environmental testing is two pronged; we test the effects of both biodiversity and environmental threats on the likelihood that land will be protected. In our political model we test treaty participation, IUCN membership organizations per million of population (a measure of nongovernmental organization (NGO) activity), and regime type. The economic model looks at gross domestic product (GDP) per capita, external debt, trade, and gross national income and posits that country wealth and global economic interactions will affect biome protection. Finally, to understand the interaction of these models, we create a multidimensional model that combines the variables from the environmental, political, and economic models. We expect that protection will increase in response to high levels of biodiversity, the presence of environmental threats, political connectedness, and economic development.

Our findings suggest that the multidimensional model best predicts when it is likely that a country will protect biomes. The environmental threats model presents us with the strongest individual reasons for land protection. Surprisingly, the political variables are poor predictors of protected land and economic factors have mixed but interesting results. However, our multidimensional model provides better results than all four independent models. This leads us to conclude that predictors of land-protection policies are quite complex and must be understood as being an interaction among political, economic, and environmental factors.

The argument proceeds in four parts. First, we discuss land protection and the biome-protection standard. Second, we examine the theoretical literature that discusses why countries might protect land. From this, we identify the political, environmental, and economic variables suggested in the literature as reasons that states may choose to protect land. This variable can be used to construct hypotheses of why states might choose to place land in protective status. Third, we present our hypotheses about how our variables should affect the level of protected land and our statistical findings that support these hypotheses. Finally, we then interpret and discuss our results, show how our findings can inform policy, and suggest further areas for research.

Measuring Protected Land: A Global Overview

As a key component of sustainable development, protected land is an umbrella term used to identify

[3] See for example O'Neill, 1996; Bates & Rudel, 2000; Hayes, 2006.

[4] Due to missing data, the N ranges from 109–129 across the five models.

areas that are managed by government for the benefit of the larger society. There are several ways to measure the amount of land under protection. The first method considers land based on management objectives for which IUCN has developed six standardized categories of protected land:

I. *Strict Nature Reserve/Wilderness Area*: Managed primarily for scientific research and/or environmental monitoring with extremely limited public access.
II. *National Park*: Managed for both ecosystem protection and public recreation.
III. *Natural Monument*: Managed for conservation of specific natural or cultural features.
IV. *Habitat/Species Management Area*: Managed mainly for conservation of a habitat and/or to provide for a particular species.
V. *Protected Landscape/Seascape*: Managed primarily for landscape/seascape protection, sustainable use, and recreation.
VI. *Managed Resource Protected Area*: Managed to support the sustainable use of natural ecosystems (IUCN, 1994).

In terms of individual parcels of protected land in the IUCN categories, Europe has the largest number of protected areas with over 43,000 sites, followed by North Eurasia with nearly 18,000 sites, North America with over 13,000 sites, and Australia and New Zealand with close to 9,000 protected areas. The Pacific, with around 320 sites, has the fewest number of protected areas. There are nearly 4,390 protected areas in Eastern and Southern Africa with a further 2,600 sites in Western and Central Africa. In terms of protected land mass, Central America and South America have the largest expanse of protected areas, covering almost 25% of each of these regions. North America is also well represented, with 4.5 million square kilometers (km^2), or just over 18% of the region's land surface, although much of that is in the sparsely populated northern regions. Protected areas cover 1.6 million km^2 (over 14.5%) of Eastern and Southern Africa and over 1.1 million km^2 (over 10.5%) in Western and Central Africa. The Pacific has over 20,000 km^2 of protected areas (approximately 1.5%) (IUCN, 2007).

While the IUCN categories are an important way to assess the amount of protected land globally, they are not consistent with the norms suggested by the Brundtland Report, the United Nations Conference on Environment and Development (UNCED), and the 1992 Fourth IUCN World Parks Congress regarding environmental sustainability that are being tested here. International conferences have moved toward privileging land protection based on specific biomes or types of ecosystem. A state could gain a high score on the IUCN categorical calculation and a low score on ecosystem protection by protecting one biome at the expense of another. For example, a large state could protect its entire desert (typically low biodiversity) and none of its rain forest (high biodiversity) and have a very high score according to the IUCN calculation, but a much lower score for ecoregion protection. Conversely, the CIESIN measure of ecoregion protection, which is derived from the World Database of Protected Areas and the World Wildlife Fund's ecoregion mapping, is consistent with the emerging global norms of interest to this research. This is the primary reason that the CIESIN measure is the dependent variable in this study. Of course, the disadvantage of this measure is that countries with fewer ecoregions may attain an unwarranted higher score; however this is reflective of larger problems with the international norm.

Theories of Protected Land Creation

It is important to explain the wide variation in land protection across states. Theoretical discussions regarding the differing tendencies of states to protect land can be grouped into three categories: environmental (threats and diversity), political, and economic. Because little has been written about the specific reasons countries would choose to protect land, an interesting area of future study, this research relies on theoretical literature that addresses more general environmental protection to provide a basis for testable hypotheses.

Protection for Environmental Reasons

Environmental arguments for land protection are rooted in the idea that protected lands are created as a means to preserve species and endangered ecosystems. These contentions are based on two subtly nuanced claims: that parks protect biodiversity and/or that they protect against environmental threats such as urban sprawl and industrial development.

Species extinction has been tied to the greater "biome crisis" in which biodiversity loss and ecological problems are intrinsically related to the larger issue of ecosystem degradation (Parrish et al. 2003; Dilsaver & Wyckoff, 2005; Hoekstra et al. 2005). While most policy initiatives to protect endangered species are tied to small "hot spots," such as Penang National Park in Malaysia, the world's smallest national park (25.62 km^2), a broader conservation policy focused on entire ecosystem protection is required to make a significant difference (Hoekstra et al. 2005). Larger expanses of protected land such as Denmark's Greenland National Park (972,000 km^2) and the Amazon Rain Forest (over 1 million km^2 under varying levels of protection) are examples of this approach.

Notwithstanding a general consensus that biodiversity must be conserved, it is still largely unclear how best to do so. Despite the emergence of the global norm of land protection, the debate continues over the value of protected land status versus sustainable use. Sustainable use arguments suggest that conservation occurs through people's use of resources (Robinson, 1993), while protected area arguments suggest that use must be strictly limited to reduce biodiversity stress. While sustainable use arguments may have political and economic appeal, they have been deemed largely insufficient from an ecological perspective (Parrish et al. 2003; Hoekstra et al. 2005; Gorenflo & Brandon, 2006). The analysis presented in this article provides a preliminary test of these competing claims to determine if measures of biodiversity in fact are related to protection in the international arena.

Forests are critical to environmental health for a number of reasons: preventing erosion, providing habitat for flora and fauna, absorbing carbon dioxide and replenishing oxygen (which is critical to preventing climate change), reducing pollution, and conserving groundwater, to name a few (Taylor, 1973; Bates & Rudel, 2000; United Nations Forum on Forests, 2007). As such, forestry issues have been a global priority since the 1992 United Nations Forum on Forests (United Nations Forum on Forests, 2007). Because forests of all types (temperate, boreal, and tropical) provide homes to a wide array of flora and fauna, and because they are less likely to face competing land uses, it seems that forested areas would be more likely to be protected (Bates & Rudel, 2000).

Perhaps the most obvious theoretical argument for protecting land is to prevent the degradation of the natural environment (Sprinz & Vaahtoranta, 1994). The motivation for protecting land in this case stems from a response to threats such as human use, deforestation, and population growth (Ridenour, 1994; Hopkins, 1995; Lowry, 1999; Macleod, 2001; The Coalition of Concerned National Park Service Retirees, 2004; Stoll-Kleemann, 2005; Hayes, 2006). The likelihood that states facing environmental threats may protect land to a greater degree is a relationship worth investigating.

One of the most significant concerns in terms of protected land is the need to safeguard ecosystems from human use. High anthropogenic impacts are problematic because they degrade the natural environment and disrupt ecosystems (Hoekstra et al. 2005). According to recent estimates, 21.8% of global land area is under human dominated use, extensively in tropical dry forests (for example, 69% of southeast Asia has been converted to human use), temperate broadleaf and mixed forests, temperate grasslands and savannas (with more than 50% lost in North America), and Mediterranean forests, woodlands, and scrub. In contrast, tundra and boreal forests remain almost entirely intact (Hoekstra et al. 2005). This posits a relationship between the amount of human impact on a state and its likelihood to protect land.

Similarly, it has been argued that deforestation is a particularly compelling threat, leading to conservation policies (broadly construed) because of visual evidence that can spur citizens and policymakers to action (Bates & Rudel, 2000). It would be expected that timber harvest rates should be positively correlated with ecoregion protection.

The rate of population growth has been linked with a number of associated environmental threats, including pollution, waste, habitat loss, water scarcity, soil erosion, development, deforestation, and increased resource demands. Many questions remain, however, about the relationship between population growth and biodiversity (Cincotta & Engleman, 2000). For instance, does population growth spur protection policies or are protection policies less likely in high growth areas because of competing demands for land?

Protection for Political Reasons

Protecting land is an inherently contentious process. Since the establishment of protection removes land from private and public development, it typically involves imposing restrictions on contact and use. This can affect a population's access to profitable natural resources (e.g., minerals) or needed subsistence resources (e.g., food or firewood). Protected land creation is most often a political decision,[5] and by and large stems from the policy process, political actors, and governmental decision making. The development of protected lands is usually the direct result of government policy and it is governments who implement that policy. In contrast, other environmental policies, such as air pollution and alternative energy, may originate with and/or be implemented by private corporations.

Due to its political nature, one theory about land protection suggests that governments are most likely to confer protective status when there is a critical mass of public support. One way to study public support on an international scale is through the activities of organized interests. Interest groups, particularly global NGOs, have been active in environmental is-

[5] There are a few exceptions of NGOs purchasing private land and putting it into a public trust or quasitrust to protect it from development. The Nature Conservancy is one example of an NGO involved in various land acquisition arrangements such as debt-for-nature swaps and conservation easements.

sues and specifically in promoting national parks and protected land. These organizations may encourage the protection of land and resources through direct political pressure or indirectly through international intergovernmental organizations (IGOs) like the World Bank or regional lenders (Bates & Rudel, 2000; Frank et al. 2000). NGOs have been instrumental in lobbying governments, purchasing land, facilitating debt-for-nature swaps, training conservation personnel, and identifying suitable land for protection (Frank et al. 2000).

The existence of public support, and the influence of NGOs domestically, are probably directly affected by the political system within a state. In a democracy, it is more likely that interest groups and public opinion affect policies than in a nondemocracy. Several authors argue that the common characteristics of democracy (e.g., freedom of speech, freedom of association, voting) allow citizens to mobilize more effectively to influence government and, in turn, act in the interests of environmental protection (Payne, 1995; Midlarsky, 1998). In a larger study, Neumayer (2002) concluded that democracies exhibit a stronger commitment to many environmental issues than nondemocracies. He specifically includes land under protected status and finds that democracies are likely to protect a larger percentage of their land. Although the connection between democracy and environmental protection continues to be questioned (Desai, 1998), evidence suggests that democracy may be a determinate of which countries choose to protect their land.[6]

The development of an international norm to protect land also seems important politically. Using event-history analysis, Frank et al. (2000) finds that parks (along with four other dependent variables) increase over time as national environmental protection becomes normalized both domestically and internationally. Country ties to world society are positively related to land protection, as is the presence of "domestic receptor sites" that can transfer information from the world to local actors, such as state organizations. These effects are present even when population and industrial development are controlled, although parks are somewhat more likely to be founded in countries with large populations. Frank et al. (2000) argue that park development may more accurately reflect organizational capacity or population pressures. While these variables are different than those we are examining, this prior study does suggest that politics plays a role in the increase in the number of parks over time.

O'Neill (1996) explores whether states create protected areas in response to pressure from international organizations and other states. She operationalizes international pressure as participation in international treaties (e.g., trade, arms control, and the environment) and uses this measure as a proxy to estimate exposure of state officials to norms of international relations, and more precisely, to conservation. We build on this work with the creation of a different measure of norms and treaty participation by creating a variable measuring participation specifically in protected land treaties.

Regime type and international norms emerge from the literature as political factors encouraging states to choose to protect land. We use Freedom House scores to test regime type. To indicate a commitment to international environmental norms, we use IGO membership and the level to which a state is party to protected land treaties.

Protection for Economic Reasons

Tradeoffs between protecting the environment and encouraging economic growth are cited in both the economics and international political economics literature. This connection between the environment and economics can be traced back to the 1960s when global environmental movements began (Meier & Rauch, 2005). Both developed and developing countries have to find ways to balance environmental concerns with promoting economic growth. For developed countries, the issues coalesce around how industrialization and economic expansion have generated pollutants such as greenhouse gases or landfill waste. In developing countries, the issues are usually couched in terms of the relationship between poverty and the environment, and how attempts to escape poor economic conditions can lead to environmental degradation. Even though the connection between countries protecting land and their economic status has not been significantly explored, we can use studies dealing with other environmental concerns to posit relationships between economics and land protection.

The advent of sustainable development paradigms facilitated the expansion of the field of environmental economics to study the interactions between economics and the environment. But the evi-

[6] There is an ongoing debate in the literature about the relationship between democracy and environmental protection. Democracies seem to be more likely to sign and ratify environmental agreements, participate in environmental intergovernmental organizations, and comply with reporting requirements. Democratic processes also tend to facilitate information sharing about environmental problems. Furthermore, the interest group tradition in most democracies enables victims of pollution or other environmental threats to organize and make demands on government. At the crux of the debate, however, is the fact that many of the democracies in the world also tend to have the highest levels of greenhouse gas emissions and pollution. The interactions between democracy, economic development, and the environment are extensive and complicated and are a rich area for further research.

dence about how the two affect each other is complicated. In some cases, economic growth improves environmental quality, and in others, it does not. Most studies on this topic are relatively recent and suggest at least three discernable relationships between economic growth and the environment; that economic growth (1) improves environmental quality, (2) at first damages, but later helps a society protect the environment or (3) hurts the environment (World Bank, 1992).

The optimistic position is that economic growth, or an increase in a population's affluence, will positively affect environmental protection. The argument is that states with greater wealth are more likely to protect land and the environment overall. Some work has been done to examine the relationship between wealth and park creation. Bates & Rudel (2000) argue, "nations that create parks are probably more prosperous than other nations" and, given the expense of park management, this correlation seems likely. Frank et al. (2000) have also found that industrial development has a positive and significant effect on the formation of parks.

The case that wealth affects environmental protection also derives from the argument that more affluent societies are more attuned to postmaterialistic needs. Inglehart (1990; 1997) argues that industrial societies have different cultures or values that derive from their affluence and the satisfaction of their more immediate needs. In his view, postmaterialist societies are more prone to value the environment and therefore are more likely to protect it. Supporting Inglehart's position are studies suggesting that attitudes about protecting the environment are more prevalent in countries with higher per capita GNP (Diekmann & Franzen, 1999; Franzen, 2003).[7]

This reasoning leads one to believe that perhaps affluence and economic growth is the savior for the environment, and conversely global poverty is the problem (Beckerman, 1992; Hollander, 2004). Such ideas are supported by the experiences of many developed countries that have fewer incidences of specific environmental problems such as contaminated drinking water or adequate sanitation (World Bank, 1992).

A corollary to this argument, and a second way that economic growth affects the environment, is the suggestion that while environmental problems may increase at early developmental stages, they will taper off as personal incomes and national wealth rise. This position has been termed the "environmental Kuznets curve." Named after a similar curve hypothesized by Simon Kuznets (1955) to explain the relationship between economic growth and income inequality, the environmental Kuznets curve literature suggests that economic growth might initially give rise to environmental damage, but environmental quality improves once incomes surpass about US$12,000 (Grossman & Krueger, 1995). The strongest support for this relationship has been found in air-quality measures and in certain pollutants across various countries (Selden & Song, 1994; Grossman & Krueger, 1995; Cole et al. 1997). However, there is significant debate about whether the environmental Kuznets curve is specific to only some pollutants and thus not generalizable across a wider range of environmental issues (Shafik, 1994; Ekins, 1997).

The final posited relationship between economic growth and the environment is that rising incomes and national wealth harm the environment. Several strands of economic and sociological literature support this contention, including those that represent an anticapitalist agenda and argue that capitalist production systems are more concerned with short-term growth than with issues like environmental protection (Redclift, 1987). Thus, industrial production systems and expanding economic growth, while possibly raising a state's economic profile, do so at the expense and exploitation of the environment (Dauvergne, 2001; 2005; Rees, 2003). Even studies that suggest some positive relationships between economic growth and the environment often also point out that rich countries are more likely than low-income countries to deal with certain environmental problems, including resource depletion and excessive waste (World Bank, 1992; Dauvergne, 2005).

One economic factor that has received significant criticism regarding its affect on the environment is trade. Scholars have argued that liberal trade causes developing countries to specialize in dirty industries, subsequently harming local environments to exploit their comparative advantage. Liberal trade policies are seen from this perspective to lead to an environmental "race to the bottom" and contribute to declining environmental quality (Rock, 1996; Grether & deMelo, 2003). But many empirical studies have had a difficult time proving that increased trade as a result of liberalization has led to environmental problems within less developed countries (Birdsall & Wheeler, 1993; Mani & Wheeler, 1998; Eskeland & Harrison, 2003) and Antweiler et al. (2001) argue that free trade appears to benefit the environment, for

[7] The environmental and affluence arguments have been countered by Frank et al. (2000), who argue that the affluence and environmental degradation arguments do not hold up to historical scrutiny. They argue that the international exponential rise in environmental activities, including park creation, is evidence that countries pursue environmental protection regardless of affluence. Indeed, affluence seems to have little effect on degree of protection, as evidenced by the oil wealth of the Middle East. Others who contradict Inglehart's thesis include Brechin & Kempton (1994) and Dunlap & Mertig (1995).

Table 1 Descriptive statistics of variables.

Variables	N	Minimum	Maximum	Mean	Standard Deviation	VIF in Multi-dimensional Model	Original data source and notes
Ecoregion Protection	132	0.0	100	62.72	31.53	n/a	Center for International Earth Science Information Network at Columbia University (CIESIN) – in conjunction with IUCN, World Database on Protected Areas and UNEP World Conservation Monitoring Centre
National Biodiversity Index	157	0.11	1.00	.55	.159	1.50	Convention on Biological Diversity (United Nations)
Forest Area	186	.00	95.00	30.12	22.56	1.55	World Development Indicators (WDI)
High Anthropogenic Impact	217	.00	100.00	8.13	16.54	1.95	CIESIN
Timber Harvest rates	132	.00	100.00	89.76	25.70	1.37	Food and Agriculture Organization (FAO) forestry database
Population Growth	199	-1.00	5.00	1.39	1.17	1.66	WDI
Trade 2004	150	31	372	93.19	47.7	1.32	WDI
GDP Per Capita (US $) (log)	169	2.02	4.70	3.30	.68	3.241	WDI (log transformation by authors)
Party to Protected Land Treaties	191	0	22	5.7	1.84	2.40	Compiled from http://sedac.ciesin.org/entri/ Environmental Treaties and Resource Indicators of the CIESIN
Freedom House Standardized scale (2000) 100 points	188	14.28	99.96	64.63	28.59	1.93	Freedom House
IUCN membership	199	.00	62.5	1.694	7.09234	1.29	IUCN

example by promoting production in areas where it is most environmentally appropriate, or by enhancing global economic development sufficiently to fund environmental programs. The relationship between trade and negative environmental effects is more strongly demonstrated in the area of land use and deforestation. For instance, López (1997) asserts that deforestation increases with expanded trade liberalization and Chichilnisky (1994) argues that many studies have confirmed that deforested areas are caused by agricultural production for the international market. Thus, a relationship might exist between a state's desire to protect land and its level of international trade, assuming higher volumes of trade would be more indicative of a more liberal trade policy.

Although the literature on economic growth and the environment comes to sometimes divergent conclusions, evidence suggests a relationship between the environment and certain economic variables. Building on this work, we are interested in understanding how, if at all, economic variables might predict whether states protect land. The economic variables we use are country affluence or wealth as expressed through gross domestic product (GDP) per capita and international trade. With regard to protected land, our general assumptions are that wealthier countries will protect land better.

Variables and Model Specifications

The theoretical literature provides adequate support for investigating the roles of environmental, political, and economic variables in predicting the amount of land that a state protects. To test these relationships, we develop five models based on these theoretical insights. A discussion of the dependent variable and our independent variables in each of our models follows. Descriptive statistics for all variables are shown in Table 1.

Dependent Variable: Protected Ecoregions

The protected ecoregions variable was calculated by CIESIN from the 2004 World Database of Protected Areas and the World Wildlife Federations map Terrestrial Ecoregions of the World.[8] The dependent variable is based on the global target of protecting 10% of the land area of each biome (i.e., desert, forest, grassland) in each country. CIESIN developed this variable by calculating the land area of 10% of each biome in a country and then comparing the values to the actual land area under protected status for each biome as a ratio. If the protected area is equal to or greater than 10%, then the country receives a score of 1 for that biome; if, for example, only 5% of the biome is protected (half the global target) then it receives a score of 0.5. These ratios are then averaged for all of the biomes in a country, and converted to percentages in the regression analysis. A score of 100% means that all biomes in a country are at least 10% protected.

Protecting land for the sake of protecting land fails to advance the sustainable development agenda. A commitment to protecting the diverse biomes of the world goes much farther in ensuring that environmental goals are being met. The correlation between the variables *ecoregion protection* and *percentage of protected land* is only 0.293, suggesting that policies to protect land do not necessarily take into account the international standard of biome protection, thereby giving reliability and confidence to the dependent variable.

Independent Variables

A number of different independent variables are employed to test five different models for protecting ecoregions. For the sake of clarity, the independent variables are introduced according to the model in which they are tested. The theoretical discussions above regarding the environmental, political, and economic explanations associated with protection policies have driven our choice of independent variables and we have grouped these variables based on their association with the model being tested.

Environmental Models

The Biodiversity Model

The theoretical literature suggests that biodiversity is a key reason for protecting land. To test this hypothesis, this model includes two independent variables that are reflective of biodiversity. First, *forest area* is derived from the World Development Indicators (published by the World Bank) and identifies the percentage of standing forest in each country. Second, we use the *National Biodiversity Index*, which is based on estimates of the richness of four terrestrial vertebrae classes and vascular plants, with each considered equally. Only countries with a land area greater than 5,000 k^2 are included in this measure. Index values range from a maximum of 1.00 (i.e., Indonesia) to a minimum of 0.00 (Greenland, excluded from study).

H_1: As the Forest Area and National Biodiversity Index increase, ecoregion protection increases.

The Environmental Threats Model

Our second environmental model is based on assumptions in the literature that a perceived environmental threat will spur countries to protect land. We use three independent variables in this model that represent threats. First, *Timber Harvest Rates* are used to understand threats to biodiversity. The data on timber harvest are sourced from the Food and Agriculture Organization (FAO) forestry database and represent all roundwood that has been felled/harvested and removed.[9] Second, *Population Growth (annual percentage)* represents the encroachment of human activity on land as this intrusion could lead to environmental problems. Population growth is calculated as an annual percentage of growth from the 2005 World Development Indicators. Third, *High Anthropogenic Impact* is used. As discussed above, human use is one of the major environmental threats contributing to the biodiversity crisis. This variable is measured as the percentage of total land area (including inland waters) with a very high anthropogenic impact. The original source of the data is the CIESIN at Columbia University (Esty et al. 2005).

H_2: As anthropogenic impacts, timber harvest rates, and population growth increase, ecoregion protection decreases.

Political Model

In the political model we are interested in testing if international norms and regime type, specifically democratic or authoritarian, affect whether a state chooses to protect land. Three independent variables are used. First, *IUCN Membership* is measured as the number of IUCN membership organizations per mil-

[8] 2004 World Database of Protected Areas is available at http://maps.geog.umd.edu/WDPA/WDPA_info/English/WDPA2005.html and the World Wildlife Federations map of Terrestrial Ecoregions of the World is available at http://worldwildlife.org/wildworld/.

[9] The original data are available at: http://faostat.fao.org/faostat/collections?version=ext&hasbulk=0&subset=forestry.

lion people.[10] IUCN members include national/international NGOs, state agencies, and state members. This variable is used to suggest how connected to international organizations a state is, assuming that organizations carry international norms with them. The original information source is the IUCN; however, the data were obtained from the Millennium Development Goals. Second, *Independent Party to Protected Land Treaties* is used to understand the environmental commitment of a state. It is assumed that a more highly committed state will be party to a larger number of treaties. This variable is measured as a count of protected land treaties to which a country is a party.[11] The variable was constructed by the authors and is used as a proxy for political commitment to the international norm of protected land. Third, *Freedom House Standardized Scores* are used as a proxy measure of regime type, or the degree of democratic freedom in a country. The variable is a standardized scale from 0–100 measuring civil liberties and political rights.

H_3: As IUCN membership, Freedom House scores, and treaty participation increase, ecoregion protection increases.

The Economic Model

The economic model tests the relationship between affluence and trade on the designation of protected public lands. We use two independent variables in the economic model. Per capita GDP is a standard measure of national income. A number of analysts have argued that the variable should be logged based on the assumption that differences of a few hundred dollars are more significant for poorer nations than wealthier nations (Brechin & Kempton, 1994; Dunlap & Mertig 1995). In working with a large data set, we concurred with this assumption and logged *GDP per capita (2004)*. The data source is the World Development Indicators produced by the World Bank. Second, *Trade* is used to measure connectedness to international markets and also to measure economic success. The relationship between trade and the environment is heavily studied. The literature concludes that trade will both impair and improve environmental quality, but makes stronger claims about the negative effects of trade on land degradation; therefore, we hypothesize that increased trade will weaken land protection. Higher levels of trade will suggest a greater commitment to liberal trading policies. This variable was derived from the World Development indicators and is measured in US$ for 2004, the most recent complete year of data.

H_4: As GDP per capita increases, ecoregion protection increases.
$H_{4\text{-}1}$: As trade increases, ecoregion protection decreases.

The Multidimensional Model

Finally, our multidimensional model recognizes that perhaps the decision to protect land is not solely expressed through environmental, political, or economic lenses, but is actually a representation of the interactions of these three sets of variables. To test this conjecture, we combine all of the variables discussed above into a single model.

One general problem with multidimensional models is issues of collinearity whereby a misspecified model includes mutually dependent, and thus redundant, predictors. To test for issues of multicollinearity we ran diagnostic tests, specifically the variance inflation factor (VIF), which is a more sophisticated test than reporting correlations of the independent variables. Although there is some dispute over the appropriate cut-off point, a generally accepted rule is that the VIF should not exceed ten (Belsley et al. 1980). The debate in this case is somewhat moot as only two variables had VIFs higher than 2.0 (*Party to Protected Land Treaties* = 2.40 and *Log GDP per capita* = 4.725) and neither of them were close to a value of concern. VIF values are included in Table 1.

Findings

Table 2 presents multiple regression results. Model 1 (Biodiversity) supports the biodiversity variable, however, forest area is not supported. Model 2 (Environmental threats) is fully supported in the regression analysis with all variables significant. However, the variable *High Anthropogenic Impact* shows a negative relationship which is counter to our hypothesis. One explanation for this outcome is that land-protection policies are lagging behind human development. This observation suggests that once an area is subject to a high anthropogenic impact, protection policies fall off the agenda, even if the area could be rehabilitated. This possibility is particularly worrisome from a protection standpoint as human development is infringing on more and more of the world. Of course, conversely it may be that these areas are

[10] While other measures of civil society would be ideal, for instance international NGO (e.g., the Nature Conservancy) activity in a country, such data are not available on the scale of this data set. Clearly there is a need for data collection on this issue.

[11] Some of these treaties are broad umbrella agreements such as *The Convention on Biological Diversity* and the *International Convention for the Protection of Birds*, while others are more specific such as the *Convention on the Conservation of Migratory Species of Wild Animals* and the *Convention on Wetlands of International Importance especially as Waterfowl Habitat.*

Table 2 Dependent variable: ecoregion protection

Variables	(1) Biodiversity		(2) Environmental Threats		(3) Political		(4) Economic		(5) Multidimensional	
	B	Beta	*B*	Beta	*B*	Beta	*B*	Beta	*B*	Beta
National Biodiversity Index	47.268* (18.313)	.239							50.645* (20.498)	.259
Forest Area	-.184 (.140)	-.121							-.024 (.173)	-.015
High Anthropogenic Impact			-1.075* (.303)	-.223					-.1.741* (.579)	-.359
Timber Harvest rates			.303** (.105)	.248					-.144 (.127)	-.114
Population Growth			6.861** (2.771)	.233					6.598* (3.265)	.222
IUCN Membership					2.645 (3.132)	.075			3.174 (3.126)	.099
Freedom House					.087 (.132)	.075			.127 (.147)	.103
Party to Protected Land Treaties					-1.595 (.901)	-.190			.470 (1.101)	.057
GDP Per Capita (US $) (log)							-9.419* (4.390)	-.203	1.903 (7.135)	.041
Trade 2004							-.081 (.083)	-.091	-.010 (.087)	-.011
R	.309		.403		.177		.236		.527	
r^2 (adj.)	.081		.142		.008		.038		.204	
N	129		128		127		111		109	

Numbers in parentheses are standard errors

*p < .05 **p < .01 (one-tailed tests)

Note: *B* refers to the independent contribution of each independent variable to the prediction of the dependent variable. Beta refers to standardized variables thereby allowing comparisons of the relative contribution of each independent variable to the prediction of the dependent variable.

no longer worth protecting because the extent of damage is so great. Additional research on this topic is needed.

Somewhat surprisingly, none of the variables were significant in Model 3 (Political). Additional correlational data on IUCN membership suggest that IGOs are not usually active in countries with strong measures of the National Biodiversity Index (-0.077) or forests (0.149), but are slightly more likely to be present in areas with high anthropogenic impacts (0.383). Political participation on the part of environmental organizations thus may be more associated with areas under environmental stress than with high levels of biodiversity. This perhaps represents a reactive rather than a proactive policy presence. Taken into context with the regression results, IGOs may have less of a need to work in high biodiversity countries, as their sensitive areas may be more likely to have some protected status. Further analysis of non-IUCN member organizations may answer some of these questions.

Model 4 (Economic) indicates that trade is not a significant variable in predicting levels of ecoregion protection; however per capita GDP (logged) is significant, with a negative relationship. Alternative specifications of economic variables, such as per capita Gross National Income (GNI), produced results that were not significant when analyzed in a bivariate regression. An external debt management variable derived from the 2004 World Development Indicators was dropped from the analysis due to multicollinearity problems, but in a bivariate regression with ecoregion protection it has a *B* of 5.527 (significant at the 0.05 level) and an adjusted R^2 of 0.028.

The economic model did not perform as we had expected. Both trade and per capita GDP relationships proved counter to our hypotheses. First, we anticipated that trade would have some effect on protected land and we advanced the conjecture that more trade would be correlated with less protection. This relationship was suggested by the literature on trade and the environment, but does not apply for protected land. Regarding trade, we found no significant relationship between trade levels and protection. The more interesting relationship is that per capita GDP was negatively correlated, a finding that suggests richer countries are less likely to protect land. Our attempts to better understand this relationship by using GNI did not yield any definitive results.

Although the literature on postmaterialism suggests industrial countries are more concerned with values that privilege the environment, our findings suggest that this is not the case. Such a contention with respect to land protection may not be counterintuitive for several reasons. Since land sells at a premium in developed countries, and land protection means removing land from use (and probably from private hands), it would be a much more expensive endeavor to protect land in a developed country than in a poorer one. Due to the increased cost, it is also probably a more contentious policy decision. Although these results were not what we expected, they are supported by literature that suggests economic development can impair portions of the environment. In addition, our results seem to contradict the postmaterialist thesis and thus contribute to a larger theoretical debate on whether affluence and environmentalism are connected. GDP is not a good predictor of land protection.

The fifth stage of the regression analysis is the multidimensional model which integrates the previous four models into a single multiple regression equation. This model notes a number of changes, in particular that both per capita GDP and timber-harvest rates drop out once the other variables are introduced. This model provides a better explanation of variance than the others, suggesting that the reasons for protecting land are quite complex and draw from many different policies and preferences.

What is a plausible explanation for the multidimensional model findings? What these analyses suggest is that biodiversity is the primary driving forces of protected land policies. Countries with high biodiversity are more likely to protect land. As such a goal is consistent with the land-protection norm, this outcome is one small indicator of success. The second driver is population growth which suggests either a reactionary policy approach as a rationale for protecting land or people's preference for beautiful areas. Case-study research would likely help to clarify this point. The only significant negative factor was high anthropogenic impact. While case studies would also shed greater light on this variable, it is likely that areas with high anthropogenic impacts have less that is still worth protecting due to the degradation caused by use. This observation suggests that while protected land policies are targeted to areas worth protecting, and in need of protecting from a biodiversity perspective, there is an urgency to protect land at risk of human-caused degradation. Politics and economics, at least based on these measures, are not as influential. This may reflect the fact that protection policies are not necessarily *meaningful* protection policies, which would require both political and economic resources.

One limitation of this statistical analysis is the small R^2 terms for each of the models. While perhaps endemic to the research question at hand, these outcomes nonetheless highlight the complexity of protection policies and suggest room for further study. To try to improve the predictive nature of our models, we did a secondary analysis that included regional variables to see if this lower level of aggregation would increase the R^2 values or change the resultant analysis. We ran regressions on just Latin America, just Africa, just tropical countries, and also included these as regional dummy variables. We also added a developing countries variable into the full data set. None of these analyses generated significant results, suggesting that regional variation does not affect land-protection policies. In fact, in the case of Latin America, the only significant variable was *High Anthropogenic Impact* (negative relationship) and for Africa none of the variables was significant. The four models were also run with standardized variables to see if the considerable variation in magnitude in the independent variables biased the results. There was no change when the variables were standardized. We also tested a variable based on the number of ecoregions per state, but this was not significant.

Conclusion

According to IUCN, by the year 2000 there were 30,000 protected areas covering more than13 million km^2 of the world's land surface (roughly the size of India and China combined). Protected areas not only conserve biodiversity and natural features, but also protect watersheds and soils. They serve important research and education needs and contribute to local economies through sustainable activities. Other areas protect and promote cultural values and, of course, can provide emotional or spiritual escapes from modern life.

Failure to protect the land from human activity results in biodiversity loss, decreases landscape variety, and diminishes ecological interactions and the evolutionary processes that sustain and promote biodiversity (Hoekstra et al. 2005). Although there are costs, and protecting land can be a difficult policy decision, the social and environmental benefits can be enormous. Moreover, as environmental issues rise on national and international policy agendas, the role of protected land will remain important.

Despite the many benefits of land protection, most countries have fallen far short of the international target of protecting 10% of each national biome. Based on the multiple regression findings in this article, land protection can be predicted based upon biodiversity factors, environmental threats (high anthropogenic impact and timber harvest rates), and economic development. Some these findings are reassuring, in particular that environmental factors are leading to more protection. As the concern for the environment continues to grow, and as states add these issues to their policy agendas, it is likely that we will continue to see more biomes come under protection. From an international policy perspective, these findings suggest that making connections with environmental issues is probably the best choice for getting land protection onto national agendas.

The political and economic results provide less hope for the future of land protection and are in need of more research. Politically, international environmental norms and treaties to address environmental issues are more prevalent than ever before, as are states with political systems and policy processes that allow public interests to be expressed. But these factors are not leading to greater land protection. Further research needs to focus on how linkages can be made between these positive political developments and land protection.

Further study also needs to be conducted regarding economic issues. The relationship between environmental and economic objectives is complicated and our results contribute to these debates, particularly whether wealthier countries are more committed to environmental issues. Our findings contradict the belief that richer countries are better at protecting the environment and therefore challenge the postmaterialist thesis. More research is needed to isolate economic variables and to test their significance for land protection. In addition, we need to better understand the policy issues surrounding land protection in developed countries. Do property values affect protection and why might it be more difficult to protect land in more economically advanced countries?

Our results confirm that the decision to protect land is a complex one that appears to be influenced by many factors. While we have begun to explore this issue and offer some much needed research, questions still remain. In particular, uncertainty still surrounds whether land protection policies are actually meaningful and which policy mechanisms can encourage more than just token protection. Since land protection is vital for so many social benefits and for continued environmental preservation, we must continue to work to understand these relationships to devise better strategies.

Authors' Note

An earlier version of this paper was presented at the 2007 International Studies Association Annual Meeting. We would like to thank the members of the panel, as well as the anonymous reviewers, for their helpful suggestions.

References

Abuzinada, A. 2003. The role of protected areas in conserving biological diversity in the kingdom of Saudi Arabia. *Journal of Arid Environments* 54(1):39.

Antweiler, W., Copeland, B., & Taylor, M. 2001. Is free trade good for the environment? *The American Economic Review* 91(4):877–908.

Bates, D. & Rudel, T. 2000. The political ecology of conserving tropical rain forests: a cross–national analysis. *Society & Natural Resources* 13(7):619–634.

Beckerman, W. 1992. Economic growth and the environment: whose growth? Whose environment? *World Development* 20(4):481–496.

Belsley, D., Kuh, E., & Welsch, R. 1980. *Regression Diagnostics: Identifying Influential Data and Sources of Collinearity*. New York: John Wiley.

Birdsall, N. & Wheeler, D. 1993. Trade policy and industrial pollution in Latin America: where are the pollution havens? *The Journal of Environment and Development* 2(1):137–149.

Brechin, S. & Kempton, W. 1994. Global environmentalism: a challenge to the postmaterialism thesis? *Social Science Quarterly* 75(2):245–269.

Brundtland Commission. 1987. *Our Common Future: Report of the World Commission on Environment and Development*. Oxford: Oxford University Press.

Campbell, N. 1996. *Biology, 4th Edition*. Menlo Park, CA: The Benjamin Cummings Publishing Company.

Chichilnisky, G. 1994. North–south trade and the global environment. *American Economic Review* 84:851–853.

Cincotta, R. & Engleman, R. 2000. Biodiversity and population growth. *Issues in Science and Technology* 16(3):80–81.

Coalition of Concerned National Park Service Retirees. 2004. *A call to action: saving our National Park system*. Tucson, AZ: The Coalition Of NPS Retirees. http://www.npsretirees.org/cnpsr/call-action-nps-governance.

Cole, M., Rayner, A., & Bates, J. 1997. The environmental Kuznets curve: an empirical analysis. *Environment and Development Economics* 2(4):401–416.

Cox, J., Kautz, R., MacLaughlin, M., & Gilbert, T. 1994. *Closing the Gaps in Florida's Wildlife Habitat Conservation System: Recommendations to Meet Minimum Conservation Goals for Declining Wildlife Species and Rare Plant and Animal Communities*. Report No.020. Tallahassee, FL: Office of Environmental Services, Florida Fish and Wildlife Conservation Commission.

Dauvergne, P. 2001. *Loggers and Degradation in the Asia–Pacific: Corporations and Environmental Management*. Cambridge, MA: MIT Press.

Dauvergne, P. 2005. Globalization and the environment. In J. Ravenhill (Ed.), *Global Political Economy*. pp. 448–478. Oxford: Oxford University Press.

Deguise, I. & Kerr, J. 2006. Protected areas and prospects for endangered species conservation in Canada. *Conservation Biology* 20(1):48–55.

Desai, U. 1998. Environment, economic growth and government in developing countries. In U. Desai (Ed.), *Ecologial Policy and Politics in Developing Countries*. pp. 1–46. Albany, NY: State University of New York Press.

Diekmann, A. & Franzen, A. 1999. The wealth of nations and environmental concern. *Environment and Behavior* 31(4):540–549.

Dilsaver, L. & Wyckoff, W. 2005. The political geography of national parks. *Pacific Historical Review* 74(2):237–266.

Dunlap, R. & Mertig, A. 1995. Global concern for the environment: is affluence a prerequisite? *Journal of Social Issues* 51(4):121–137.

Ekins, P. 1997. The Kuznets curve for the environment and economic growth: examining the evidence. *Environment & Planning A* 29(5):805–830.

Eskeland, G. & Harrison, A. 2003. Moving to greener pastures? Multinationals and the pollution haven hypothesis. *Journal of Development Economics* 70(1):1–23.

Esty, D., Levy, M., Srebotnjak, T., & de Sherbinin, A. 2005. *2005 Environmental Sustainability Index: Benchmarking National Environmental Stewardship*. New Haven, CT: Yale Center for Environmental Law and Policy. http://www.yale.edu/esi/ESI2005.pdf .

Frank, D., Hironaka, A., & Schofer, E. 2000. The nation–state and the natural environment over the twentieth century. *American Sociological Review* 65(1):96–116.

Franzen, A. 2003. Environmental attitudes in international comparison: an analysis of the issp surveys 1993 and 2000. *Social Science Quarterly* 84(2):297–308.

Gorenflo, L. & Brandon, K. 2006. Key human dimensions of gaps in global biodiversity conservation. *Bioscience* 56(9):723–731.

Grether, J.–M. & de Melo, J. 2003. Globalization and dirty industries: Do pollution havens matter? CEPR Discussion Paper No. 3932. London: Center for Economic Policy Research.

Grossman, G. & Krueger, A. 1995. Economic growth and the environment. *The Quarterly Journal of Economics* 110(2):353–377.

Gutman, P. 2002. Putting a price tag on conservation: Cost benefit analysis of Venezuela's National Parks. *Journal of Latin American Studies* 34(1):43–70.

Hayes, T. 2006. Parks, people, and forest protection: An institutional assessment of the effectiveness of protected areas. *World Development* 34(12):2064–2075.

Hoekstra, J., Boucher, T., Ricketts, T., & Roberts, C. 2005. Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters* 8(1):23–29.

Hollander, J. 2004. *The Real Environmental Crisis: Why Poverty, Not Affluence, Is the Environment's Number One Enemy*. Berkeley, CA: University of California Press.

Hopkins, J. 1995. *Policymaking for Conservation in Latin America: National Parks, Reserves and the Environment*. Westport, CT: Praeger.

Inglehart, R. 1990. *Culture Shift in Advanced Industrial Society*. Princeton, NJ: Princeton University Press.

Inglehart, R. 1997. *Modernization and Postmodernization: Cultural, Economic, and Political Change in 43 Societies*. Princeton, NJ: Princeton University Press.

IUCN. 1994. *Guidelines for Protected Area Management Categories*. Gland, Switzerland: World Conservation Monitoring Centre.

IUCN. 2007. IUCN Protected Areas Program. http://www.iucn.org/themes/wcpa/ppa/protectedareas.htm. December 15, 2007.

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

Lopez, R. 1997. Environmental externalities in traditional agriculture and the impact of trade liberalization: The case of Ghana. *Journal of Development Economics* 3(1):17–39.

Lowry, W. 1999. Providing intergenerational goods: Implementation of national park system plans in Canada and the United States. *Policy Studies Journal* 27(2):328-347.

Macleod, D. 2001. Parks or people? National parks and the case of Del Este, Dominican Republic. *Progress in Development Studies* 1(3):221–235.

Mani, M. & Wheeler, D. 1998. In search of pollution havens? Dirty industry in the world economy, 1960 to 1995. *The Journal of Environment and Development* 7(3):215–247.

Meier, G. & Rauch, J. 2005. *Leading Issues in Economic Development*. New York: Oxford University Press.

Midlarsky, M. 1998. Democracy and the environment: an empirical assessment. *Journal of Peace Research* 35(3):341–61.

Naughton–Treves, L., Holland, M., Buck, M., & Brandon, K. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources* 30:219–52.

Naughton-Treves, L., Alvares-Berrios, N., Brandon, K., Bruner, A., Holland, M., Ponce, C., Saenz, M., Suarez, L., & Treves, A. 2006. Expanding protected areas and incorporating human resource use: a study of 15 forest parks in Ecuador and Peru. *Sustainability: Science, Practice, & Policy* 2(2):32–44. http://ejournal.nbii.org/archives/vol2iss2/0602-009.naughton.html.

Neumayer, E. 2002. Do democracies exhibit stronger international environmental commitment? A cross–country analysis. *Journal of Peace Research* 39(2):139–164.

O'Neill, K. 1996. The international politics of national parks. *Human Ecology: An Interdisciplinary Journal* 24(4):521–539.

Parris, T. 2005. Watching over our protected areas. *Environment* 47(5):3.

Parrish, J., Braun, D., & Unnash, R. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. *Bioscience* 53(9):851–860.

Payne, R. 1995. Freedom and the environment. *Journal of Democracy* 6(3):41–55.

Redclift, M. 1987. *Sustainable Development: Exploring the Contradictions*. New York: Routledge.

Rees, W. 2003. Economic development and environmental protection: an ecological economics perspective. *Environmental Monitoring and Assessment* 86(1–2):29–45.

Ridenour, J. 1994. *The National Parks Compromised: Pork Barrel Politics and America's National Treasures*. Merrillville, IN: ICS Books.

Robinson, J. 1993. The limits to caring: Sustainable living and the loss of biodiversity. *Conservation Biology* 7(1):20–28.

Rock, M. 1996. Pollution intensity of GDP and trade policy: Can the World Bank be wrong? *World Development* 24(3):471–479.

Rodrigues, A., Akcakaya, H., Andelman, S., Bakarr, M., Boitani, L., Brooks, T., Chanson, J., Fishpool, L., Fonseca, G., Gaston, K., Hoffmann, M., Marquet, P., Pilgrim, J., Pressey, R., Schipper, J., Sechrest, W., Stuart, S., Underhill, L., Waller, R., Watts, M., & Yan, X. 2004. Global gap analysis: priority regions for expanding the global protected–area network. *Bioscience* 54(12):1092–1100.

Selden, T. & Song, D. 1994. Environmental quality and development: Is there a Kuznets Curve for air pollution emissions? *Journal of Environmental Economics and Management* 27(2):147–162.

Shafik, N. 1994. Economic development and environmental quality: an econometric analysis. *Oxford Economic Papers* 46(S):757–773.

Sprinz, D. & Vaahtoranta, T. 1994. The interest–based explanation of international environmental policy. *International Organization* 48(1):77–105.

Stoll-Kleemann, S. 2005. Voices for biodiversity management in the 21st century. *Environment* 47(10):24–36.

Taylor, D. 1973. Some ecological implications of forest fire control in Yellowstone National Park. *Ecology* 54(6):1394–1396.

United Nations Forum on Forests. 2007. History and Milestones of International Forest Policy. http://www.un.org/esa/forests/about-history.html. January 22, 2008.

World Bank. 1992. *World Development Report: Development and the Environment*. Washington, DC: World Bank.

Zimmerer, K., Galt, R., & Buck, M. 2004. Globalization and multi–spatial trends in the coverage of protected–area conservation (1980–2000). *Ambio* 33(8):520–529.

Sustainability: Science, Practice, & Policy
http://ejournal.nbii.org

COMMUNITY ESSAY

Comprehensive conservation planning and ecological sustainability within the United States National Wildlife Refuge System

Richard L. Schroeder

Fort Collins Science Center, United States Geological Survey, 2150 Centre Avenue, Building C, Fort Collins, CO 80526 USA (email: Rick_Schroeder@usgs.gov)

Author's Personal Statement:

For the past ten years, I have had the privilege of working with the National Wildlife Refuge System (NWRS) of the United States Fish and Wildlife Service as it develops Comprehensive Conservation Plans (CCP) for each refuge unit. I have read and studied published CCPs, and paid particular attention to the scientific and biological aspects of these plans. Of particular interest to me has been the mandate to sustain healthy populations of fish, wildlife, and plants and the biological integrity, diversity, and environmental health of the refuge system, or, essentially, the "ecological sustainability" of the system. One of the great difficulties in trying to implement a concept as profound and complex as ecological sustainability is to determine how one might measure progress toward its achievement. In this essay, I have tried to select a few simple but relevant factors to serve as indicators of such progress. A wise older friend of mine, in explaining her personal view of changing the world, said that some of the problems we face are like a huge ball blocking our path. She knew that she alone could not move the ball, but her goal was to at least nudge it in the right direction. It is my hope that this essay serves as a nudge to NWRS as it moves toward the goal of ecological sustainability.

Introduction

The National Wildlife Refuge System (NWRS) of the United States Fish and Wildlife Service (FWS) has been in existence for over 100 years, but it was only recently that these designations received a systemic mandate with passage of the Refuge Improvement Act (RIA) of 1997. Previously, the nation's national wildlife refuges lacked organic legislation that provided a clear, central mission, and individual units were established through a patchwork of executive orders and other laws. The refuges allowed a wide variety of uses that did not always complement the objectives of wildlife management. With passage of RIA, the refuge system received a new statutory mission statement. According to FWS, "This Act states first and foremost that the mission of the National Wildlife Refuge System be focused singularly on wildlife conservation." The RIA directs FWS to provide for the conservation of fish, wildlife, and plants, and their habitats throughout the refuge system. The legislation defines the terms "conserving," "conservation," "manage," "managing," and "management" as meaning to sustain and, where appropriate, restore and enhance, healthy populations of fish, wildlife, and plants. The RIA further requires FWS to ensure that the biological integrity, diversity, and environmental health of the refuges are maintained for the benefit of present and future generations of Americans. As Meretsky et al. (2006) note, the provision related to biological integrity, diversity, and environmental health is "[o]ne of the most emphatic ecosystem conservation directives ever written by Congress." Fischman (2003) provides an excellent history of the earlier laws guiding NWRS management.

Given the RIA mandates, the key ecological sustainability aspects of concern within the refuge system are sustaining healthy fish, wildlife, and plant populations and, on a broader basis, sustaining biological integrity and diversity and environmental health. It should be noted that many definitions of the term "sustainability" refer to three dimensions: social, economic, and ecological. However, the mission of the refuge system focuses strictly on wildlife conservation which falls within the realm of ecological sustainability. Thus, this essay will be restricted to the ecological dimension.

One of RIA's mechanisms for moving toward ecological sustainability of NWRS is the requirement to complete comprehensive conservation plans (CCPs) for the more than 500 units in the system by the year 2012. These plans provide management direction for a 15-year period for each refuge unit. For the past ten years, I have worked closely with FWS in providing technical assistance in this planning effort, primarily in the development of science-based and detailed biological objectives for the CCPs. The purposes of this essay are: 1) to provide an overview of FWS policies and guidance that relate to ecological sustainability in the comprehensive planning process; 2) to assess the strengths and weaknesses of the planning process in meeting these directives; and 3) to offer ideas for future planning within the refuge system.

Overview of FWS Policies and Guidance Related to Planning and Ecological Sustainability

Subsequent to the passage of RIA in 1997, FWS issued several policies that provide more specific guidance and direction for planning and management of the refuge system. The first of these was *Refuge Planning* (602 FW) (U.S. Fish and Wildlife Service, 2000). The specific chapter on the CCP process (602 FW 3) provides the following guidance related to ecological sustainability: "[CCPs] describe the desired future conditions of a refuge and provide long-range guidance and management direction to achieve refuge purposes; help fulfill the National Wildlife Refuge System mission; maintain and, where appropriate, restore the ecological integrity of each refuge and the Refuge System."

In April 2001, FWS issued a policy titled *Biological Integrity, Diversity, and Environmental Health* (601 FW 3) that provided detailed guidance on the meaning of the terms "biological integrity," "diversity," and "environmental health" and how to manage units to maintain or restore these attributes (U.S. Fish and Wildlife Service, 2001). In discussing management goals, this policy notes, "The highest measure of biological integrity, diversity, and environmental health is viewed as those intact and self-sustaining habitats and wildlife populations that existed during historic conditions." This policy also links the goal of ecological sustainability to the planning process by noting that through the CCP process FWS will determine the appropriate management direction to maintain and, where appropriate, restore biological integrity, diversity, and environmental health, while achieving refuge purpose(s).[1]

The FWS policy on *National Wildlife Refuge System Mission and Goals and Refuge Purposes* (601 FW 1) states that the refuge system's overarching goal is to conserve a diversity of fish, wildlife, and plants and their habitats for the benefit of current and future generations (U.S. Fish and Wildlife Service, 2006a). Three of the five specific goals outlined in this policy contain provisions related to ecological sustainability:

- Conserve a diversity of fish, wildlife, and plants and their habitats, including species that are endangered or threatened with becoming endangered.
- Develop and maintain a network of habitats for migratory birds, anadromous and interjurisdictional fish, and marine mammal populations that is strategically distributed and carefully managed to meet important life-history needs of these species across their ranges.
- Conserve those ecosystems, plant communities, wetlands of national or international significance, and landscapes and seascapes that are unique, rare, declining, or underrepresented in existing protection efforts.

This policy states that these goals will help guide development of specific management priorities during development of CCPs, and thus links the goal of ecological sustainability to the planning process.

The above summary of key FWS policies indicates that the concepts of ecological sustainability and planning are tightly interrelated in the refuge system. The FWS has had ten years since the passage of RIA to develop refuge plans and to move toward ecological sustainability. In the following section, I use several criteria to assess the strengths and weaknesses of the current planning effort in building a NWRS capable of sustaining biological integrity, diversity, and environmental health, including healthy populations of fish, wildlife, and plants. This essay focuses on FWS planning outside of Alaska.[2]

Assessment of Comprehensive Conservation Planning and Ecological Sustainability within the Refuge System

Based on FWS policies, I selected four key considerations to help evaluate the success of the CCPs in sustaining healthy populations of fish, wildlife, and

[1]The FWS defines historic conditions as the "[c]omposition, structure, and functioning of ecosystems resulting from natural processes that we believe, based on sound professional judgment, were present prior to substantial human related changes to the landscape." In practice, the historic time frame often refers to pre-European settlement. Historic conditions are often determined from an assessment of early explorer records, archeological data, or historic vegetation maps (e.g., Marschner, 1974).

[2] The Alaska National Interest Lands Conservation Act of 1980 directed FWS to prepare and periodically update conservation plans for all NWRs in Alaska. The first such plans were completed between 1985 and 1988 and are now being revised according to current FWS policies.

plants and the biological integrity, diversity, and environmental health of the refuge system. These considerations are:

- Use of science
- Maintenance or restoration of historic conditions;
- Inclusion of an ecosystem perspective; and
- Incorporation of adaptive management and monitoring.

Use of Science

The importance of science in achieving sustainability was emphasized by Mooney & Sala (1993) and more recently by Palmer et al. (2005). The FWS Planning Policy (602 FW 3) states that a CCP goal is "[t]o support management decisions and their rationale by using a thorough assessment of available science derived from scientific literature, on-site refuge data, expert opinion, and sound professional judgment." A cornerstone of CCPs is their biological objectives that describe the desired future conditions for wildlife and refuge habitats. Developing objectives is similar to formulating hypotheses and should be guided by the scientific method to provide a transparent and rigorous approach that can be empirically tested and subjected to peer review (Tear et al. 2005).

In an analysis of the first 60 completed CCPs covering refuges distributed widely across the system (completion dates ranged from 1997 to 2004) I found that the amount and quality of the science used to support the biological objectives were both often quite limited (Schroeder, 2006). My evaluation used the following question and ranking criteria:

How well was available science used in the development of the biological objectives?
(Note: general sources include materials such as field guides and overview texts; high quality sources include materials such as articles from scientific journals).

1. Poor (very few or no science sources cited)
2. Fair (limited number of science sources provided and sources mostly general)
3. Good (limited to many science sources provided and sources mostly of high quality)
4. Excellent (extensive number of science sources provided, from high quality sources, as described above)

The average score for the amount and quality of the science in the biological objectives in the 60 CCPs was 1.38 (with a range from 1.00 to 3.62). CCPs that scored the lowest provided no scientific documentation to explain the biological objectives, whereas the CCP with the highest score provided over 200 high-quality science citations and extensive explanations of how this science was used to develop the biological objectives. Average scores for the science criteria for the 60 CCPs were calculated for each year (1997 to 2004) and regression of average scores against year of plan completion showed a significant positive relationship ($R^2 = 0.66$, $P = 0.015$), indicating improved use of science over time.

Maintenance or Restoration of Historic Conditions

As discussed earlier, FWS policy on biological integrity, diversity, and environmental health notes that intact and self-sustaining habitats and wildlife populations that existed during historic conditions (defined in the policy as prior to substantial human-related changes to the landscape) represent the highest measure of biological integrity, diversity, and environmental health. Thus, it follows that restoration of historic conditions should be a major emphasis of current NWR planning. Indeed, this appears to be the case, as the first 55 CCPs expressed intent to conduct some form of ecosystem restoration in accordance with this aim (Schroeder, 2004). Specific examples include:

- Rydell NWR CCP (Minnesota) – "The majority of refuge wetlands, uplands, and woodlands will be restored to reflect the original natural character of the landscape."
- Windom Wetland Management District CCP (Minnesota) – "Restore native prairie plant communities of the Northern Tallgrass Prairie Ecosystem."
- Ten Thousand Islands NWR CCP (Florida) – "Restore natural sheetwater flows to the Refuge."

Many refuges were established on lands with a history of providing crops (e.g., corn, soybeans) and following designation many areas continued to be cropped to provide a food source for wildlife or to ameliorate crop-depredation problems by wildlife on adjacent private lands. In recent years, however, cropland acres have been reduced. A theme repeated in many CCPs is reduction or elimination of croplands and restoration of these areas to native plant communities. Refuges with pine or other tree plantations also plan to restore these areas to native plant communities. In an article concerning the management of refuges to restore historic conditions, Schroeder et al. (2004) note that in almost all instances it will be impossible to completely restore conditions that existed prior to substantial human-related changes. Difficulties could include the presence of upstream dams that have altered the hydrology, the relatively small size of areas available to reintroduce extirpated large carnivores or herbivores, or the inability to mimic natural processes such as wildfire. Hilderbrand et al. (2005) offer similar warnings and provide an excellent discussion of the difficulties inherent in restoration ecology, while Meretsky et al. (2006) issue a cautionary note that factors such as climate change may further restrict restoration to historic conditions. Restoration of federally listed threatened or endangered species will also be a significant challenge, as the current refuge system only supports 186 of the 514 listed animal species (Czech, 2005).

Inclusion of an Ecosystem Perspective

The importance of an ecosystem-level approach for biodiversity conservation and ecological sustainability has long been recognized. Franklin (1993) argues strongly that "[l]arger-scale approaches—at the levels of ecosystems and landscapes—are the only way to conserve the overwhelming mass—the millions of species—of existing biodiversity." In accordance with this analysis, the FWS Planning Policy (602 FW 3) encourages an ecosystem approach for refuge planning (U.S. Fish and Wildlife Service, 2000). The policy further states that CCP objectives should consider regional and FWS ecosystem objectives. Five years before this publication, FWS developed a specific policy on the Ecosystem Approach to Fish and Wildlife Conservation that called for the creation of "ecosystem teams" and the development of "ecosystem plans" with measurable objectives (U.S. Fish and Wildlife Service, 1995). However, plans have been developed for few of the 52 watershed-based ecosystems of the lower 48 states and the majority of the published plans lack specific and quantitative wildlife and habitat objectives at the ecosystem level. An assessment carried out by Christensen et al. (1998) found that FWS personnel have a wide variety of definitions for the Ecosystem Approach and that the concept and associated activities had not been integrated into daily FWS business.

An additional concern at the ecosystem level is that many refuges are becoming islands within a landscape increasingly dominated by urban and agricultural development (Scott et al. 2004). Future management will need to be concerned not only with refuge lands, but more and more with management practices on adjacent and surrounding acreage.

A few recent CCPs have developed their biological objectives in consideration of other ecosystem-level planning efforts such as the North American Waterfowl Management Plan (2004) or Joint Ventures Plans (U.S. Fish and Wildlife Service, 2007).[3] For example, the CCP for the Lake Ophelia NWR in Louisiana calls for reforesting over 1,000 acres of cropland to contribute to creating forest blocks of 100,000 acres for the benefit of neotropical migratory birds as identified in the Mississippi Alluvial Valley Migratory Bird Conservation Plan (U.S. Fish and Wildlife Service, 2005a).[4] However, such integration of local and regional planning is rare and appears to rely on the initiative of specific FWS personnel.

Incorporation of Adaptive Management and Monitoring

The FWS Planning Policy (602 FW 3) contains a section that addresses both monitoring and adaptive management. The policy notes that biological objectives and management activities should be monitored and modified as needed through adaptive management, a strategy closely related to a requirement in the policy to develop detailed CCP objectives that can be measured during monitoring to assess progress. Martin (2006) states that objectives related to sustainability must have an empirical basis that provides the ability to measure the steps necessary for achievement.

In my reviews, I have not yet encountered a CCP that has a detailed explanation of how adaptive management will be approached and that provides information on the level of monitoring that will be conducted. In fact, most CCPs contain only a short section on monitoring, which tends to have fairly generic and boilerplate wording. Excerpts from two published CCPs illustrate this point:

> Seedskadee NWR CCP (Wyoming) – "Monitoring and evaluation will utilize the adaptive management process which includes goal and objective setting, applying management tools and strategies, and monitoring and feedback to validate objectives. Adaptive management provides a framework within which biological measures can be evaluated by comparing the results of management, to results expected from objectives" (U.S. Fish and Wildlife Service, 2002).

---

[3] The North American Waterfowl Management Plan (NAWMP) is a joint effort of the United States, Canadian, and Mexican governments to develop a strategy to restore waterfowl populations through habitat protection, restoration, and enhancement. Joint Ventures Plans are partnerships involving federal, state, provincial, tribal, and local governments, businesses, conservation organizations, and individual citizens that work to implement NAWMP at the regional level, focusing on areas of concern identified in the plan.

[4] For the complete plan, see Twedt et al. (1999).

> Sherburne NWR CCP (Minnesota) – "Monitoring will be developed to measure progress toward meeting the objectives set forth in this plan. Based on the results of monitoring, the objectives will be reviewed and revised as necessary" (U.S. Fish and Wildlife Service, 2005b).

The lack of specific and measurable details in many of the biological objectives, combined with the very general guidance on monitoring and adaptive management in most plans, indicates that it will be difficult for FWS to monitor progress toward ecological sustainability through the current CCPs (Schroeder, 2006). Johnson (1999) notes that most agencies face rigorous time and money constraints and I believe that these may be limiting factors in the application of adaptive management throughout the refuge system. The challenge of monitoring was emphasized by Bernhardt et al. (2005) who analyzed more than 37,000 river-restoration projects across the entire United States and noted that only 10% of project records document any form of monitoring.

Ideas for the Future

The FWS has made progress in the first round of developing and publishing CCPs. As initiated by the mandates in RIA, a major shift in emphasis has occurred toward planning and managing for biological integrity and diversity and environmental health and, thus, the ecological sustainability of the refuge system. However, far more improvement is feasible based on the criteria that policy and other published guidance have established. Specifically, CCPs could be improved by strengthening their scientific foundation, providing more detailed and measurable objectives related to ecological sustainability, and integrating approaches across ecosystems.

What are some areas to look toward in the future as the first CCPs begin to be revised and the next phase of long-term planning begins?

The FWS, in partnership with other federal, state, and private groups, is currently developing a long-term plan for Strategic Habitat Conservation (SHC) and ecological sustainability is a key provision (National Ecological Assessment Team, 2006). The document specifically notes the importance of SHC in future refuge planning:

> The Refuge System will incorporate information derived from the SHC framework into the refuge planning process. This information will provide valuable assistance to refuge staffs and planners when evaluating and identifying the appropriate contribution that each refuge can make to larger landscape conservation priorities. Considered with Refuge System mandates, policies, and guidance, the SHC framework will help facilitate development of wildlife and habitat management goals and objectives for comprehensive conservation plans (CCPs) and habitat management plans (HMPs) that will guide future management on over 540 refuges.

If the SHC effort is successful it will offer tools and models of tremendous value in allowing future CCPs to provide biological objectives stepped down from higher level ecosystem objectives. However, earlier cautions should be revisited and considered anew, for instance those generated by the assessment of the 1995 Ecosystem Approach (Christensen et al. 1998). These concerns include the lack of a clear definition of the "ecosystem approach," the poor integration that exists across all programs within FWS, the need to use the "best science," and the importance of improved use of partnerships.

The FWS Ecosystem Approach Concept document (052 FW 1) states that management decisions will "consider the full array of biological and socioeconomic parameters" (U.S. Fish and Wildlife Service, 1996). However, this type of all-encompassing statement is exceedingly difficult to incorporate into land-management plans and has potentially far-reaching implications. The wildlife-management field increasingly perceives a conflict between continued economic growth and the sustainability of wildlife resources. Czech (2000) states that "[a] plethora of evidence indicates that economic growth is the limiting factor for wildlife conservation."

Scientific information has become much more readily available in recent years. For instance, FWS has an excellent online system that provides access to scientific abstracts and electronic journals. However, an ongoing challenge for field biologists is finding the time to review the literature (Pullin et al. 2004; Schroeder, 2006). This situation suggests the need for a coordinated system to synthesize scientific information for key species and habitats, perhaps similar to the Habitat Suitability Index models that FWS developed in the 1980s (U.S. Fish and Wildlife Service, 1981). The FWS could also benefit by evaluating new tools and methods being developed to assess ecological conditions. Meretsky et al. (2006) recommend that the refuge system gives serious consideration to multimetric indices, for example of biological or ecological integrity, to assess extant conditions. Other recent and relevant tools include indices of grassland integrity (Coppedge et al. 2006) and an ecological integrity index for littoral wetlands

(Ortega et al. 2004). As individual refuges in the national system implement their current CCPs, FWS will accumulate both quantitative and anecdotal information on various habitat-management and restoration activities. It would be helpful to future planning efforts to document these results and establish improved mechanisms for networking between refuge units, perhaps in the form of Internet-based approaches, including online “blogs.”

In addition, as CCPs are implemented and results monitored there will be enhanced need to practice adaptive resource management. The biological objectives in CCPs represent hypotheses and, as these are evaluated, publication of the results will create a permanent record. An increased emphasis on publication would allow for long-term documentation of refuge-management successes and failures to benefit future managers and researchers. FWS field staff could collaborate with outside scientists to facilitate such publications.

The management of natural systems is extremely complex; ecologist Frank Egler (1977) notes that “ecosystems are not only more complex than we think, but more complex than we can think.” On a very pragmatic note, it will be important for FWS to review and update appropriate training courses (such as the national CCP course) and various guidelines and directives to reflect new knowledge and “lessons learned” in the first round of CCP publication.

The goal of long-term sustainability of fish, wildlife, and plants, as well as the biological integrity, diversity, and environmental health of the refuge system as a whole is both admirable and daunting. One mechanism to move toward this goal is to continue a science-based, ecosystem-oriented, and adaptive system of planning. The CCP effort is likely to be critical in determining the level of ecological sustainability that NWRS is eventually able to achieve.

Acknowledgement

I would like to acknowledge Brian Czech and Natalie Sexton for their valuable feedback on an earlier draft of this essay.

References

Bernhardt, E., Palmer, M., Allan, J., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G., Lake, P., Lave, R., Meyer, J., O'Donnell, T., Pagano, L., Powell, B., & Sudduth, E. 2005. Synthesizing U.S. river restoration efforts. *Science* 308(5722):636–637.

Christensen, J., Mullins, G., Danter, K., Norland, E., & Griest, D. 1998. *U.S. Fish and Wildlife Service, Ecosystem Approach to Fish and Wildlife Conservation*. Ohio Cooperative Fish & Wildlife Research Unit, Work Order 30 Final Report. Columbus, OH: Ecological Communications Lab, The Ohio State University.

Coppedge B., Engle, D., Masters, R., & Gregory, M. 2006. Development of a grassland integrity index based on breeding bird assemblages. *Environmental Monitoring and Assessment* 118(1–3):125–145.

Czech, B. 2000. Economic growth as the limiting factor for wildlife conservation. *Wildlife Society Bulletin* 28(1):4–14.

Czech, B. 2005. The capacity of the National Wildlife Refuge System to conserve threatened and endangered animal species in the United States. *Conservation Biology* 19(4):1246–1253.

Egler, F. 1977. *The Nature of Vegetation: Its Management and Mismanagement*. Norfolk, CT: Aton Forest.

Fischman, R. 2003. *The National Wildlife Refuges: Coordinating a Conservation System through Law*. Washington, DC: Island Press.

Franklin, J. 1993. Preserving biodiversity: species, ecosystems, or landscapes? *Ecological Applications* 3(2):202–205.

Hilderbrand, R., Watts, A., & Randle, A. 2005. The myths of restoration ecology. *Ecology and Society* 10(1):19.

Johnson, B. 1999. The role of adaptive management as an operational approach for resource management agencies. *Conservation Ecology* 3(2):8.

Marschner, F. 1974. The Original Vegetation of Minnesota (Map). St. Paul, MN: U.S. Forest Service, North Central Experiment Station.

Martin, L. 2006. *Establishing Empirical Bases for Sustainability Objectives*. Monitoring Science and Technology Symposium: Unifying Knowledge for Sustainability in the Western Hemisphere Proceedings. September 20–24, Rocky Mountain Research Station, Fort Collins, CO: U.S. Department of Agriculture Forest Service.

Meretsky, V., Fischman, R., Karr, J., Ashe, D., Scott, J., Noss, R., & Schroeder, R. 2006. New directions in conservation for the National Wildlife Refuge System. *BioScience* 56(2):135–143.

Mooney, H. & Sala, O. 1993. Science and sustainable use. *Ecological Applications* 3(4):564–566.

National Ecological Assessment Team. 2006. *Strategic Habitat Conservation.* A Report from the National Ecological Assessment Team. U.S. Fish and Wildlife Service and U.S. Geological Survey. http://www.fws.gov/southwest/About%20Us/SHC%20NEAT_Final_Rpt.pdf.

North American Waterfowl Management Plan. 2004. *North American Waterfowl Management Plan 2004. Implementation Framework: Strengthening the Biological Foundation*. Canadian Wildlife Service, U.S. Fish and Wildlife Service, and Secretaría de Medio Ambiente y Recursos Naturales. http://www.fws.gov/birdhabitat/NAWMP/files/ImplementationFramework.pdf.

Ortega M., Velasco, J., Millán, A., & Guerrero, C. 2004. An ecological integrity index for littoral wetlands in agricultural catchments of semiarid Mediterranean regions. *Environmental Management* 33(3):412–430.

Palmer, M., Bernhardt, E., Chornesky, E., Collins, S., Dobson, A., Duke, C., Gold, B., Jacobson, R., Kingsland, S., Kranz, R., Mappin, M., Martinez, M., Micheli, F., Morse, J., Pace, M., Pascual, M., Palumbi, S., Reichman, O., Townsend, A., & Turner, M. 2005. Ecological science and sustainability for the 21st century. *Frontiers in Ecology* 3(1):4–11.

Pullin, A., Knight, T., Stone, D., & Charman, K. 2004. Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119(2):245–252.

Schroeder, R. 2004. *Setting Objectives for Ecosystem Restoration: An Examination of National Wildlife Refuge Comprehensive Conservation Plans*. The First National Conference on Ecosystem Restoration (NCER): Sustainable Ecosystem Restoration through Integration of Science, Planning and Policy. December 6–10, Lake Buena Vista, FL.

Schroeder, R., Holler, J., & Taylor, J. 2004. Managing national wildlife refuges for historic or non-historic conditions: determining the role of the refuge in the ecosystem. *Natural Resources Journal* 44(4):1185–1210.

Schroeder, R. 2006. A system to evaluate the quality of restoration objectives using National Wildlife Refuge Comprehensive Conservation Plans as a case study. *Journal for Nature Conservation* 14(3–4):200–206.

Scott, J., Loveland, T., Gergely, K., Strittholt, J., & Staus, N. 2004. National Wildlife Refuge System: ecological context and integrity. *Natural Resources Journal* 44(4):1041–1066.

Tear, T., Kareiva, P., Angermeier, P., Comer, P., Czech, B., Kautz, R., Landon, L., Mehlman, D., Murphy, K., Ruckelshaus, M., Scott, J., & Wilhere, G. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. *BioScience* 55(10):835–849.

Twedt, D., Pashley, D., Hunter, C., Mueller, A., Brown, C., & Ford, B. 1999. Mississippi Alluvial Valley Bird Conservation Plan. http://www.blm.gov/wildlife/plan/MAV_plan.html. February 21, 2008.

U.S. Fish and Wildlife Service. 1981. *Standards for the Development of Habitat Suitability Index Models.* ESM 103. Washington, DC: Division of Ecological Services, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 1995. *Ecosystem Approach to Fish and Wildlife Conservation*. National Policy Issuance 95-03. Washington, DC: Division of Policy and Directives Management, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 1996. *Ecosystem Approach to Fish and Wildlife Conservation*. 052 FW 1, Concept. Washington, DC: Office of the Director, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 2000. *Comprehensive Conservation Planning Process*. 602 FW 3. Washington, DC: Division of Refuges, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 2001. *Biological Integrity, Diversity, and Environmental Health*. 601 FW 3. Washington, DC: Division of Natural Resources, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 2002. *Seedskadee National Wildlife Refuge.* Comprehensive Conservation Plan. Denver, Colorado: Division of Refuge Planning, U.S. Fish and Wildlife Service, Region 6. http://www.fws.gov/mountain-prairie /planning/States/Wyoming/Seedskadee/Final/sdk_final_ccp.pdf.

U.S. Fish and Wildlife Service. 2005a. Lake Ophelia National Wildlife Refuge Comprehensive Conservation Plan. http://www.fws.gov/southeast/planning/LakeOFinalPg.html. February 21, 2008.

U.S. Fish and Wildlife Service. 2005b. Sherburne National Wildlife Refuge Comprehensive Conservation Plan. http://www.fws.gov/Midwest/Planning/sherburne/ccp/FinalCCP.pdf. February 21, 2008

U.S. Fish and Wildlife Service. 2006a. *National Wildlife Refuge System Mission and Goals and Refuge Purposes*. 601 FW 1. Washington, DC: Division of Conservation, Planning and Policy, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. 2007. Defining Joint Ventures. http://www.fws.gov/birdhabitat/JointVentures/DefineJV.shtm. February 21, 2008.

COMMUNITY ESSAY

Can biotech companies enable ethanol biofuels to achieve sustainability?

K. John Morrow, Jr.

Newport Biotechnology Consultants, 625 Washington Avenue, Newport, KY 41071 USA (email: kjohnmorrowjr@insightbb.com)

Author's Personal Statement:

This article is based on my readings and interviews during an Energy Forum in Lucerne, Switzerland in 2007. The forum was unlike any scientific meeting that I had previously attended in that the participants were passionate in their commitment and many of the talks had a strong sense of advocacy. I was struck by the concern, the sense of immediacy, and the "out of the box" proposals presented.

While elegant molecular biology-based approaches to energy self sufficiency are an essential component of the solution to the world's energy problems, even their most fervent supporters admit widespread application is years into the future. In the interim, I believe that it is absolutely essential that we consider energy alternatives that could bring about immediate energy and carbon-dioxide savings.

I find it ironic that today Americans are hammered by rising energy prices, yet no politician has suggested conservation as a respite from the run up in energy costs. I hope that this review will help to stimulate informed debate and realistic solutions to the current energy dilemma in which we are mired.

Introduction

A perfect storm of forces has coalesced to send world corn-ethanol production through the roof. Driven by insatiable consumer appetite, fears of global warming, political pressures by agricultural conglomerates, and high prices for petroleum from unstable regions of the world, biofuel production–principally corn-based ethanol–has increased prodigiously in the last few years. The United States Department of Agriculture (USDA) predicts that corn-ethanol production will reach 10 billion gallons by 2009, up from 5 billion gallons in 2006 (Westcott, 2007a).

Yet, there probably is no more controversial technology for generating sustainable fuels. Political, economic, and scientific problems challenge the viability of corn-based ethanol as a major alternative to gasoline. As these concerns multiply, alternative schemes for satisfying the exploding demand for automobile fuel have acquired new appeal. Hoping to ride on the crest of this wave are academic scientists and biotechnology companies pushing what they believe to be a greener, more efficient alternative to traditional farming practices and industrial-scale fermentation processes for generating biofuels.

Biofuel production, of course, is not new. Brazil's program, based on cane sugar, goes back three decades. Using conventional agricultural and fermentation technology, Brazilian ethanol production of 3.1 billion gallons accounts for 15% of the country's total liquid fuel supply (Valdes, 2007). With government subsidies and higher yields per hectare of cane versus corn, this strategy has been viable in Brazil, but was never seriously entertained in the United States where corn has been the main source of sugar for fermentation into ethanol.

Contemporary biofuel technology has received added impetus from advances in molecular biology that make available a variety of tactics for improving productivity. These options include genetic engineering and the redesign of plant architecture to improve productivity or alternatively targeting the organisms responsible for the fermentation process to increase robustness and improve fermentation efficiency (Stephanopoulos, 2007).

Biotech Presence in Biofuel Technology

Codon Devices, based in Cambridge, Massachussetts, is one of a number of firms currently seeking to optimize gene designs for specific applications related to biofuel production (Codon Devices, 2007). The firm provides "operon variant libraries" for screening and selection in metabolic engineering. Such libraries are collections of DNA sequences retained in bacterial plasmids that run the gamut of multiple promoters, terminators, and genes pulled in from different families. The libraries can be extracted, manipulated, and inserted into plant genomes so as to optimize synthetic performance of the plant strain. According to press releases, the firm has signed an agreement with another company, Agrivida, for the discovery, development, and commercialization of engineered proteins for use in biofuel applications (Agrivida, 2006).

The latter firm, an agricultural biotechnology company, will use the Codon Devices platform to develop enzymes optimized in corn varieties for ethanol production. Since conventional methods for manufacturing ethanol use the corn grain only, leaving the remaining cellulosic material in the field, the use of the entire plant should greatly improve efficiency and yield per hectare.

Agrivida claims the collaborative work between the two companies would utilize the remaining 50% of the total biomass yield per acre of farmland, which currently is lost in production (Agrivida, 2006b). Central to the firm's ethanol-optimized corn technology are engineered enzymes that are incorporated into the corn plants themselves. These enzymes will efficiently degrade the entire mass of plant material into small sugars that can then be readily converted to ethanol. The optimized enzymes that Codon Devices is developing will allow Agrivida to further improve the degradation process throughout the entire plant, promising significant improvements to ethanol production. This process could dramatically improve ethanol yields. The partnership between a firm specializing in recombinant DNA manipulation and a company with experience in plant manipulation seems ideal here, allowing Agrivida to dramatically enhance the volume of ethanol through cellulose degradation.

The Codon Devices/Agrivida program takes on the thorny problem of biomass recalcitrance, that is, the natural resistance of plant-cell walls to microbial and enzymatic breakdown, which currently renders the industrial-scale production of ethanol from biomass material unachievable (Himmel et al. 2007). Plants have evolved an extremely complex array of structural and chemical devices to protect them from external assault, including epidermal tissue, vascular bundles, thick wall tissues, and molecular arrays of microfibrils and polymers, all of which constitute a formidable series of barriers. Current conversion technology is costly, complex, and energy intensive (Koonin, 2006).

Overcoming these roadblocks to cost-effective biomass conversion will require a variety of approaches, including new ways for removing lignins and hemicellulose, as well as dissembling the cell at the nanoscale to allow the penetration of pretreatment chemicals and hydrolytic enzymes. Moreover, other biotechnology companies are pursuing strategies to genetically engineer the plant cell so as to make it more amenable to chemical and enzymatic digestion.

However, such proposals to radically redesign plant-cell walls to make them more receptive to cellulosic conversion are challenging and fraught with peril. It is likely that modifying the plant-cell wall would make the plant more fragile and subject to structural failure and sensitivity to fungal pathogens (Palmer, 2007). In addition, a number of estimates have been made of energy yields from biofuels, a hornet's nest of controversy. Hill et al. (2006) calculate that corn ethanol yields 25% more energy than that invested in its production, whereas biodiesel yields 93% more energy. Pimentel et al. (2007) report estimates of corn-ethanol energy yields that challenge these claims and conclude that with current technology, 1.43 kilocalories (kcal) of fossil energy is expended for every 1 kcal of ethanol generated. They further contend that previous estimates ignored various energy inputs (transport equipment and other farm machinery) and generate an inaccurately optimistic estimate of the net overall energy yield in corn-ethanol production. Farrell et al. (2006) compare a number of different studies and, while estimates of energy ratios varied, argue that ethanol produced from switchgrass would be much more favorable in terms of its production of greenhouse-gas emissions. But all commentators agree that large-scale biofuel generation from corn or soybeans cannot replace much petroleum without drastically affecting food supplies. The draconian move of dedicating all United States corn and soybean production to biofuels would meet only 12% of gasoline and 6% of diesel demand (Farrell et al. 2006).

Indeed, the dramatic increase in corn-ethanol production appears already to be driving up farm commodity prices. According to USDA, corn prices have risen sharply, from $1.75/bushel in 2000 to $3.50/bushel in July of 2007 and to $6.00/bushel in the first half of 2008 and these market dynamics have driven up prices of other crops and meat (Westcott, 2007b; Associated Press, 2008). For the foreseeable future, food prices can be expected to rise faster than the general rate of inflation. There is general agreement throughout academic and commercial sectors that 12 to 15 billion gallons of ethanol is the maximum that could be produced from corn without severe disruption of the entire price structure for farm commodities (Pimental et al. 2007).

Even beyond today's inefficient methods, a serious objection remains to technologies that convert all the components of the corn plant (i.e., leaves, stems, roots) to a source of cellulosic material for conversion to sugar. Such a process would return zero nutrients to the soil. In addition, the presence of these materials on the field protects from wind erosion. Standard farming practices use the unproductive components of the plant as fertilizer for the next year's crop by mechanically crushing and recycling at least 50% of the unharvested portions (known as corn stover). If this cycle is interrupted, farmers will essentially be mining their fields and the only way to maintain productivity will be by application of synthetic fertilizers. Such activity raises the energy requirements of production to a negative energy balance, comparable to current corn-production technology (Pimental et al. 2007). Indeed, any scheme based on recovering all of the plant material from annual crops will eventually collapse as a failed perpetual motion machine. In addition, the infrastructure for marketing, transporting, and storing corn stover does not exist and will require years to construct.

The case for ethanol production from grasses or woody material is more appealing from the energy-balance standpoint. An authoritative and carefully researched study carried out under the auspices of the Oak Ridge National Laboratory documents that a billion tons of biomass is available for conversion to biofuel without serious economic, environmental, or agricultural disruption (Perlack et al. 2006). The Cambridge-based Mascoma Corporation is building demonstration facilities to convert waste biomass into ethanol. The company has developed technology for improving the early steps in the process, including the removal of the lignin that shields the cellulose, and the next phase, the conversion of cellulose to sugar through an improved enzyme cocktail. The firm estimates that this technology could produce ethanol from wood chips for about the same price as from corn, and eventually for much less (Mascoma, 2008). However, there are still daunting basic scientific and technical problems. Himmel et al. (2007) estimate that developing biomass conversion for cost-effective motor-fuel production could be realized by 2030.

Verenium, a company recently formed from the merger of Diversa and Celunol, is also seeking to develop a cellulosic biofuels program (Verenium, 2007). The firm has developed enzyme products for the conversion of plant material, including corn and agricultural waste, into ethanol. Its first biofuels product, Fuelzyme™-LF enzyme, is intended to increase the efficiency of ethanol production from corn. This product is an LF alpha-amylase designed to increase the throughput of corn, providing superior liquefaction. A second product, Fuelzyme™-CX, is aimed at the conversion of cellulosic biomass to ethanol. The company is developing industrial-scale facilities for cellulosic ethanol production and by 2010 intends to have a plant that will produce 25 to 30 million gallons annually. Its business plan outlines a long-term commitment to alternative fuels.

Controversial Issues in Biofuel Development

Much of the controversy currently surrounding biofuels may result from overreaching promotion that can never be fulfilled. According to Righelato & Spracklen (2007), critical issues need to be considered when weighing the efficacy of biofuels as mitigators of fossil-fuel emissions. Since vast amounts of agricultural acreage would have to be taken over to grow crops for biofuels, one needs to calculate the loss of carbon sequestration due to changes in land use. These authors calculate that a 10% substitution of gasoline and diesel fuel by biofuels would require 43% and 38% of the current crop land in the United States and the European Union (as the European Union was comprised in 2001) respectively. Since this reallocation would cause a loss of almost half of United States food production capacity, huge tracts of forest and grassland would have to be converted to crop production with attendant loss of carbon storage. Righelato & Spracklen (2007) further estimate that in all cases forestation will sequester two to nine times the amount of carbon emissions avoided by biofuels raised on this land over a 30-year period. They furthermore argue that if the object of biofuels policy is to decrease carbon dioxide (CO_2)-induced global warming, a much better approach would be to increase conservation of fossil fuels and, at the same time, restore and conserve natural forests and savannahs.

In the same vein, Berkeley petroleum engineer Tad Patzek has recently argued that the economic, environmental, and social costs exacted by a massive corn-ethanol program would far outweigh the benefits obtained.[1] A number of other presentations at the same forum counsel conservation.

Indeed, if a massive biofuels program achieved 30% of United States automobile fuel needs by 2030, the amount by which petroleum use would be decreased would be much less than that which could be obtained by replacing American cars with their European model counterparts. At present, select European models average 52 miles per gallon (mpg) versus 32 mpg for the American version of the same car. Accordingly, the European automobile fleet obtains 61% better mileage than the same models manufactured for the United States market due to the substitution of highly efficient diesel engines and manual transmissions.[2] Since there are more carbon atoms

[1]Presentation at the European Sustainable Energy Forum, July 3–6, 2007, Lucerne, Switzerland. See http://www.efcf.com.

[2] See http://www.gas-cost.net. The website claims that "across the board, European models get an average of 52 miles per gallon (MPG) versus 32 MPG for the US version of the same car. So the [European version of the] same car…gets 60% better gas mileage than" the American version.

per gallon of diesel (12-16 carbon atoms/molecule) as compared to gasoline (approximately 8 atoms/molecule), CO_2 mitigation from the diesel fuel itself would be limited (Bullis, 2006). However, there would be tremendous savings of petroleum resources from the conversion to diesel fuel as well as from manual transmissions and other energy-saving features that could decrease automotive CO_2 emissions. These calculations (ca. 60% fuel reduction with automobile fleet changeover versus 30% reduction with massive biofuels program) clearly favor conservation over biofuels and business as usual.

There are, of course, many other strategies for improving mileage performance in automobiles and it would likely be much more effective to target tax credits toward fuel-efficient vehicles than to invest the same amount of taxpayer dollars in ethanol price supports. The counterargument is that both strategies should be adopted, but on a short-term basis a proven, immediately available technology is preferable to a long-term, untested proposition with substantial environmental costs.

So if corn ethanol is technically an energy sink and economically a black hole–and biomass ethanol is years away from commercialization–where does that leave the industry? All indications are that the strategy has shifted back to the political arena where the short-term push toward corn-based biofuels may prove unstoppable, despite the lack of scientific or economic practicality (Rosenthal, 2007; Martin, 2008). Not only are major tax incentives being promoted at the federal level in the United States, but many states, such as North Carolina, Iowa, and Ohio, have their own biofuels initiatives that include large tax breaks and subsidies.

Backers of corn ethanol (agribusiness and politicians from farm states) have been forced to admit that corn ethanol does not make much sense, but argue that producing corn ethanol will add to the infrastructure for transportation and storage, as well as consumer acceptance, of ethanol. Matthew Wald (2006) stated a couple of years ago that "[b]ackers defend corn ethanol as a bridge technology to cellulose ethanol, but for the moment it is a bridge to nowhere." While this observation may reflect a degree of hyperbole, the program as it now stands raises serious concerns that need to be answered before the country embarks upon a program costing billions and billions of dollars. It does not appear that these concerns can be allayed by application of elegant genetic engineering technologies.

Author's Note

The information concerning biotechnology companies and their research programs was obtained from company websites and press releases. As a caveat, it should be stressed that this information is available in the public domain, but is based on the claims of corporate representatives and other public pronouncements rather than peer-reviewed journal articles. While descriptions of currently available technology are verifiable, predictions for future goals must be considered hypothetical.

References

Agrivida. 2006a. Agrivida News: 2 May 2008–Milestone Achieved in Agrivida, Codon Devices Collaboration. http://www.agrivida.com/news.html. July 11, 2008.

Agrivida. 2006b. Agrivida News: 1 August 2007–Agrivida and Codon Devices Enter into Development Agreement for Biofuel Applications. http://www.agrivida.com/news.html. July 11, 2008.

Associated Press. 2008. Corn pops to record $6 per bushel. *Boston Globe* April 4. http://www.boston.com/business/articles /2008/04/04/corn_pops_to_record_6_per_bushel.

Bullis, K. 2006. Clean diesel from coal. *Technology Review* April 19. http://www.technology review.com/Energy/16713.

Codon Devices. 2007. Industrial Biotechnology & Bio-Energy. http://www.codondevices.com/applications.aspx?id=110. July 11, 2008.

Farrell, A., Plevin, R., Turner, B., Jones, A., O'Hare, M., & Kammen, D. 2006. Ethanol can contribute to energy and environmental goals. *Science* 311(5760):506–508.

Hill, J., Nelson, E., Tilman, D., Polasky, S., & Tiffany, D. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proceedings of the National Academy of Sciences* 103(30):11206–11210.

Himmel, M., Ding, S., Johnson D., Adney, W., Nimlos, M., Brady, J., & Foust, T. 2007. Biomass recalcitrance: engineering plants and enzymes for biofuels production. *Science* 315 (5813):804–807.

Koonin, S. 2006. Getting serious about biofuels. *Science* 311 (5760):435.

Martin, A. 2008. Fuel choices, food crises, and finger-pointing. *The New York Times* April 15. http://www.nytimes.com/ 2008/04/15/business/worldbusiness/15food.html?scp=1&sq= biofuel+political+pressure&st=nyt.

Mascoma. 2008. Cellulosic Ethanol 101. http://www.mascoma. com/technology/technology.html. July 11, 2008.

Palmer, M. 2007. Biofuels and the environment. *Science* 317 (5840):897.

Perlack, R., Wright, L., Turhollow, A., Graham, R., Stokes, B., & Erbach, D. 2006. *Biomass as Feedstock for a Bioenergy and Bioproducts Industry: The Technical Feasabiltiiy of a Billion-Ton Annual Supply.* DOE/GO-102995-2135. Oak Ridge, TN: Oak Ridge National Laboratory. http://feed stockreview.ornl.gov/pdf/billion_ton_vision.pdf.

Pimentel, D., Patzek, T., & Cecil, G. 2007. Ethanol production: energy, economic and environmental losses. *Reviews of Environmental Contamination & Toxicology* 189:25–41.

Righelato, R. & Spracklen, D. 2007 Carbon mitigation by biofuels or by saving and restoring forests? *Science* 317(5840):902.

Rosenthal, E. 2007. Europe pushes to get fuel from fields. *The New York Times* May 30. http://www.nytimes.com/2007/05/30/ business/worldbusiness/30biofuels.html?_r=1&scp=3&sq=bi ofuel+political+pressure&st=nyt&oref=slogin.

Stephanopoulos, G. 2007. Challenges in engineering microbes for biofuel production. *Science* 315(5813):801–804.

Valdes, C. 2007. Ethanol Demand Driving the Expansion of Brazil's Sugar Industry. Sweet and Sweeteners Outlook/SSS-249. Washington, DC: Economic Research Service, USDA. http://www.ers.usda.gov/Briefing/Sugar/sugarpdf/EthanolDemandSSS249.pdf

Verenium. 2007. Cellulosic Ethanol: Fueling the Future. http://www.verenium.com/Pages/Biofuels/BiofuelsCellulosicEtoh.html. July 11, 2008.

Wald, M. 2007. Is ethanol for the long haul? *Scientific America* 296(1):42–9.

Westcott, P. 2007a. U.S. ethanol expansion driving changes throughout the agricultural sector. *Amber Waves* 5(4):10–15.

Westcott, P. 2007b. Ethanol Expansion in the United States: How Will the Agricultural Sector Adjust? FDS-07D-01. Washington, DC: Economic Research Service, USDA. http://www.ers.usda.gov/Publications/FDS/2007/05May/FDS07D01/fds07D01.pdf.

BOOK REVIEW PERSPECTIVES

Peter Rogers, Kazi Jalal, & John Boyd, *An Introduction to Sustainable Development*

Earthscan, 2007, 416pp, ISBN: 1844075206

Claire Quinn[1] & Carolyn Snell[2]

[1]Sustainability Research Institute, School of Earth and Environment, University of Leeds, Leeds, LS2 9JT, UK
(email: C.H.Quinn@leeds.ac.uk)

[2]Department of Social Policy and Social Work, University of York, Heslington, York, YO10 5DD, UK
(email: cjs130@york.ac.uk)

Sustainable development has evolved as a concept partly in response to the inherent tensions between economic development and environmental protection. The long-held view has been that economic growth would inevitably lead to environmental degradation through the consumption of nonrenewable resources, the overuse of renewable resources, and the production of waste and pollution (Dryzek, 1997). Sustainable development offers the possibility that this is not inevitable; economic development can occur while still protecting the environment. Understandably, this prospect has had great appeal and not only governments, but many nongovernmental organizations and businesses, have taken up the principles of sustainable development. Sustainable development makes possible strategic and holistic policy making that recognizes the important relationship between society and the environment. However, there is very little consensus on what makes for desirable outcomes, or how they might be achieved, under sustainable development. The resulting responses potentially encompass both the radical transformation of social structures and markets and the reform of existing political and institutional structures to better account for the environmental impact of human activities.

In *An Introduction to Sustainable Development*, Peter Rogers, Kazi Jalal, & John Boyd extensively discuss a number of the concepts and issues relevant to sustainable development. The book's premise is to provide a "comprehensive textbook" for those who need a "thorough grounding in the subject," including both students and practitioners. It is difficult to produce an introductory text that can appeal to such a broad target audience and the authors have not always succeeded. The title does not do the book justice; given the background of the authors, the main strength of the book lies in its consideration of the economics of sustainable development. The text, in the main, is accessible, but in places the informal writing style may be off putting to some readers.

In Chapter 1, "From Malthus to Sustainable Development," the authors move from a basic account of sustainable development to a rather technical discussion. This might alienate readers without an economics background (or indeed those who are seeking a broader approach than an economics-based one), especially coming so early. Chapter 2 considers some of the challenges of sustainable development, but with limited reference to the literature or supporting examples. Chapter 3 goes on to consider, somewhat superficially, some of the key global environmental issues such as population trends, food, and energy demands. Some important global dimensions, such as deforestation and water scarcity, are only given a couple of sentences and as a result readers uninformed on the subject may deem them unimportant. Chapters 4 to 8 cover sustainable development indicators, environmental assessment and management, and environmental law and policy. However, the structure both within and between these chapters is not always logical, with sometimes vague headings and subheadings, making it difficult to follow the argument. What is consistent throughout this material is a technocentric approach. The authors do not challenge the predominant liberal economic paradigm where a free market and economic growth are essential to human welfare, although the necessity of economic growth is highly contested in sustainability literature. Once again, an exploration of these issues in the earlier chapters and some initial guidance or introduction about the specific approach taken in the textbook would have been of benefit.

Where this book excels is in its consideration of the economics of sustainable development in Chapters 9, 10, 11, and, to some extent, Chapter 12. In particular, Chapters 9 and 10 are excellent, concise, and provide a good grounding in the economics of sustainability. Chapter 13 introduces the actors involved in international cooperation on sustainable development, but perhaps misses the opportunity to consider their evolution, worldviews, and influence.

This book takes a very broad approach, but in places it feels spread too thinly–concepts are introduced briefly, but not fully explained, and do not always flow in a logical way. This is perhaps most problematic in the discussion of the concept of sustainable development. The authors use the Brundtland Report's definition of "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987), but numerous other definitions of sustainability and sustainable development have been formulated. As discussed above, the concept of sustainable development is highly contentious and is subject to numerous interpretations, ranging from the technocentric—where environmental problems are viewed as a threat to human quality of life and technology and science are viewed as the solution—to the ecocentric—where emphasis is placed on the need for radical change in political structures and human organization. More discussion of this debate and an introduction to the literature in this area could, and should, have been presented. The authors do concede that covering these issues in any detail would have lengthened the book substantially; however, the complex and contested nature of some of these issues is lost in their treatment. In the conclusion the authors suggest that they have focused on "methodologies, institutions, and policy frameworks" rather than substantive issues. While laudable, this approach and structure is not always obvious and could be outlined more clearly in the introduction.

Overall, this book may be of more use to students and practitioners with an interest in environmental economics than those focusing on sustainable development's social or philosophical dimensions.

References

Dryzek, J. 1997. *The Politics of the Earth: Environmental Discourses*. New York: Oxford University Press.

World Commission on Environment and Development (WCED). 1987. *Our Common Future*. New York: Oxford University Press.

About the Authors

Claire Quinn is an environmental social scientist with experience working on interdisciplinary projects both in Africa and the UK. Her research interests lie in the relationship between ecology, livelihoods, and institutions in natural resource management. She is currently a research fellow in the Sustainability Research Institute (SRI) at the University of Leeds.

Carolyn Snell is a Lecturer in Social Policy at the University of York and holds a joint position with the Stockholm Environment Institute and the Department of Social Policy and Social Work. Her current research interests are sustainable development, poverty, and public policy.