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ABSTRACT. This paper reviews the evolution of the field of environment and development over the last two decades. I argue that a central concern of the field has been the relation between natural resource use, income and growth, under the institutional and market conditions that prevail in developing countries. Particular attention is paid to the demographic and other drivers of change in the asset base, the linkages between poverty, property rights and the allocation of natural resources, the valuation of environmental assets and investment of resource rents, and the development of policies for managing environmental externalities and environmental public goods. I consider how the balance between topics and the treatment of individual topics has changed over time, and indicate how the field might be expected to move in the future.

1. Introduction

This paper has its origins in an exchange of views over an NBER working paper produced in the last year by Michael Greenstone and Kelsey Jack. Writing under the title 'Envirodevonomics: a research agenda for a young field', Greenstone and Jack argue that there exist a number of urgent economic and policy questions about environmental quality in developing countries that require the integration of the tools of environmental and development economics. They claim that the new field might be organized around a central question: why is environmental quality so poor in developing countries? They then offer, as candidate answers to their question: the high marginal utility of consumption at low income levels and the high marginal cost of environmental quality improvement, together with market and policy failures (Greenstone and Jack, 2013). Since they cite almost none of the existing literature on these topics, it prompted me to consider how well their perceptions might map to the evolution of the field as it has developed over the last two decades. Since the journal *Environment and Development Economics* is now in its 20th year of publication, it provides a useful lens through which to investigate the development of the field. While it is clear that Greenstone and Jack's central question has not been the focus of work on the relation between environment and development, it is also clear that market and policy failures, and the significance of poverty for the way in which people use their environment, have been enduring themes.

If there is a central question addressed by the field in the last two decades, it concerns the relation between natural resource use, income and growth, under the institutional and market conditions that prevail in developing countries. There are many aspects to this, including the demographic and other drivers of change in the asset base, the linkages between poverty, property rights and the allocation of natural resources, the valuation of environmental assets and the investment of resource rents, and the development of policies for managing environmental externalities and environmental public goods. Issues surrounding the depletion of exhaustible and renewable natural resources have in general been more prominent than issues surrounding the emission of waste to soil, water and air. There have certainly been studies of pollution, both local and global. Indeed, these account for an increasing proportion of papers in the field. But a more consistent concern has been the use and abuse of common pool resources, both locally in regions where a majority of the population makes a living from agriculture, forestry and fisheries, and globally.

The range of policy issues addressed by the field is wider than that addressed in any of the component fields or disciplines. It includes, for example, the usual design of instruments to correct for the failure of domestic markets, policies and regulatory regimes, and to assure the supply of local environmental public goods. But it also includes the design of mechanisms to secure the international interests at stake in environmental change in developing countries. The UN REDD+ scheme for combatting the effect of deforestation on climate and biodiversity is a case in point. In addition, it includes the policy issues raised by the connection between income/asset distribution and environmental change. The field has seen the development of various mechanisms that are intended both to alleviate poverty and to enhance the efficiency of environmental resource allocation. In this it reflects the findings of the Brundtland Report (World Commission on Environment and Development, 1987), which hypothesized a relationship between poverty, environmental change and economic development that has helped to structure research on the topic ever since.

The range of management issues is wider still. The 'production functions' by which people exploit common pool resources to obtain foods, fuels and fibers include basic ecosystem processes. These may be transformed both by direct management action, and by wider landscape-scale processes such as forest fragmentation. Such extended production functions accordingly reflect both the resource-based activities involved – agriculture, aquaculture, forestry, fisheries, wildlife 'ranching' – and the ecological processes that support those activities. Although the term 'ecosystem services' has been part of the language of environmental and resource economics for decades, the treatment of ecosystem services as the output of extended production functions that include critical ecological processes has become commonplace only since publication of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005; Carpenter *et al.*, 2009).

In what follows I consider how research on the relation between environmental change and economic development has evolved over the last two decades, drawing extensively – but not exclusively – on papers published in *Environment and Development Economics* since its inception in 1995. The journal was originally designed both to report research at the intersection of environmental, resource, and development economics, and to help build the capacity to undertake work of this kind in developing countries. The research recorded in its pages reflects the evolution of research questions, techniques and data used by people working in the field. While the choice of techniques and types of data responds to trends in the parent fields, the topics addressed are unique to the field. They also reflect trends in the identity of the researchers themselves. Not surprisingly, the questions posed by economists from low- and middle-income countries are different from those posed by economists from high-income countries.

My intention is not to provide a comprehensive review of the evolution of the field, but to capture its main characteristics. To do this I consider four broad areas: (a) poverty, population, and common pool resources; (b) the valuation of non-marketed environmental assets and the development of inclusive wealth accounts; (c) market failure and the internalization of local environmental externalities; and (d) trade policy, the internalization of transboundary externality and the provision of global environmental public goods. I conclude by reviewing how the approach to these topics has changed in the last two decades, and considering where the field might go in the future.

2. Poverty, population and common pool resources

The Brundtland Report argued that poverty is both a cause and effect of environmental degradation. Environmental degradation exacerbates poverty both within and between generations, and deepening poverty further degrades the environment (World Commission on Environment and Development, 1987). The testing of that hypothesis has been an important part of the research agenda in the field, with interestingly mixed results. While the protection of common pool resources has been shown to offer benefits to the poor, poverty alleviation has only occasionally improved environmental quality, and has frequently had the opposite effect (Markandya, 2001). The increasing scarcity of common pool resources that satisfy basic needs such as water or fuel wood have long been shown to affect the poor more than the rich (Kumar and Hotchkiss, 1988). It is not therefore surprising that environmental change that reduces the supply of such things might reduce the welfare of low-income groups.

The basic proposition – restated in many papers – is that low-income rural households that are dependent on the exploitation of common pool resources will be adversely affected by the degradation of those resources (Dasgupta, 1996; Barbier, 2010). At the same time, the poor face incentives that lead to the degradation of the same resources. Various mechanisms have been identified in the literature. One is the effect of poverty on rates of time preference, and therefore on the incentive to conserve environmental resources (Perrings, 1989). A study of rates of time preference among rural households in Indonesia, Zambia and Ethiopia, for example, found that poverty in assets, or cash liquidity constraints, was correlated with higher rates of time preference, implying that the poor were less likely to invest in the conservation of such resources (Holden *et al.*, 1998). The authors concluded that poverty reduction might therefore reduce the 'intertemporal externality' due to high rates of time preference.

A second set of mechanisms stem from the effect of poverty on population growth. A number of papers in the 1980s and 1990s advanced evidence that population growth was implicated in environmental degradation in low-income countries (De Janvry and Garcia, 1988; Lopez, 1992; Cleaver and Schreiber, 1994). Among the proposed mechanisms is the tendency for poor households with low expectations of secure future income to respond by increasing fertility rates. This in turn increases pressure on the environment – especially where access to environmental resources is unregulated. Moreover, since unregulated access to environmental resources itself increases uncertainty about future income, there is argued to be a positive feedback between poverty, fertility decisions and environmental degradation (Dasgupta, 1993a, 2001).

Other proposed mechanisms suggest a quite different relation between poverty and population growth. Declining mortality is widely regarded as the main cause of population growth during the demographic transition, and declining mortality is frequently associated with improving living standards (Caldwell, 1976). But so too is migration. A study of deforestation in Belize, for example, found that in-migration accounted for around one-third of deforestation in that country (Lopez and Scoseria, 1996), and in-migration is frequently driven by at least the perception of greater opportunities in the destination area. It is also frequently a feature of either local or regional economic growth poles (Todaro, 1969; Lall *et al.*, 2006; Mansoor and Quillin, 2007).

There is at least some evidence that the relation between poverty and population growth, and the impact of population growth on the environment, are both strongly influenced by institutional conditions. A policy forum constructed around a paper by Paul and Ann Ehrlich (Ehrlich and Ehrlich, 2002) illustrates the points at issue. The Ehrlichs argued that the main challenge for both environment and development was to incorporate the bottom four-fifths of the world's still-expanding population into the global economy while preserving essential life support systems. They asserted that the main threat to life support systems stems from the ecological impact of population growth and increasing levels of per capita consumption. At the same time, they claimed that the institutions needed to manage the threat had not evolved as rapidly as the technologies or consumption patterns that lie behind that threat (Ehrlich and Ehrlich, 2002). This suggests that the population response to consumption growth is institutionally constrained. In a reaction to the paper, Arrow observed that while it was unquestionable that population and consumption growth would, other things being equal, increase stress on planetary resources, there was reason to be more optimistic about the institutional response. Examples from the global North included the emergence of a conservation ethic at the beginning of the 20th century, and the success of at least some of the multilateral agreements established to manage the global commons. He noted that since these things were not responses to current environmental conditions, but reactions to scientific predictions of future adverse consequences, they represented 'a new and higher level of social response' (Arrow, 2002). In fact Sub-Saharan Africa offers several instances where population growth has stimulated productivity increases that have more than compensated for any reduction in environmental resources (Pingali *et al.*, 1987; Tiffen *et al.*, 1994). Some of the more specific responses reported in the literature are described in later sections of this paper.

In all cases the effect of population growth on the resource base increases with the openness of access to common pool resources. Indeed, the role of property rights in mediating the poverty–environment connection has been another persistent topic in the literature. Among the many case studies of this phenomenon, an investigation of the role of common pool forests in alleviating poverty in rural Malawi found that access to forest income significantly reduced measured income inequality, asset-poor households being most reliant on forest access (Fisher, 2004). Whether or not the poverty of people having access to forest resources degrades those resources has generally been found to depend on the rules of access (Ostrom, 1990). A study of the relation between poverty, forest use and dependence, and forest degradation in Iran, for example, found the connections between poverty, forest dependence and forest degradation to vary with institutions for forest management. Depending on forest institutions, poor households or households with high forest dependence did not contribute more to forest degradation than others (Soltani *et al.*, 2014). In fact, it has been a central proposition of the field that institutions largely determine the effect of poverty and population growth on the environment (Heath and Binswanger, 1996).

Interestingly, the evidence on dependence on common pool resources and household income does not all point in the same direction. A study of the contribution of community forestry to household income in Nepal, for example, found that poorer households benefited less from community forestry than richer households. Specifically, the relation between income and the contribution of forests to income had an inverted U-shape. This is largely because households with livestock assets gained more from access to the commons than households without livestock (Adhikari, 2005).

At the country level, the relationship between environmental stress and income was explored in the 1990s in the literature on the so-called environmental Kuznets curve (Stern, 1998, 2004). This literature, stimulated by Grossman and Krueger's assessment of the environmental implications of Mexico's inclusion in the North American Free Trade Area (Grossman and Krueger, 1995), showed that various indicators of environmental quality might be expected first to deteriorate and then to improve as per capita incomes increased. Since the relation mirrored a relation between income and income inequality uncovered much earlier by Simon Kuznets, the relation was dubbed the environmental Kuznets curve.

The relation between per capita income and various indicators of environmental change were studied during the 1990s using a range of databases and econometric approaches. An inverted 'U'-shaped curve was found for the relation between per capita income and various atmospheric pollutants (Seldon and Song, 1994; Shafik, 1994; Cole *et al.*, 1997; Stern and Common, 2001). The relation was not, however, consistent. For some measures of environmental quality it was found to be monotonically increasing (for example, carbon dioxide or municipal waste) or decreasing (for example, coliform in drinking water). For others it was found to have more than one turning point. Moreover, even where the best fit to the data was given by a quadratic function – the inverted 'U' – there were wide discrepancies in estimates of the level of per capita income at which the particular measure of environmental quality started to improve as per capita incomes rose.

What is interesting about this literature is not the specific form of the relation, or the precise value of the turning point(s), but the indirect evidence it offered on the nature of the institutional response to environmental stress. Almost all of the indicators of environmental quality involve environmental public goods of one kind or another. Improvement in those indicators therefore implies more effective environmental regulation, enhanced public expenditure on corrective or defensive measures, or both. A policy forum constructed around an assessment of the environmental Kuznets curve originally published in *Science* (Arrow *et al.*, 1995), summarized what was then known about the institutional response in developing countries. The core argument made by Arrow *et al.* was that the environmental improvements observed by Grossman and Krueger, and those who followed, was evidence that the institutional reforms required to confront resource users with the social cost of their actions had occurred at some times and in some places. It was not evidence that economic growth would, in some sense, take care of the environment. Nor was it evidence that institutional reform would happen in time to avert the worst environmental consequences of economic growth.

The question this raises about the relation between per capita income growth and environmental change is why institutional responses should be weaker in low-income countries than in high-income countries. A range of answers has subsequently been explored in both case studies and cross-sectional and/or panel analyses of country-level data. A common explanation relates to the responsiveness of governments to public pressure. Where environmental damage imposes significant costs on society it is more likely to result in corrective action – whether defensive public expenditures or environmental regulation – if those who hold office are responsible to the public. It has subsequently been hypothesized that the patterns observed in the environmental Kuznets curve studies reflect the relative strength of democratic institutions in low-income and high-income countries (Barrett and Graddy, 2000). A related argument is that one characteristic of non-democratic states is high levels of corruption, and that this allows those who are the source of social cost to block corrective measures (Damania, 2002).

Environmental Kuznets curves have most frequently been found for local air pollutants, while indicators with global, or indirect impacts either increase monotonically with income or else peak at high per capita income levels. Furthermore, concentrations of local pollutants in urban areas tend to peak at a lower per capita income levels than total emissions per capita, while transport-generated local air pollutants peak at a higher per capita income level than total emissions per capita (Cole *et al.*, 1997). What this suggests is that the institutional capacity to deliver local environmental public goods has been easier to develop than the institutional capacity to deliver national or international environmental public goods. But it also indicates that where public bodies have failed to respond to local environmental damage, it may be because there are higher private and public priorities.

The decision process that leads to environmental improvements has also been found to be much less smooth and gradual than one might think. A study of the structural transition of per capita CO_2 emissions and per capita GDP in the 16 countries that had undergone such a transition by the mid-1990s found that the transition correlated not with income levels but with the oil price shocks of the 1970s, and the policies that followed them (Moomaw and Unruh, 1997).

3. Valuing environmental externalities and environmental assets

Since there are few measures to internalize the external effects of resource use, or to secure environmental public goods, it is not surprising that some effort has been expended on the valuation of non-marketed environmental goods and services in developing countries. Nor is it surprising that this has been used to press for reforms of the system of national income accounts.

Methodologically, the evolution of the approach to the valuation of non-marketed resources in papers in *Environment and Development Economics* has mapped to the evolution of valuation approaches elsewhere. Contingent valuation and choice experiment approaches have dominated, and there have been fewer studies using hedonic pricing, benefit transfer or travel cost methods. More surprisingly, there have also been fewer studies using production function methods, although those that have appeared have turned out to be very significant.

Where valuation studies at the intersection of environment and development differ from studies elsewhere is in the ecosystem services and environmental assets treated. There have, for example, been a number of studies using contingent valuation methods to estimate willingness to pay for the services offered by wetlands in Taiwan (Hammitt *et al.*, 2001) and Korea (Kwak *et al.*, 2007), by forests in Ethiopia (Mekonnen, 2000) and India (Köhlin, 2001), by mangroves in Micronesia (Naylor and Drew, 1998), and by marine parks in the Seychelles (Mathieu *et al.*, 2003). More recently, there have been studies of willingness to pay to avoid air pollution in the industrializing parts of the developing world, particularly in China (Du and Mendelsohn, 2011). There have also been contingent valuation studies of the many issues surrounding water supplies, including flood risk in Bangladesh (Brouwer *et al.*, 2009), and drinking water supplies in Brazil (Rosado *et al.*, 2006) and Nicaragua (Vásquez *et al.*, 2012).

The scientific challenges raised by studies of this kind relate less to the nature of the resource being evaluated than to the socio-economic conditions informing peoples' responses. For instance, a dichotomous choice contingent valuation study of willingness to pay to reduce flood risks in Bangladesh found that both subjective risk aversion and objective baseline risk exposure affected stated willingness to pay. However, half of all respondents were unwilling (unable) to pay anything in financial terms, but were willing to make a positive contribution in kind. The authors concluded that combined use of monetary and non-monetary measures of willingness to pay would have significantly reduced the number of zero bids received (Brouwer *et al.*, 2009).

The geographical specificity of willingness to pay estimates has also been shown to limit the applicability of methods commonly applied elsewhere. A study in Costa Rica and Portugal, employing similar contingent valuation surveys of willingness to pay to avoid the effects of seawater pollution, was used to test the scope for benefit transfer methods (Barton and Mourato, 2003). It found that benefit transfer from Portugal led to errors of the order of 100 per cent, whether or not income and other easily accessible socio-demographic variables were controlled for.

There are few examples of revealed preference methods in the literature on developing countries. In one, Pattanayak and Kramer applied producer welfare theory to the valuation of drought mitigation services from protected watersheds in Indonesia. They first characterized the effect of changes in forest cover on hydrological baseflows, then modeled the effect of changes in baseflow for agricultural productivity, and finally estimated the effect of productivity changes for the economic welfare of agricultural households living around the protected watershed (Pattanayak and Kramer, 2001; Pattanayak, 2004). While such methods avoid many of the biases inherent in contingent valuation and related stated preference methods, they do make considerably greater demands in terms of the science, and they do assume that markets exist for the goods produced on the basis of the ecosystem services being valued.

In a parallel effort to improve estimates of the value of goods and services produced in the economy, the 1990s also saw a sharp increase in the number of papers addressing the weakness of aggregate measures of value such as gross domestic product (GDP). The inability of such measures to indicate changes in the value of environmental assets had long been recognized (Repetto *et al.*, 1989; Pearce and Atkinson, 1993). Since the accounts exclude most non-marketed production and consumption, externalities, environmental deterioration and public lands, but include defensive or remedial expenditures (repairing depreciation), they provide unreliable measures of the value of goods and services produced in the economy. Moreover, the errors they contain tend to be greatest where a large proportion of economic activity depends on the exploitation of common pool environmental resources.

In the first issue of the journal, Pearce, Hamilton and Atkinson argued for the development both of a measure of green national income, and for a measure of what they termed 'genuine savings' (gross savings adjusted for loss of assets including environmental assets). They claimed that such a measure was essential to test whether a country was on a sustainable development path. Genuine savings rates that were persistently negative would be evidence that the development path of the country concerned was unsustainable (Pearce *et al.*, 1996). The issue became a major theme of the field with a number of contributions on both green accounts and inclusive wealth (Hamilton and Clemens, 1999; Dasgupta and Mäler, 2000; Haripriya, 2000; Ferreira and Vincent, 2005; Vouvaki and Xepapadeas, 2008; Dasgupta, 2009).

The aim of green accounting is to measure the value of the contribution to human wellbeing made by ecosystem processes, and to incorporate that value into the most appropriate of the national income accounting measures of wellbeing – net national product (NNP) (Cairns, 2002). In the absence of market prices, green accounts require estimation of shadow or accounting prices. Hence there is at least potentially a direct link between the individual valuation studies used to test the efficiency of particular policies or projects involving non-marketed environmental resources, and the development of green accounts. Since NNP is only as accurate a measure of welfare as the prices that are used, it also places a considerable burden on those who attempt to generate accounting prices.

What such studies sought to do was to estimate the value of ecosystem services that are not captured in the market value of resources in private hands. Of the four categories of ecosystem identified in the Millennium Assessment – provisioning, cultural, regulating and supporting – the provisioning services are most likely to be priced in the market, and to be subject to well-defined property rights. There are certainly well functioning markets for some cultural services, such as tourism and recreation, but many cultural services and most regulating services are not priced in the market. Variants of the Millennium Assessment classification have been proposed to address the potential for double counting among such ecosystem services (Boyd and Banzhaf, 2007; Johnston and Russell, 2011). To the extent that the regulating and supporting ecosystem services needed for agricultural production are reflected in the price of the private land, for example, they will be appropriately measured in the system of national accounts. However, the offsite benefits or costs of land management – flows of nutrients, pests and pesticides, siltation of rivers and the like – are not. The task is not therefore to account for all ecosystem services. It is to account for ecosystem services that are not already explicitly or implicitly priced (and so reflected in the national income accounts), and that have a significant impact on wellbeing (Perrings, 2012).

A study of the national income accounts in Chile identified several potential sources of bias in the standard approach, including the structure of property rights. It estimated four measures of green income from Chile's mining sector for the period 1977–1996. The measures differed in the method used to estimate the value of mineral resources and exploration expenditures. The study showed that the standard approach overestimated the income generated by the Chilean mining sector during the period by between 20 and 40 per cent, and its rate of growth by between 3 and 20 per cent. Moreover, the bias was similar for the different methodologies used (Figueroa *et al.*, 2002). There have also been studies of the effect of including positive production externalities. A study of the impact of carbon and hydrological externalities in forests in South Africa, for example, estimated that they added around 0.6 per cent to the measure of value added in NNP (Hassan, 2000; Hassan, 2003).

The second challenge identified by Pearce in 1996 was the development of better estimates of the value of the underlying environmental stocks or wealth accounts. Since the value of any stock is simply the discounted flow of services it yields, there is a connection between attempts to improve estimates of the value of the flows derived from ecosystems and attempts to improve estimates of change in the value of the underlying stocks. The flow estimates make it possible to determine the social opportunity cost of particular activities. The stock estimates make it possible to check the sustainability of a program of activities – a development path. Of the two main measures of changes in environmental stocks, both genuine savings (now 'adjusted net savings') and inclusive wealth are designed to provide improved estimates of the depreciation of natural assets.

The central requirement of a sustainable consumption program is that it should not reduce the consumption possibilities available to future generations. This idea was embodied in Lindahl's definition of 'income' as the maximum amount that could be consumed without reducing the value of the capital stocks available to future generations (Lindahl, 1993). Income in the Lindahl sense is equivalent to NNP or net national income. The common theoretical foundations of efforts to build wealth accounts from this lie in the work of Solow and Hartwick in the 1970s. They showed that a necessary condition for a consumption program based on the exploitation of depletable natural resources was that the rents from resource depletion should be reinvested in reproducible assets (Solow, 1974a, 1974b, 1986; Hartwick, 1977, 1978). By the 1990s, these ideas were being harnessed to the task of constructing wealth accounts (Hartwick, 1990, 1994, 2000; Pearce and Atkinson, 1993; Hamilton, 1994; Pearce *et al.*, 1996; Hamilton and Clemens, 1999).

Initially, the corrections to measures of gross national savings recorded in the national income accounts focused on the depreciation of mineral and other non-renewable resource stocks, but these were later extended to include at least some renewable resource stocks (standing forests), and proxies for the stability of the general circulation system (carbon emissions). Currently, adjustments to gross savings include: (a) subtraction of the depreciation of produced capital; (b) addition of expenditure on education as a proxy for investment in human capital; (c) subtraction of the rents on depleted resource stocks; and (d) subtraction of specific pollution damages. The resource stocks currently included comprise energy (oil, gas and coal), minerals (non-renewable mineral resources) and forest (rent being calculated on timber extraction in excess of the 'natural' increment in wood volume). Pollution damages currently recorded include carbon dioxide and PM10 damages.

The results, reported in the World Bank's time series for adjusted net savings, and associated estimates of changes in aggregate wealth (World Bank, 2006, 2011), showed that, when corrected for the depreciation of natural assets, adjusted net savings in many resource-dependent economies were negative for much of the last quarter of the last century. Indeed, in per capita terms, Sub-Saharan Africa appears to have lost around one-third of its wealth in that period (Dasgupta, 2001). This outcome is consistent with the observation that resource-rich developing countries generally performed worse, economically, than resource-poor developing countries in that period – the so-called 'resource curse'. The observation incidentally prompted investigation of why this should have occurred, which showed that two critical features were frequently missing: that there should be a responsible agency to recover and invest resource rent, and that investments in alternative assets should yield as much as the natural capital they replace (Lange, 2004; Lange and Wright, 2004).

Latterly, a number of papers have considered how best to extend the corrections to gross national savings to encompass changes in the state of ecosystems (Barbier, 2010, 2013). Most recently, a report by the United Nations University, the International Human Dimensions Program and the United Nations Environment Programme offered preliminary estimates of changes in inclusive wealth for a selected set of countries, based largely on market prices and a different aggregation procedure from that used by the World Bank (UNU-IHDP and UNEP, 2012). Since the measure of environmental wealth in the report included only assets for which there existed market prices, it did not correct for externalities. A second attempt to estimate comprehensive wealth, reported in *Environment and Development Economics*, used World Bank data for stocks of produced and natural capital, but separately estimated stocks of human capital (Arrow *et al.*, 2012).

Investments in human capital were calculated from projected changes in the labor force and labor productivity associated with different levels of education, and from changes in human health. Estimates of investment in natural capital were similar to those made by the World Bank, depending on the Hotelling assumption that exhaustible resource prices would rise at the rate of interest. Changes in selected resource stocks were then weighted by their shadow price: the impact of a marginal change in those stocks on wellbeing. For non-renewable resources, the change in resource stocks was the amount extracted and the shadow value was the resource rent. For renewable resources, the change in stocks was the difference between the natural rate of regeneration and harvest, and the shadow value was calculated in the same way. Focusing on five countries, the USA, China, India, Brazil and Venezuela, it was shown that natural capital had declined in all countries except for the USA, and that the increase in total (comprehensive) wealth was mainly due to investment in human capital (Arrow *et al.*, 2012).

Among the responses to the paper, Hamilton (2012) considered the differences between the estimates reported and those reported by the World Bank (2011). He conjectured that the discrepancies between Arrow *et al*. and the World Bank for produced and natural capital were linked to differences in data sources, and assumptions about depreciation and discount rates, but drew particular attention to discrepancies in estimates of the value of human capital. He observed that the big difference in the two wealth accounts lies in the value of health. Whereas the World Bank followed the system of national accounts in this respect, Arrow *et al*. started from the US value of a statistical life, converted this to the mean value of a life-year, and then multiplied by the discounted expected years of life to arrive at total 'health capital'. Hamilton questioned whether the value of a statistical life could so narrowly be tied to health, arguing that it more properly reflects everything that people value in life – consumption of goods and services, leisure, environmental amenity and the like, in addition to good health – and hence that it more closely approximates the value of intergenerational wellbeing (Hamilton, 2012).

At this stage, the most that can be said is that, although the estimation of inclusive wealth has been an abiding concern of those working at the intersection of environment and development, it remains work in progress. The identification and valuation of many critical ecosystem functions has yet to be resolved satisfactorily. Nor do we yet have a solution to the problem of accounting for the environmental drivers of total factor productivity growth (Perrings, 2012). In the meantime the intergovernmental organizations are moving ahead with the development of satellite accounts – the *System of Environmental and Economic Accounts* (SEEA), still under development by the UN, the EC, the OECD, the IMF and the World Bank. The SEEA (2003 version) includes measures of the effect of environmental change on capital stocks. As in Arrow *et al*. (2012), UNU-IHDP and UNEP (2012) and World Bank (2011), it takes changes in aggregate capital as a test of sustainability. Development is regarded as unsustainable if it relies on stocks of natural capital, and these are degraded to the point where they are no longer able to adequately provide what are referred to in the SEEA as 'resource', 'service' or 'sink' functions (loosely corresponding to the MA provisioning, cultural and regulating/supporting services). The SEEA comprises four accounts: flow accounts for pollution, energy and materials; environmental protection and resource management expenditure accounts; natural resource asset accounts; and valuation of environmental depletion, degradation and defensive expenditures. The SEEA's ecosystem assets deliberately introduce an element of double counting. As long as the services they yield are recorded in physical terms, the double counting is not an issue, but when the assets are valued, this does not make as much sense.

Issues still to be addressed include the fact that the system of accounts excludes a number of natural resources that are important to human wellbeing, but that cannot be privately co-opted – especially resources in public ownership or those that lie beyond national jurisdiction. The stock of assets should include all lands that generate off-site benefits or costs as a result of environmental flows. This includes built environments: urban and industrial ecosystems that generate benefits and costs to people that are sometimes similar and sometimes different from ecosystems in other areas.

In developing countries, the dependence of many people on the non-market exploitation of natural resources in open or weakly regulated access common pool resources is not currently reflected in the accounts. The evidence from the adjusted net savings estimates suggests that many of the poorest countries have seen the value of their assets decline over much of the last four decades. But this cannot be confirmed in the absence of comprehensive wealth estimates.

4. Addressing environmental externalities

Given the pervasiveness of market failures in developing countries, much of the work at the intersection of environment and development has been concerned with why markets either fail to emerge or are ineffective, and with the design of instruments to internalize externalities or to improve market performance. Dasgupta has recently observed that the externalities in the chain linking poverty, population growth and the degradation of natural resources in poor countries have prompted those working at the intersection of environmental and development economics to revise the economics of the household and other non-market institutions (Dasgupta, 2013). As in the development economics literature, much has been made of the lack of well-functioning capital, labor and land markets, and the effect this has had on the exploitation of common pool natural resources (Dasgupta, 1982, 1993a, 1996, 2001; Barbier, 2010; Groom and Palmer, 2010). However, since the primary focus of work in the field has been on the environmental implications of development strategies under different institutional and biophysical conditions, most attention has been paid to the efficiency implications of missing markets for environmental assets.

The overexploitation of many common pool resources, including surface and groundwater, rangelands, wetlands and forests, stems from the rules governing access, and the incentive users have to neglect their effects on others. This generates reciprocal externalities of the kind characterized by Hardin as the 'tragedy of the commons' (Hardin, 1968). As has already been remarked, it has given rise to numerous studies of the institutional reforms required to address the problem – frequently drawing on the approach pioneered by Ostrom (1990). Examples include the analysis of the external effects of the overdraft of groundwater for irrigation in India. The solutions in such cases typically include the allocation of property rights, mechanisms to regulate extraction, and the development of infrastructure to enhance groundwater recharge (Reddy, 2005; Diwakara and Chandrakanth, 2007).

One of the more interesting effects of property rights in natural resources in developing countries is the impact it has on migration decisions. Studies of the relation between property rights and the propensity of herders to move with their livestock in Kenya, for example, showed that the establishment of well-defined property rights discouraged migration with livestock. Under common property rights migration is productivity enhancing. Under private property rights it is not. It was also shown, however, that the establishment of private rights more than compensated for the productivity losses in restricting migration (Kabubo-Mariara, 2003, 2005). In a similar way, the propensity of people dependent on forests to migrate in search of employment opportunities in India has been shown to be highly sensitive to the nature of property rights. Reacting to the conventional wisdom that environmental degradation leads to stress migration from rural areas, Chopra and Gulati showed that changes in the institutional arrangements governing access to forests in India had the opposite effect (Chopra and Gulati, 1998).

Some results from work on common pool resources in developing countries are likely to be fully general. One example is the impact of the discount rate on the incentive effects of the payoff to non-compliance with regulated access rules. Akpalu considered non-compliant activity that generates a flow of returns until the offender is caught and punished. Using an artisanal fishery in Ghana as an illustration, he showed how sensitive non-compliance was to the discount rate being applied (Akpalu, 2008; Akpalu and Normanyo, 2014).

The literature on the linkages between environment and development includes fewer studies of classic unidirectional local externality problems of the type that motivated the Coase 'theorem' (Coase, 1960). Nevertheless, there are some (Jack, 2009). One very good example is an analysis of soil erosion externalities between upstream (extensive) and downstream (intensive) agriculture in the Philippine province of Palawan, in which downstream labor productivity and demand were affected by upstream erosion. The author showed that environmental payments to upland households to allocate labor away from erosion-increasing activities had the potential to enhance overall productivity and welfare (Shively, 2006). There are, however, a growing number of studies of the process by which markets for the offsite benefits of land use are being created in developing countries (for a review, see Arriagada and Perrings, 2013). Many are on a small scale and involve either local watershed or biodiversity protection linked to ecotourism operations. Others are on a much larger scale, and address the international externalities of land use and land cover change in developing countries. The bigger challenges lie in cases where externalities are jointly produced. For example, fossil fuel combustion produces both CO_2 and SO_2, one a major greenhouse gas, the other a local pollutant that also happens to alleviate global warming through its effect on radiative forcing. The result is a pair of negatively correlated local and global stock externalities (Yang, 2006). I return to the question of market creation later.

There is a very marked difference in the treatment of environmental taxes in developing and developed countries. Many of those writing on the introduction of environmental taxes in the global North have focused on the side-effects of tax reforms: the 'double dividend' to be had from the introduction of environmental taxation. This is the double benefit from reforms that move existing taxes closer to the optimum – that they deliver both a welfare benefit and an environmental benefit (Bovenberg and van der Ploeg, 1994; Goulder, 1995; Bovenberg and Goulder, 1996; Bosello *et al.*, 2001; Sartzetakis and Tsigaris, 2009). There are certainly some studies of the double dividend in developing countries. For example, a study of carbon taxes to reduce emissions of CO_2 in China showed that the introduction of a revenue neutral carbon tax would transfer income from consumers to producers, leading to increased investment, and hence to a double dividend of reduced emissions of CO_2 and a long-run increase in GDP and consumption (Garbaccio *et al.*, 1999). However, the topic has not generally attracted as much attention in developing countries as it has in developed countries.

One reason is that the governments of developing countries are more concerned about the distributional effect of taxes and other economic instruments than they are about the efficiency gains. A review of the double dividend literature in 2001 noted that, despite progress in modeling the conditions under which a doubled dividend might be realized, there were few convincing empirical studies or tests of the sensitivity of the double dividend to alternative ways of recycling environmental fiscal revenues in developing countries (Bosello *et al.*, 2001). An earlier assessment of interactions between tax and environmental policy by Whalley had concluded that future research would likely go beyond the double dividend to discuss internalization of environmental externalities via tax policy, abandoning ideas of revenue neutrality, and concentrating on environmentally harmful methods of production and distributional issues (Whalley, 1999).

The importance of the distributional effects of efforts to internalize environmental externalities is particularly obvious in the treatment of deforestation. The environmental benefits of a reduction in the rate of deforestation, or an increase in the rate of afforestation, lie in their role in sequestering carbon. In establishing systems of payments for the climatic effects of enhanced carbon sequestration, governments, NGOs and the intergovernmental organizations have all been motivated by the fact that those payments at least potentially alleviate poverty (Ferraro and Simpson, 2002; Pagiola *et al.*, 2007; Bulte *et al.*, 2008; Pagiola, 2008; Pagiola *et al.*, 2008; Wunder, 2008; Wunder and Wertz-Kanounnikoff, 2009). Evidence of the poverty alleviation role of payments for ecosystem services schemes is sparse. Indeed, a review of extant schemes found that, while many payments for ecosystem services schemes have the potential to provide benefits to the poor, actual benefits are generally very small relative to governments' poverty alleviation goals. Moreover, schemes where poverty alleviation is the primary goal are less effective in securing conservation or carbon benefits than schemes that focus on the delivery of specific ecosystem services (Pattanayak *et al.*, 2010).

Nevertheless, payments for ecosystem services are the most popular instrument for the internalization of the positive externalities involved in the provision of environmental public goods. This is precisely because they appear to be able to offer benefits to poor land holders while securing the global public interest in conservation and climate regulation. Analysis of schemes of this sort has likewise become a popular research topic. The largest of such schemes, the scheme for reducing emissions from deforestation and forest degradation (REDD), is also the most studied (Brown *et al.*, 2008; Angelsen *et al.*, 2009; Chhatre and Agrawal, 2009; Long, 2009; Blom *et al.*, 2010; Phelps *et al.*, 2010; Henry *et al.*, 2011). It aims to promote forest conservation, sustainable management of forests and enhancement of forest carbon stocks, and is directly motivated by the fact that deforestation is a significant source of carbon emissions and hence climate change. Economists who have considered its potential impact on both efficiency and equity have concluded that these things depend heavily on REDD policy design and implementation. It has, for example, been argued that while REDD has the potential to curb deforestation, to reduce the cost of climate mitigation and to alleviate poverty, it also has the potential to accelerate deforestation and to become the next resource curse (Bosetti *et al.*, 2011).

From an efficiency perspective the attraction of REDD (as with many other payment for ecosystem services schemes) is that it provides private incentives for the provision of a public good. There are currently five main institutional options for the provision of environmental public goods (Stavins, 2003): (a) regulations such as zoning restrictions, harvest quota, open and close harvest seasons, or the informal rules that govern the use of common-pool resources; (b) direct incentives such as payments for ecosystem services, taxes, user charges and access fees; (c) mixed regulation and incentive systems such as cap-and-trade schemes, tradable fisheries quotas and tradable development rights; (d) self-regulation and voluntary agreements; and (e) direct investment in the ecological and manufactured infrastructures needed for ecosystem service provision such as the protection of water supply, habitat for endangered species or storm damage protection. Which approach is the most appropriate depends on both biophysical and social conditions.

The evidence for efficiency gains from payments for ecosystem services schemes is, however, quite mixed. While such schemes do involve transactions between buyers and sellers of a service, they do not often have the characteristics of well-functioning markets. They do not, for example, allow free exit and entry, or iteration towards a market-clearing price. The 'prices' used in payments for ecosystem services schemes are frequently insensitive to changing conditions (Kinzig *et al.*, 2011). Indeed, while they aim to internalize externalities, payment schemes themselves may also be the source of external effects. There are many instances where a focus on a single service has had unanticipated consequences, among them being the impact of incentives for biofuels production on the conversion of wildlands (Hill *et al.*, 2009), conversion of wetlands and stream corridors that reduces the water-purification capacity of ecosystems (Boyer and Polasky, 2004), channel dredging that inhibits sediment delivery and reduces storm protection (Montague, 2008), or homogenization of agricultural landscapes that reduces the capacity of ecosystems to contain pest or pathogen outbreaks (Hutchinson *et al.*, 2009). Payment systems that focus on a single service may be expected to have the same effect. The bigger issue, though, is that if payments for ecosystem services schemes are to signal the true scarcity of ecosystem services, they cannot simultaneously be aimed at poverty alleviation. Pursuit of that particular double benefit is likely to compromise both objectives (Kinzig *et al.*, 2012).

5. International trade, aid and global environmental public goods

The archetype of the developing economy is a small, open, resource-dependent country in which growth is initially based on mining, forestry, fisheries and agriculture. The environmental impacts of development in such economies have typically been analyzed from two perspectives. The first addresses the problem of resource dependence. It concerns the optimal extraction of natural resources, and the identification of conditions under which development based on either exhaustible or renewable resources may be efficient and sustainable. This perspective evaluates development strategies on the basis of their treatment of (a) resource rents and (b) the environmental external effects of mineral extraction and the harvest of wild and domesticated species. The second perspective addresses the openness of developing economies. It evaluates development strategies in terms of their treatment of international trade and transfers – a major concern of development economists – but also in terms of their treatment of the externalities of international trade, and the supply of international environmental public goods.

The core theory of natural resource-based economic growth stems from Hotelling's original work on the optimal exploitation of exhaustible resources (Hotelling, 1931), the extension of the results obtained for exhaustible resources to biological systems (Clark, 1973, 1976), and their generalization to a theory of the relationship between economic growth and environmental change (Mäler, 1974; Dasgupta and Heal, 1979). The core theory of sustainable resource-based economic development stems from work by Solow and Hartwick on the conditions for the sustainable exploitation of environmental resources (Solow, 1974b; Hartwick, 1977, 1978). Dasgupta has recently argued that, while standard approaches in development economics recognize the importance of the accumulation of productive assets, they limit themselves to a subset of those assets and largely ignore environmental assets. Citing recent books by Bhagwati and Panagariya (2013) and Dreze and Sen (2013) as examples, he claims that development economics is 'built on a model which presumes that in any institutional setting, a combination of labour (more broadly human capital), knowledge, and reproducible capital is the basis of production, exchange, and consumption. Nature doesn't get a look in except as a bit player' (Dasgupta, 2013). By contrast, a substantial body of literature on the development of resource-dependent economies aims to quantify the value of renewable and non-renewable resources when property rights are ill defined, in order to test whether national resource-based development strategies satisfy the Hartwick condition on resource rents (Dasgupta, 1993b; Hamilton, 1995; Dasgupta and Mäler, 1996, 2000; Hassan, 2000; Lange, 2004; Lange and Wright, 2004; Hamilton and Hartwick, 2005).

The existence of the resource curse in the development literature was ascribed to a range of factors quite independent of the value of the resources themselves – including the impact of resource abundance on domestic prices in developing economies, poor economic policies and their lack of openness to international markets (Sachs and Warner, 1997, 2001). The motivation for the development of wealth accounts with which to test the Hartwick condition in resource-dependent economics was a concern that the investment of rents within the country failed to generate assets of equivalent value to the depleted, degraded or depreciated natural assets. A study of the phenomenon at a sub-national scale within a country that was both resource dependent and had yet experienced significant economic growth, Malaysia, found that whereas the economy as a whole had developed sustainably by the Hartwick criterion, this was not the case in two of three regions. In those regions, trends in net investment and net domestic product associated with the exploitation of natural resources indicated that consumption levels were unsustainable (Vincent, 1997). The question this poses is what is the appropriate spatial scale at which to assess the sustainability of resource extraction.

The diversion of resource rents and their role in building assets elsewhere is not a new story. The sustainability of consumption patterns in the resource-rich regions themselves then depends on the entitlements that people have to the yield on remote investments. There are two sets of issues involved. One is the assignment of property rights in the exploited resource. In most economies, rights to all natural resources are vested in the state. So while multilateral agreements like the Convention on Biological Diversity oblige member states to implement benefit-sharing measures for those who have custody of wild living resources, the royalties paid for exploiting those resources accrue to the state. This set of issues has been relatively neglected (Cooper, 2001). The second set of issues relates to the potential gains from trade in resource-based commodities. These issues span the role of trade policies in stimulating trade, and the mechanisms for ensuring that the gains from trade accrue to people in resource-rich regions. They center on the relationship between trade liberalization, growth and the environment.

Much of this literature has been cast in terms of the scale and composition effects of trade liberalization, the greater levels of both depletion and emissions associated with higher levels of output being the scale effect, and the reallocation of resources towards activities in which a country has comparative advantage being the composition effect. The two effects have the capacity to move in the same or opposite directions depending on the nature of environmental policies in all countries (Abler *et al.*, 1999; Barrett, 2000; Coxhead and Jayasuriya, 2004; O'Ryan *et al.*, 2011).

Once again, the standard approach in development economics has been to concentrate on the welfare gains to be had from liberalization, and to discourage – as Bhagwati put it – 'burdening trade treaties and negotiations with social agendas' (Arda, 2000; Bhagwati, 2000; Griffin, 2000). The perception that the regulation of environmental assets involves a 'social agenda' that is different from the regulation of other assets turns out to be a point of disagreement between development and environmental economists. There are several reasons why multilateral trade negotiations should reflect the environmental impacts of trade. The most important reason is the existence of transboundary environmental externalities (spillovers) that, in some cases, are large enough to compromise the welfare gains from trade (Cooper, 2000). Others include the fact that, if targeted multilateral environmental agreements are not an option, the second-best solution may not be to liberalize trade anyway (Repetto, 2000), and if targeted multilateral environmental agreements are an option, that trade and environmental multilateral agreements may be mutually reinforcing (Xepapadeas, 2000).

Since it is recognized that environmental policy has frequently been used as a surrogate for trade policy, there is also a case for ensuring that trade considerations are part of multilateral environmental negotiations. Where trade reforms have made it impossible for countries to maintain domestic protection, environmental policy has become a second-best instrument.

Trade liberalization has induced importers of environmentally harmful goods to weaken environmental standards, and exporters of environmentally harmful goods to raise their environmental standards (Cooper, 2000). It has been shown that when pollution is local, trade liberalization without any constraint on environmental policy can induce a non-cooperative game between countries in pollution policy. Indeed, in the absence of agreement on environmental policy, trade negotiations are unlikely to lead to an efficient outcome. When pollution is global, countries importing pollution-intensive goods have an incentive to try to link trade agreements with environmental agreements, while countries exporting pollution-intensive goods have an incentive to try to keep trade and environmental negotiations separate (Copeland, 2000).

The central point at issue is the fact that the social opportunity cost of natural resources reflects the value of all the services and disservices they provide, including any transboundary externalities. In the last decade the transboundary externalities of resource depletion that have attracted most attention have been those associated with the emission of greenhouse gases, particularly carbon dioxide and nitrous oxide. The impact of deforestation and forest degradation on emissions and sequestration of carbon dioxide, and hence on climate change, have stimulated numerous studies of the options for internalizing these externalities. Initially these focused on the instruments being developed under the Kyoto Protocol of the UN Framework Convention on Climate Change (Rose *et al.*, 1999; Millock, 2002; Tietenberg, 2003), but more recently have focused on ad hoc measures such as the REDD+ scheme already described.

REDD+ recognizes that among the services generated by forest cover is a global public good – the regulation of climate change – and that this is an important component of the value of forests to humankind (Barrett, 2007). It follows that any test of the sustainability of deforestation that appeals to the insights of Solow and Hartwick should include estimates of the forgone benefits of global climate regulation in the valuation of forest systems. As Pearce had put it in the 1990s, there are compelling reasons for doing a 'global cost-benefit analysis' of the drivers of climate change (Pearce, 1998). Forest clearance may pass a private cost-benefit test, but fails a social test at most scales. In fact analyses of the expected costs of climate change by region have consistently shown that developing countries are likely to be more strongly impacted than developed countries, in part because of their greater dependence on natural resource-based economic activities (Mendelsohn *et al.*, 2006; Kala *et al.*, 2012; IPCC, 2014). While measures such as REDD+ are designed to change the private calculus, however, the main determinant of private land use decisions continues to be the international price of foods, fuels and fibers – no account being taken of externalities.

Increasing carbon emissions and declining carbon sequestration are not the only transboundary externalities of resource-based economic activities. The closer integration of agriculture, aquaculture, forestry and fisheries in developing countries in the global economy has also been responsible for a rapid increase in the rate at which species are being dispersed across national borders, sometimes as traded commodities, and sometimes as an incidental effect of trade (Perrings, 2010; Perrings *et al.*, 2010). The introduction of domesticated species in agriculture, and the accidental introduction of pests and pathogens along with traded agricultural inputs and outputs, have led to the establishment and spread of many invasive species (Williamson, 1996, 1999; Mack *et al.*, 2000). These have been a source of significant damage (Olson, 2006; Pejchar and Mooney, 2010), with particularly adverse effects recorded in developing country agriculture (Rangi, 2004), freshwater fisheries (Kasulo, 2000), marine fisheries (Knowler, 2005), groundwater (Turpie and Heydenrych, 2000; van Wilgen and Richardson, 2010). Of particular concern are emerging zoonoses – diseases that cross from animal to human hosts – many of which originate in developing countries as forest systems are cleared for agriculture or tapped for timber and non-timber forest products (Daily and Ehrlich, 1996; Jones *et al.*, 2008). In most cases the dispersal of pests and pathogens is a direct result of trade (Levine and D'antonio, 2003; Perrings *et al.*, 2005; Costello *et al.*, 2007; Waage *et al.*, 2009).

Within the environment and development literature this has led to a number of attempts to characterize countries in terms of their vulnerability to invasive species (McNeely, 2001; Perrings, 2007; Gren *et al.*, 2011), to identify the implications for the management of the resource-based sectors (Albers *et al.*, 2006; Gaff *et al.*, 2007; Rai and Scarborough, 2013), and to discuss the implications for the regulation of trade, travel and transport (McAusland and Costello, 2004; Waage and Mumford, 2008). As in the case of deforestation, however, the invasive species externalities of resource use are seldom taken into account in national or international market interactions, and the default response under the relevant multilateral agreements is temporary trade bans imposed after an outbreak.

6. Future directions

Since a number of strands in the literature on environment and development are best characterized as work in progress, they may be expected to be recurrent themes in the future evolution of the field. Among these I include the development of measures of income and wealth that more fully capture the value of natural assets, particularly those in the public domain. I also include the impact of poverty and poverty alleviation on the use made of natural resources. The treatment of both has, however, changed over the last two decades. The adjusted net savings measures generated by the World Bank originally focused on the depletion of exhaustible natural resources – oil and minerals – but are increasingly focused on biological stocks. This partly reflects the rise of climate change as an issue, which has led to an upward revision of the value of standing timber. But it also reflects awareness that ecological systems typically generate multiple services of which carbon sequestration and the production of foods, fuels and fibers are just a few (Barbier, 2008).

The treatment of poverty and poverty alleviation has also changed. It is now less tied to demographic change, and is more frequently regarded as an ancillary benefit of measures to improve the management of forests (López-Feldman *et al.*, 2007; López-Feldman and Wilen, 2008; Ojha, 2009; Soltani *et al.*, 2014), water resources (Waage *et al.*, 2005), and other features of the landscape (Uchida *et al.*, 2007). Most noticeable in recent years has been the tendency – already noted – to treat poverty alleviation as a side benefit of payments for ecosystem services (Bulte *et al.*, 2008).

I would expect to see increasing coverage of three other phenomena. One is due to the changing balance of the world economic system. The growth of China, India, Brazil, Mexico and South Africa (among others) as industrial producers has affected both the proportion of papers addressing issues in those countries, and the nature of the issues being addressed. While all of these countries still have elements of the archetypal developing economy, they are all increasingly urbanized, and are all increasingly experiencing the challenges of rapid industrialization when environmental regulations are weak. The central question posed in the paper that originally stimulated this review (Greenstone and Jack, 2013) is most relevant in Beijing, Delhi, Sao Paulo, Johannesburg or Mexico City, where air, soil and water pollution are major issues.

The coverage given to China in recent years illustrates the trend nicely. This year's special issue on environmental policy in China includes papers that span the range of problems addressed in both developing and developed economies (Xu and Berck, 2014). It includes, for example, papers on the exploitation of common pool resources (Yi *et al.*, 2014) and agri-environment (payments for ecosystem services) schemes (Yang and Xu, 2014), but also on air pollution in industrial cities (Jin and Lin, 2014; Qin *et al.*, 2014). Over the last decade, there has been a steadily growing number of papers in the journal on four main questions: CO_2 emissions and climate change (Garbaccio *et al.*, 1999; Aunan *et al.*, 2007; Carraro and Massetti, 2012; Qi *et al.*, 2013a, 2013b), non-CO_2 pollution (Yongguan *et al.*, 2001; De Groot *et al.*, 2004; Di, 2007; He, 2009; Bao *et al.*, 2011; Shi, 2011; Duvivier and Xiong, 2013; Jin and Lin, 2014), land conservation and afforestation (Shen *et al.*, 2006; Sun *et al.*, 2006; Yang and Xu, 2014; Yi *et al.*, 2014), and water management (Huang *et al.*, 2008, 2010; Tang *et al.*, 2012). A similar balance between papers on pollution and depletion in China occurs in other environmental journals. I would expect this broad trend to continue into the future. There is likely to be an increasing emphasis on the larger developing economies and on the environmental issues associated with urbanization and industrialization.

A second phenomenon I would expect to see covered in greater depth in the future is the dispersal of species as a result of both trade and climate change. To this point, economists interested in the global environmental consequences of economic development have focused almost exclusively on the effect of carbon emissions on the global climate. While this is likely to have a significant impact on welfare, it is not the only global impact of economic development, and may not be the most important. Outside the journals on environment and development, considerable attention is currently being paid to the role of developing countries in the emergence and transmission of infectious zoonotic diseases (Jones *et al.*, 2008; Beutels *et al.*, 2009; Martin *et al.*, 2011; Morse *et al.*, 2012). At the same time, horizon-scanning exercises aimed at identifying significant future threats have identified the consequences of species dispersal – and especially the dispersal of pests and pathogens – as among the most severe (King *et al.*, 2006). Since the control of disease risk is an international public good, the problem of species dispersal has at least some of the characteristics of the climate change problem, although the strategic interests of nation states in the problem are different (Barrett, 2003, 2005; Sandler, 2004). I would expect this issue to become more prominent in the future.

The closer integration of the global economy, and the increasing importance of managing global resources in the public domain, is also likely to raise the emphasis given to a third area: international environmental agreements. This has been an intermittent concern in this journal over the past two decades (Barrett, 1994, 2000; Fernandez, 2002). It received some attention after the Kyoto Protocol was agreed, but has not received attention consistently. More recently, however, the flow of papers addressing both theoretical and empirical aspects of multilateral agreements has increased (Bayramoglu and Jacques, 2011; Caparrós and Péreau, 2013; Dinar and Zaccour, 2013; Pavlova and de Zeeuw, 2013). As might be expected, a key issue is the capacity for multilateral agreements to be self-enforcing when countries are asymmetric with respect to the costs and benefits of environmentally damaging activities. An encouraging finding of this work is that asymmetries in costs and benefits can yield large stable coalitions, irrespective of whether there are transfers between signatories. However, consistent with the wider literature on the topic, such large stable coalitions are also only possible if the gains from cooperation are small. In cases where the potential gains from cooperation are larger, transfers become more important (Pavlova and de Zeeuw, 2013).

Finally, a potential growth area I am less confident about is the interdisciplinary modeling and management of coupled human–natural systems. One motivation for the establishment of *Environment and Development Economics* was the provision of a forum in which the insights of both the natural and economic sciences could be brought to bear on the links between environmental change and economic development, but it has proved harder to realize this goal than others. Over the last two decades natural scientists have been engaged in a series of policy fora and synthesis papers (Arrow *et al.*, 1996; Bolin, 1998; Levin *et al.*, 1998, 2013; Ehrlich and Ehrlich, 2002; Ehrlich, 2008), but the majority of contributions to the journal derive from economists whose primary home lies in either environment or development. This is not to decry these contributions. However, improvement in our capacity both to understand the interactions between economic and environmental processes, and to develop instruments that will not have unanticipated consequences, depends on breaking down the barriers between economics and the natural sciences. Among the most important insights from theoretical ecology is that the ability of systems to function over time depends on their resilience – a measure of the capacity of the system to maintain functionality when subject to stress or shocks. This is turn depends on heterogeneity within the system, which we can think of as the size of the portfolio of assets available to the system. Although the implications of the approach for the stability of resource-dependent developing economies have attracted some attention (Perrings, 2006), this remains a Cinderella problem. That said, it is too early to say whether the latest attempt (Levin *et al*., 2013) to bring these arguments before economists working on environment–development interactions will bear fruit. My hope is that it will.

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