ASSESSING THE ENVIRONMENTAL IMPACTS OF CGIAR RESEARCH: TOWARD AN ANALYTICAL FRAMEWORK

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Introduction

During the course of its nearly forty years in existence, the genetic improvement, natural resource management, and policy research of the Consultative Group on International Agricultural Research (CGIAR) has generated a broad array of technology, management, and knowledge products. These have produced a similarly broad set of economic, social, and environmental impacts. Over the past two decades formal *ex post* assessment of these impacts has become increasingly institutionalized within the CGIAR (Walker, et al. 2008). This emphasis has followed on the heels of escalating demands on the part of donors and System managers for evidence that specific research investments have generated a large benefits and a reasonable rate of return.

Not all of these impacts are easily measured, however. For example, the current state of knowledge regarding economic impacts of crop genetic improvement (CGI) improvement technologies far outstrips that for natural resource management (NRM) and policy research (Renkow and Byerlee 2010). And while a large body of evidence documents and quantifies the direct and indirect pecuniary effects of CGIAR research using economic surplus approaches (e.g., Evenson and Gollin 2003; Raitzer 2003), only a very few studies quantify social impacts (on poverty and gender issues) or environmental impacts.

Ideally, a unified analytical approach would consider jointly the impacts across all three of these dimensions – economic, social, environmental. Achieving this ideal is constrained, however, by two factors. First, economic impacts are far more readily measured than social or environmental impacts in terms of monetary estimates compatible with cost-benefit analysis. Economic impact assessments benefit from a ready-made metric for analysis – market prices for traded goods and services whose existence can be attributed to research outputs. Combining price and quantity data renders economic impact assessment a relatively straightforward (although by no means easy) exercise. In contrast, social and environmental impacts arise to a large degree from changes in flows of goods and services for which there is no market. Missing markets for environmental or social goods pose significant – and to date largely

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unresolved – valuation problems . Second, social and environmental outcomes of a given research endeavor result from fundamentally more complex interrelationships among humans or between humans and their natural environment. This too renders social and environmental impact assessment a much more difficult task.

At the outset, it is useful to distinguish a bit more clearly between "economic," "environmental," and "social" impacts. Here we will follow Bennett's (2008) lead in making these distinctions. *Economic* impact will refer to changes in flows of agriculture-related goods and services – both inputs and outputs – that are transacted in markets.[2] In contrast, both social and environmental impacts will refer to flows of goods and services that are to a large extent not traded (and thus not valued) in markets. *Social* impact will refer to public goods and bads associated with changes in health, education, gender relations, and relative poverty (and more generally, the size distribution of income and wealth). *Environmental* impact will refer to public goods and bads associated with ecosystem services in all their various forms – as inputs into production processes; as consumption goods that confer well-being directly (e.g., via enjoyment of ambient environmental quality); as consumption goods whose benefits are conferred more indirectly (e.g., drinking water of a given quality); or via non-use values associated with knowledge of a particular environmental resource's existence.

Figure 1 provides a schematic diagram of the pathways whereby the economic, environmental, and social impacts of a particular farming system are transmitted. Farming systems represent a combination of agricultural practices and the extant natural resource base – the fundamental "inputs" of the agricultural production process. Agricultural practices refers to the full gamut of genetic and management activities associated with crop production, livestock production, aquaculture and forestry. The natural resource base comprises the land, water, air, and genetic resources available at a particular location and point in time.

Farming systems produce a range of food, fiber, fodder, forage and tree products that generate contemporaneous economic impacts at different scales (on-farm, local, regional, etc.). Farming systems also generate environmental impacts in the form of physical changes to the natural environment – alterations of physical structures (e.g., alterations of soil structure) and emissions (e.g., pesticide runoff). Both economic and environmental impacts are dynamic in that they feedback onto agricultural practices of

[2] Of course, some agricultural goods and services – both inputs and outputs – will be untraded by some households for whom transactions costs are sufficiently large (de Janvry, Fafchamps, and Sadoulet 1991). But the key point here is that widespread markets for agricultural goods and services provides a ready metric for establishing their value. In contrast, markets seldom exist for many environmental goods and services – developing markets for carbon being a notable exception.

individual producers at a later point in time, as well as on the natural resource base within which those producers operate. Finally, impacts on the economy and on the environment jointly give rise to social impacts – again, both contemporaneously and over time. Although not shown in Figure 1, these social impacts will in many circumstances alter economic and environmental conditions (with some lag), with attendant (feedback) implications for both the natural resource base and agricultural practices.

As noted above, significant headway has been made in developing methods for estimating economic impacts – the upper portion of the flow chart in Figure 1. The goal of this paper is to elaborate on what it will take to achieve comparable progress in pursuing assessment of the environmental impacts of research conducted by the CGIAR in collaboration with its national agricultural research system partners (the lower portion of the chart).

The plan for the rest of the paper is as follows. I begin by reviewing existing evidence on the environmental outcomes associated with agricultural technologies developed by CGIAR research. I then introduce a set of definitions and concepts to establish a common vocabulary for use in the ensuing discussion. Next, I offer a typology of environmental impacts that differentiates between scale over which the impact is felt and also the kind of agricultural system – intensive versus extensive – in which impacts occur. I then discuss separately five elements that need to be addressed in order to adequately and meaningfully conduct the environmental impact assessment of various types of CGIAR research products: biophysical measurement, scale, attribution, valuation, and counterfactual development. Finally, I offer some observations on steps that must be taken to facilitate environmental impact assessment becoming a more standard element of the CGIAR's self-evaluative activities.

Environmental Impacts of CGIAR Research: Review of the Evidence

It is widely believed, even within the CGIAR system, that negative environmental consequences have followed more or less directly from agricultural intensification, and that the Centers have been instrumental in facilitating that intensification process dating back to the Green Revolution. For example, this is how IRRI described the situation in a 2004 document outlining its environmental agenda:

> *There is no denying the adverse environmental consequences of agricultural intensification brought about by widespread adoption of the high-yielding varieties that heralded the Green Revolution of the early days. Nonjudicious use of farm chemicals to attain high yields and in response to heightened disease and pest pressure results in widespread environmental pollution. Heavy demand for water through surface-water and groundwater irrigation affects natural wetlands and water bodies and raises the water table, causing a buildup of salinity and other*

> *soil-related problems. Intensified rice cultivation increases the emission of greenhouse gases such as methane, which is an important component of gases contributing to climate change.* (IRRI, 2004)

It is also widely acknowledged that substantial research emanating from the CGIAR has made positive contributions to reducing, or helping to internalize, negative externalities originating in both intensive and extensive agricultural systems. This is particularly the case for the growing body of CGIAR natural resource management (NRM) research that has been conducted over the past few decades.[3] Indeed, the very definition of NRM research employed by the CGIAR makes clear that it is oriented both toward increasing agricultural productivity <u>and</u> toward improving ecosystem function:

> *[NRM research encompasses} research on land, water, and biodiversity resources management that is focused on producing knowledge that results in technology options, information, and methods or processes that enhance the productivity and stability of ecosystem resources.* (Kelley and Gregersen 2005)

However, as will be discussed below, NRM research has focused almost exclusively on agricultural productivity impacts. Consequently, the current state of knowledge as to environmental impacts of the outputs of the CGIAR's NRM research is very limited.

This section reviews existing empirical evidence on the environmental impacts of technology or knowledge products developed at least in part by one or more CGIAR Centers (Table 1). Remarkably little has been done in the way of seriously tracing the entire chain of outputs, outcomes, and impacts of CGIAR research as it pertains to the natural environment. Moreover, what studies have been conducted tend to focus on positive outcomes – technologies or knowledge-based management regimes that redress some negative environmental externality. Importantly, I am aware of no study that tackles head-on the extent to which countervailing environmental effects reduce the large economic benefits attributable to CGIAR-related productivity increases. Nonetheless, the work reviewed below offers insights that are of potential value in formulating an approach to considering those issues.

Deforestation and policy

Raitzer (2008) describes the entire impact pathway of CIFOR's work on the political economy of Indonesia's pulp and paper sector. Research *outputs* from that work chronicled inefficiencies in fiber sourcing practices and in the administration of very large subsidies on large forest products firms. Those outputs produced *outcomes* that

[3] Note that another positive externality often ascribed to CGIAR research relates to reducing conversion of forested and other (low-potential) uncultivated lands to agricultural uses. This is the topic of a companion research study currently underway, and so will not be considered in the current paper.

included changes in Indonesian government's implementation of those policies due to pressure of various external watchdog groups like Friends of the Earth, World Wildlife Fund, and other NGOs who were influenced by CIFOR's research. Having established (as well as could be expected) a clear attribution of these outcomes to CIFOR, the study developed quantitative measures of the *ex post impacts* on the basis of expedited policy change – i.e., natural forests being cleared more slowly than would have otherwise been the case.

This study is one of the most successfully executed policy research *ex post* impact assessment (epIA) studies to have been conducted within the CGIAR. It is notable for the clarity with which it traces out the impact pathway of CIFOR's work. Also of note is its use of benefit transfer methods (i.e., using existing empirical estimates of the value of watershed service, carbon sequestration benefits, and avoided biodiversity losses drawing on) as a means of valuing environmental benefits. Neither measurement or modeling of biophysical effects were conducted, as the activity being evaluated was policy analysis rather than technology products.

Pesticide use

Two research programs that computed benefits of research in an epIA framework focused on the human health and ecological impacts of research on pesticide use. These two efforts were conducted during the 1990s – by IRRI in the Philippines (Pingali and Roger 1995) and by CIP in Ecuador and Peru (Crissman, Antle, and Capalbo 1998). Both found very large health benefits from their respective Centers' research and subsequent extension efforts to promote reductions in farmer exposure to toxic pesticides; and both found, somewhat unexpectedly, that off-site negative environmental consequences of excessive pesticide use were in fact minimal.

Interestingly, the two Centers' research programs took quite different approaches to valuing the outcomes of their respective research. The IRRI work used econometric methods to compute health benefits associated with avoided medical costs and increased farmer productivity due to averted sickness. Follow-on policy research conducted a decade later by Templeton and Jamora (2007) estimated the private health savings attributable to that research – via regulation of highly toxic insecticides in rice production, labeling requirements, and training of rural health officers – to have a net present value of $117 million and an internal rate of return of 65%.

The CIP team, on the other hand, made the explicit decision not to convert research benefits into monetary terms in pursuit of a conventional benefit-cost analysis. Instead, they opted to present the results in the form of a "tradeoff analysis," wherein stakeholders were presented with quantitative indicators of economic performance (annual or present value of net returns under different agricultural production scenarios), environmental outcomes (related to soil quality, soil erosion, chemical leaching, etc.), and human health effects, as well as the distribution of these various outcomes across different groups (Antle, et al. 2003). Using this approach, stakeholders and decision-makers essentially impose their own values on each of these various impacts.

Another interesting difference in these two projects had to do with findings regarding productivity impacts of reduced pesticide use. The Philippines work found no significant productivity losses when pesticide use on rice was lowered, whereas the work in Ecuador found that pesticides did in fact contribute to higher potato yields. Thus in the Ecuador case there appeared to be an implicit productivity-environmental quality tradeoff, while in the Philippines pesticide application was excessive in the sense that no such tradeoff appeared to be in play. This highlights limitations on generalizing environmental impacts beyond the geographic frame of analysis and/or across different crops.

Agriculture and ecosystem services

ICRAF was a central participant in the Trans-Vic research project, a multi-year, multi-institution activity that investigated agriculture-environment interactions in two watersheds of the Lake Victoria basin. The study is noteworthy for its use of GIS-based models to assess sediment yields and hydrologic flows. These were combined with spatial data on agricultural production – gathered via remote sensing – to track land use changes and their subsequent impact on provisioning and regulating environmental services (Swallow, et al. 2009). The research stopped well short of quantifying the impacts of specific technologies or knowledge products on environmental outcomes, however, opting instead to focus on associations between soil losses and the type of agriculture production system (high-value versus low-value). In addition, the research was primarily oriented to assessing on-site impacts. Negative off-site impacts of sedimentation on environmental quality of Lake Victoria were not addressed – a distinct shortcoming given that these damages to the aquatic ecosystem had important effects on the biological productivity of that ecosystem and on the people whose livelihoods are tied to it.

CIMMYT's research on the environmental impacts of zero-tillage (ZT) wheat systems in the Indo-Gangetic plains of India and Pakistan represents another example of research seeking to understand the interactions between agriculture and a host of ecosystem services. Reduced tillage intensity in wheat production has proven to be cost-reducing, which explains its spread in the region. While this work mainly focused on agricultural profitability impacts, the research has also documented some environmental benefits coming out of the wheat side of the system – reduced diesel consumption (and associated reduction in CO_2 emissions) and modest water savings due to improved irrigation efficiency (Erenstein 2009). However, the fact that tillage of the rice side of the system generally has not been reduced appears to have greatly limited the potential environmental benefits (in the form of improved soil quality characteristics) from being realized.

ICRISAT sponsored a wide-ranging study of methods for assessing economic and environmental impacts of NRM research, culminating in an edited volume (Shiferaw, Freeman and Swinton 2005). Two case studies in that book feature bioeconomic models of soil conservation technologies at the farm and village levels in Ethiopia (Shiferaw and Holden 2005; Holden and Lofgren 2005). These studies did not focus on measuring *ex post* environmental impacts, nor did they examine technologies that were developed by the ICRISAT. Rather they are oriented toward simulating the (on-site) effects of alternative fertilizer subsidy and land tax policies on input use and subsequent environmental outcomes. Note, however, that in these sorts of bioeconomic models one could use a set of new NRM practices as the initiating "shock" that creates the environmental impacts to be traced out.

CIAT has taken part in a multi-faceted assessment of agriculture-environment linkages as part of the on-going Ecosystem Services and Poverty Alleviation program. To date, the primary output of that program is a "situation analysis" of environmental services, some of which are directly related to agricultural production activities (ESPA-AA 2008). This *ex ante* analysis, conducted via consultation with various stakeholders in the region (researchers, civil society organizations, government and nongovernmental organizations), and appears to have been mainly oriented toward documenting baseline conditions in the region and re-creating "conventional wisdom" regarding the vulnerability of the poorest rural dwellers to changes in availability of environmental services.

Other technologies

Other research conducted by CGIAR Centers has touched more obliquely on issues related to environmental impact of their outputs. WorldFish's *ex post* analysis of their integrated aquaculture-agriculture (IAA) technologies found that several indicators of sustainability – on-farm species biodiversity, recycling of biological outputs and by products, and overall biomass yield – were greater on farms that adopted IAA (Dey, et al. 2007). That same research additionally found that substitution of fishpond sediments for inorganic nitrogenous fertilizers reduced fertilizer consumption by 50 percent. No effort was made to value these positive outcomes in monetary terms, however.

IWMI has devoted considerable effort to studying irrigation management transfer (IMT) programs. This research responded to widespread evidence of poor performance by publicly owned irrigation schemes and the attendant belief that devolution of oversight of these systems to farmer organizations would improve their management and make irrigated agriculture more productive and more sustainable. But while IWMI's self-assessment indicates that its efforts have made a substantial contribution to knowledge about design and implementation of these programs, it falls well short of documenting *ex post* whether or not the purported environmental benefits of IMT have actually come to be (Giordano 2006).[4]

Greenhouse gas emissions

The global alternatives to slash-and-burn program (ASB) motivated research that investigated the net greenhouse gas emissions and profitability of a range of land-use alternatives to tropical forest conversion. Palm, et al. (2004) used data generated at three of six "sentinel sites" monitored by the ASB to assess the tradeoffs between global environmental and private economic aspects of land-use systems in the humid tropics. That *ex post* analysis indicates that many tree based systems had moderate levels of carbon storage, and that on balance this reduced net global warming potential compared to annual cropping and pasture systems. However, it also found that widespread adoption of tree-based systems is often limited by the substantial start-up costs, credit limitations, and number of years to positive cash flow, in addition to the higher labor requirements. While not an assessment of impacts per se, this work is nonetheless relevant here as it represents one of the few large-scale efforts to measure (in physical terms) tradeoffs between agriculture and the environment.

Environmental Indicators and Monitoring Systems

Two other Center-related research activities meriting mention here are more oriented to monitoring environmental outcomes via development of indicators of environmental outcomes. CIP researchers have studied the environmental and human health risks associated with pesticide use through the development of an environmental impact quotient (EIQ). The EIQ summarizes the total hazard posed by all pesticides applied over different potato cultivars in a particular location (Pradel, et al. 2009). The EIQ is a summary measure of overall (biophysical) environmental impact – essentially a weighted average that factors in relative toxicities, half-lives, leaching potential, and surface loss potential of the various chemicals applied to potato fields. The basis upon

[4] The authors of this study provide three reasons for focusing on research outcomes rather than research impacts: (a) long lags between research investments and measurable research outcomes; (b) difficulties in attributing policy changes to IWMI research; and (c) lack of baseline data.

which the specific weights were imposed is unclear. The study's principle finding, that the correlation between EIQ and production outcomes is low, is not really an impact assessment so much as an indicator that use of less environmentally harmful pesticides would not compromise agricultural output.

IFPRI developed a Policy Relevant Monitoring Systems (PRMS) in Costa Rica to facilitate management of natural resources in settings characterized by multiple resource users with conflicting interests whose activities impose negative environmental externalities on others (Hazell, et al. 2001). The goals of PRMS are quite ambitious. They include (a) providing an apparatus for deciding which resource problems to focus on; (b) generating early warning information on emerging problems and conflicts (including their causes and possible means of identifying corrective actions); and (c) establishing an institutional framework for promoting agreement on solutions (i.e., for internalizing externalities).

The centerpiece of the PRMS is a "payoff matrix" that combines both direct impacts and externalities created by different stakeholder groups on each other. These stakeholders included farmers, foresters, fishermen, dairymen, and a large electricity generation facility. By quantifying the net benefits and costs to specific stakeholders (based on the testimony of knowledgeable local experts), this payoff matrix circumscribes potential Coasian solutions to environmental conflicts. That is, it quantifies the transfers needed to fully compensate "losers" for the costs of negative externalities imposed on them by "winners." Thus, the study offers an interesting mix of environmental impact assessment and an approach to developing (local) institutions for internalizing environmental externalities.

Two aspects of this work are particularly relevant here. First, this approach is clearly one that needs to be tailored to specific, discrete spatial units of observation. That is, generalizing its findings beyond a fairly localized geographic scale (e.g., a specific watershed) is probably not feasible in most cases. Second, this study computes estimates of off-site costs that the actions of specific stakeholders impose on each other. To do so, projected land use changes under simulated scenarios of resource use by various stakeholders are combined with assumed relationships between resource use and biophysical outcomes. It would seem that a similar analytical approach could be implemented using biophysical models that are more closely calibrated to *ex post* observations within a particular study area.

Missed Opportunities

In three general areas it would appear that very large environmental benefits from CGIAR research remain unquantified: IITA's work on biological control of insects and water hyacinth; the growing body of NRM research that has been conducted within the System; and the value of land savings due to increased productivity in areas in which improved varieties have been widely adopted.

Substantial documentation exists regarding the positive production impacts of IITA's biological control program. One of the best known cases is the control of the cassava mealybug in 20 countries of Sub-Saharan Africa (Zeddies et al., 2000). The biological control provided by an introduced wasp was so effective that the cassava mealybug is now largely contained. Even when using the most conservative assumptions, the return on this research investment has been extremely high (net present value estimated at US $9 billion). Subsequent *ex post* Impact Assessment studies on biological control of cassava green mite, mango mealybug, and water hyacinth computed similarly large returns to the investments in these programs as well (Coulibaly, et al. 2004; De Groote, et al. 2003).

These estimates of net benefits from biological control research only account for productivity impacts, however. Importantly, there does not exist any sort of accounting for ecological benefits of biological control research against the counterfactual of increased use of chemical pest control (Alene, et al., no date). Yet if the CIP and IRRI research on pesticide use is any indication, potential benefits in terms of both environmental quality and human health are likely large as well. This seems like a very promising candidate for future impact assessment research.

NRM research represents a second general area of CGIAR activity whose positive environmental impacts remain uninvestigated to date. For example, none of the research reported in the recent volume of case studies of NRM research by CGIAR Centers did more than chronicle limited environmental benefits (Waibel and Zilberman 2007). Nonetheless, there is a general presumption within the CGIAR that the value of these benefits is substantial.[5] Hence, this too would appear to be a fruitful avenue for future impact assessment research.

Finally, an effort was initiated by the CGIAR in the late 1990s to explore the extent of land savings attributable to the large productivity increases that followed widespread dissemination of improved varieties (Nelson and Maredia 1999; Maredia and Pingali 2001; Evenson and Rosegrant 2003; Nelson and Maredia 2007). That work estimated that absent CGIAR crop genetic improvement activities, an additional 200 million hectares of land in developing countries would have needed to be cultivated

[5] For example, the Science Council's review of those case studies contended that the environmental benefits of NRM research "probably outstrip benefits from crop genetic improvement research, but that is subject to future research" (Science Council 2006, pg. 1).

during the 1990s to produce the same amount of cereal output. This aggregate figure did not explore geographic differences in land-saving impacts. Neither did it attempt to ascertain (or value) the positive environmental impacts associated with a greater fraction of global cereal production originating in intensive agricultural production systems as opposed to extensive systems on ecologically more fragile lands. Pursuing these lines of inquiry in more detail is the subject of a current SPIA research initiative.

Summary

This review has revealed a very thin record of research assessing the environmental impacts of technologies and knowledge products generated by CGIAR research. Some progress has been noted on quantifying *ex post* impacts of pesticide use; but these have focused primarily on human health impacts. A couple of pieces of *ex post* policy-oriented research have quantified environmental impact of CGIAR analyses of timber harvesting policies and pesticide reduction policies. Some work has taken steps toward documenting outcomes related to improvements in nutrient management and soil and water quality associated with CGIAR research activities. But notably, there has been no work done to date tracing the entire impact assessment pathway from research investment through measurement of off-site biophysical effects on ecosystem services and on to the ultimate economic impacts on agents located on receiving sites. In short, there are no extant studies of CGIAR research outputs that can be regarded as a "template" for guiding future *ex post* environmental impact assessment.

The studies that have been reviewed here <u>do</u> offer examples of analytical tools that will need to be brought to bear in order satisfactorily pursue environmental impact assessment. Examples include

- the bioeconomic modeling work highlighted in the Shiferaw, Freeman and Swinton, et al. (2005) volume;
- discussions and implementation of environmental indicators found in that same volume, as well as in CIP's work on environmental impact quotients;
- use of GIS-based spatial modeling and remote sensing in the Trans-Vic project;
- the use of "sentinel sites" for long-term monitoring of environmental impacts developed under the aegis of the Alternatives to Slash and Burn program; and
- attention to quantifying tradeoffs among various stakeholders whose actions impose negative externalities on one another in IFPRI's Costa Rica work (Hazell, et al. 2001).

In sum, the dearth of efforts to quantify the impacts of CGIAR research on the environment is striking.[6] There are several possible explanations for this. First, measuring environmental services in a consistent manner over a period of time is difficult. It requires sampling a large number of variables that one would want to hold constant in any meaningful statistical analysis, beginning at a very early stage in the adoption/diffusion process. And the larger the number of variables exerting influence on a given environmental outcome, the more difficult it is to establish links between that outcome and a particular agricultural practice (and thence attribution to research).

Second, valuing those services poses a distinct challenge as well. With the exception of CIFOR's work on deforestation and the work of CIP and IRRI on pesticide use, this appears to have been an insurmountable obstacle in most CGIAR research in this area. Particularly noticeable is the absence of non-market valuation – using either revealed preference or stated preference techniques – of environmental services affected by CGIAR technology, management or knowledge products.

There are no doubt other, more prosaic reasons for the paucity of effort devoted to measuring the environmental impacts of the products of CGIAR research. That line of inquiry typically requires substantial inter-disciplinary collaboration, the organization and administration of which can be challenging. The substantial field research required to pursue environmental impact assessment analysis is costly as well, particularly for research efforts scaled at the regional, national or global levels. Such field work requires sampling at multiple points in time, extending the duration (and cost) of the activity. Finally, particularly in the case of assessing negative environmental impacts of CGIAR research, there is a fundamental matter of institutional appetite for pursuing this sort of activity. Simply put, Centers have little incentive to pursue research that has some positive probability of putting them in a bad light.

Toward an Environmental Impact Assessment Framework

For decades there has been substantial interest among donors, policy makers, and agricultural scientists in understanding the environmental impacts of CGIAR research outputs (and agricultural research in general). Yet, as the review of past work in the previous section indicated, surprisingly little headway has been made in achieving that

[6] In order to begin bridging this gap, in 2009 SPIA recently commissioned a set of six *ex-post* impact assessment case studies to quantify the environmental impacts of specific Center research activities. Those case studies encompass a variety of topics, including zero tillage in the Indo-Gangetic Plains, supplemental irrigation, water control in mixed rice and shrimp areas, rubber agroforestry, potato biodiversity preservation, and the land-saving impacts of improved bean cultivation. These studies are scheduled to be finalized by the end of 2010.

goal. Clearly, some intervening factors have severely constrained investigators' ability to pursue this line of inquiry.

In the next three sections, I lay out a set of issues that need to be addressed in order to satisfactorily pursue meaningful assessment of the environmental impacts of CGIAR research. I first introduce a set of definitions and concepts to establish a common vocabulary for use in the discussion. Next, I offer a typology of environmental impacts that differentiates between scale over which the impact is felt and also the kind of agricultural system – intensive versus extensive – in which impacts occur. This discussion describes the primary impacts of CGIAR outputs on land, water, climate, and genetic resources. I then discuss separately five elements that need to be addressed in order to adequately and meaningfully conduct the environmental impact assessment of various types of CGIAR research products. Some of these challenges are common to all *ex post* impact assessment – i.e., those related to attribution, scale of analysis, and establishment of appropriate counterfactuals. Others are more specific to environmental impact assessment *per se*: the measurement and modeling of changes in ecosystem services resulting from specific interventions, and the subsequent valuation and integration of these biophysical outcomes into behavioral (economic) models.

Environmental Impact Assessment: Some Definitions and Concepts

Our concern here is with the impacts of agricultural activities – in particular, activities that are affected by the CGIAR's technology, management and knowledge products – on the natural resource base. Those products alter, to varying degrees, the air, water, land and genetic resources that comprise the natural resource base.

Many of these alterations to the natural resource base will be felt first, and foremost, by the farmers whose actions directly caused them. For example, on-site fertility losses due to nutrient depletion or soil erosion directly affect the productivity and profitability of the farm on which it occurs. For purposes of this paper, I will refer to these types of on-site environmental impacts as *production effects*. In general, our focus here is <u>not</u> on how to measure these production effects, since they are effectively internal to the farming operations that created them and would be reflected in "standard" *ex post* economic impact assessments.[7]

Rather, the center of attention here will be on the impacts of changes to the natural environment that are external to the individuals directly responsible for those changes.

[7] One exception to this would be the case of inter-generational impacts wherein current on-site alterations affect the productivity potential of that site at a future date. Such inter-generational impacts are at the heart of debates over the sustainability of agricultural production systems and will be more fully discussed below.

For purposes of this paper, I will use the term *off-site environmental impacts* to refer to alterations to the natural resource base that affect other (off-site) users of those natural resources. Correspondingly, the term *environmental impact assessment* (EIA) will be used here to encompass the suite of activities required to measure changes in off-site stocks and/or flows of environmental services accompanying adoption of an agricultural innovation and then to assign monetary values to those changes. Defined this way, EIA accounts for the impacts on the natural resource base not already accounted for by standard *ex post* economic impact assessment.

This dichotomy between off-site environmental impacts and on-site production effects is depicted in Figure 2, which provides a schematic diagram of the pathway from research and extension efforts to ultimate economic and environmental impact assessment. Research and extension inputs create innovations ("outputs") in the form of technology, management, or knowledge products. Adoption of these products gives rise to both environmental impacts and production effects as described above.

Standard *ex post* impact assessment studies – termed "economic impact assessment" in the upper portion of the figure – focus on evaluating production effects within a conventional cost-benefit analysis framework. Doing so requires attention to issues of attribution, scale of analysis, and establishment of appropriate counterfactuals; these can be difficult tasks, but ones whose complexities have been well laid out elsewhere (Walker et al. 2008).

Pursuit of environmental impact assessment (the lower portion of the diagram) also requires substantial attention to attribution, scaling, and counterfactual establishment. In addition, two key factors complicate the environmental impact assessment process *vis-à-vis* economic impact assessment. First, measurement and/or modeling of the physical environmental outcomes resulting from agricultural innovations will often be more difficult. A very large number of biophysical interactions condition the functioning of ecosystems at even a small scale (e.g., farm-level); and of course, the complexities of ecological relationships intensify as the scale of analysis broadens to the watershed and beyond. Second, the valuation of environmental goods and services is a distinct challenge because so many of those goods and services are generally not traded in markets.

Before turning to a more detailed discussion of issues related to assessing environmental impacts of specific types of CGIAR research products, three other general aspects of the assessment process merit some mention here. First, note that the depiction in Figure 2 of the pathway to environmental impact assessment explicitly includes extension (along with research) as an initiating input. Complementary investments in extension often play a prominent role in facilitating the adoption of some CGIAR products, especially NRM technologies.[8] This aspect of NRM technologies, coupled with often-weak institutional capacity in locations where they are adopted, often means that the projects' outreach components are critical to their having significant positive impacts (Renkow and Byerlee, 2010). As will be further discussed below, the importance of extension and complementary institutions in the diffusion process can complicate both the attribution of environmental effects to specific Centers and the design of appropriate counterfactuals against which those outcomes are evaluated.

Second, environmental outcomes from agricultural practices may be positive or negative. Agro-chemical runoff that worsens water quality for downstream water users, pump irrigation that depletes the groundwater available to others, and soil erosion leading to siltation of nearby waterways are all examples of a negative environmental impacts attributable to agricultural practices. NRM practices that reduce soil erosion and IPM strategies that reduce the use of toxic chemicals are example of a positive environmental effects. Positive environmental impacts resulting from a new technology or practice will in many cases be intended consequences of the research that generated it; hence it is reasonable to expect research managers to build in capability to establish environmental benchmarks as part of the research design. In contrast, negative environmental effects generally are unintended outcomes. That negative outcomes tend to be unanticipated complicates efforts to assess their environmental impacts *ex post* since critical baseline data are unlikely to have been collected.

Third, environmental impacts will be felt by a variety of different actors via different pathways. These include:

- Direct consumers of an environmental resource – e.g., those who gain well-being from ambient environmental quality *per se* (both residents and visitors to an area);

- Indirect consumers of an environmental resource – e.g., individuals drinking from water sources whose quality is compromised by agricultural run-off;

- Non-agricultural producers affected by alterations to a natural resources that is an input into their production process – e.g., fisherman whose livelihoods are affected by changes in waterways due to erosion-caused siltation;

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[8] Other institutions, particularly those related to the establishment and enforcement of property rights, are important complementary inputs into the process as well. Extension is singled out here because a large fraction of CGIAR projects, particularly knowledge- and management-intensive NRM projects, involve direct links to extension through collaboration with NARS partners.

- Agricultural producers – e.g., farmers whose livestock are negatively affected by pesticide residues or whose cost of irrigation are increased due to groundwater depletion or siltation of canals;

- Non-local individuals for whom (non-use) option and existence values of a particular environmental resource are affected by changes to the resource – e.g., conversion of forest land resulting from agricultural extensification.

The multiplicity of types of individuals potentially impacted by agricultural innovations complicates the process of environmental impact assessment as well. Both physical measurement and valuation of the environmental effects of agricultural innovations will, in many circumstances, require different approaches depending on which receiving group is being considered. In addition, when the incidence of costs and benefits differs across stakeholders, assessing distributional impacts becomes a significant challenge.

Environmental Impacts by Source and Type of Agriculture

Table 2 presents a typology of environmental impacts of agriculture and the scale(s) over which those impacts are generally felt – on-site (at the plot or farm level); locally (at the village or watershed level); or globally. The typology distinguishes between intensive and extensive agricultural systems – an important distinction given the substantial differences in the kinds of environmental problems associated with these two types of agricultural systems .

Intensive systems are characterized by substantial monocropping, significant use of non-labor inputs, and, in many important production zones, irrigation. In general these are irrigated or high rainfall areas that were most profoundly affected by seed-fertilizer technologies that have traditionally been the mainstay of the CGIAR's commodity centers (and whose diffusion accelerated the intensification process in many of these areas). Environmental problems associated with intensive agricultural systems reflect the high demands that may be placed upon the natural resource base by the intensification process – soil degradation due to continuous cropping, salinity problems and waterlogging associated with excessive (and improperly administered) irrigation, negative side effects resulting from use of chemical inputs, and loss of *in situ* biodiversity.

Extensive agricultural systems, on the other hand, tend to be found on lands that are of lower agronomic potential due to a variety of abiotic stresses – low or highly variable rainfall, fragile soils, limited fertility, and the like. Increasing production in extensive systems often entails bringing ever greater amounts of land under cultivation, although in some situations it involves conserving resources or using them more efficiently. Many CGIAR investments in NRM are oriented around conserving on-site fertility or moisture resources (e.g., zero tillage or certain kinds of agro-forestry projects). In addition, innovations to crop management practices tend to assume a greater role in efforts to enhance agricultural productivity in areas of lower agronomic potential. The primary off-site environmental problems associated with extensive systems relate to the interaction of agriculture with other land uses – especially conversion of forested land and rangeland to agricultural uses, with the attendant implications for global climate change and biodiversity. Other environmental problems in extensive system relate to cultivation of ecological fragility of lands characterized by poor soils or steep slopes (World Bank 2003).

Land

Negative environmental impacts of agricultural activities on land resources reflect some combination of excessive extractive demands and improper management of those extractive activities. Positive impacts, on the other hand, typically take the form of management regimes meant to counter those excesses (via reversal or prevention). In intensive systems, major environmental impacts include salinization and waterlogging due to poor irrigation management; fertility losses due to improper nutrient management; and loss of organic matter due to soil erosion.

A significant share of problems related to irrigation and nutrient depletion take the form of on-site production effects (Pingali, et al. 1997).[9] These might potentially give rise to longer-term, intergenerational impacts to the extent that damages persist over a considerable amount of time and/or the positive impacts of reversing them benefits future users of those land resources. Note, however, that even these long-term impacts are readily accommodated by standard impact assessment, although doing so will in many circumstances require projection of trends in production and prices.

Soil erosion (and attendant loss of organic matter) will have both on-site and off-site impacts. Important off-site impacts include siltation of irrigation infrastructure, sediment buildup in lakes and reservoirs, and increased flooding risk (Cruz et al. 1998). These pose significant measurement challenges for analysts because erosion rates vary widely depending on the soil, topographic, and hydrologic characteristics of affected lands (both source and receiving sites).

[9] Note, however, that off-site effects of irrigation can be important too –the disappearance of the Aral Sea and salinization of downstream lands in Central Asia being perhaps the most widely known example.

An abundance of models have been developed and used to track soil and hydrologic dynamics. These tend to be applied on a short-time frame basis, though, and are best suited to plot- or farm-level analyses (e.g., Shepherd and Soule 1998).[10] In light of substantial inter-farm heterogeneity, scaling up the results of model-based simulations to a meso-level places a premium on careful sampling. In addition, some combination of spatial modeling and field checks is essential for validating predicted (simulated) biophysical outcomes. GIS and digital elevation models are possible means of dealing with the spatial modeling issue; field inspections combined techniques such as reflectance spectroscopy, remote sensing, and use of satellite imagery are possible means of validating on-the-ground effects (Roy, et al. 2003).

The primary land-based externalities associated with extensive agriculture center on conversion of non-agricultural lands to agricultural uses. A central question regarding crop genetic improvements associated with the CGIAR revolves around whether yield increases associated with improved varieties cause a reduction in such land conversion due to less land being needed to produce the same amount of food (the so-called "Borlaug hypothesis") or, in contrast, lead to expansion of area under cultivation as an outgrowth of farmers taking advantage of the greater productivity of higher yielding varieties (Angelsen and Kaimowitz 2001). Addressing this question is the emphasis of current research being conducted by SPIA.

Water

Groundwater depletion is a classic example of a negative externality attributable to over-exploitation of a common property resource. In irrigated agricultural systems in particular, the substantial yield response of modern varieties to timely water application has contributed to over-pumping of groundwater and subsequent lowering of water tables.

Several salient points pertain to assessing the impacts of CGIAR technologies' contributions to groundwater depletion.[11] First, while there can be no doubt that modern varieties are a central part of the over-pumping story, so too are pricing policies for electricity or for irrigation infrastructure that have held the cost of accessing groundwater resources well below market rates. Thus, a substantial attribution issue exists regarding how to allocate the "fault" for over-pumping.

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[10] Note that even at the plot level, measuring soil loss is not straightforward. For example, the Universal Soil Loss Equation requires accurate data on six parameter values that can pose distinct measurement challenges (Stocking 1996).

[11] Note that these points also apply to the potential negative impacts associated with surface irrigation.

Second, ascertaining the social costs of over-exploitation of groundwater resources will generally be aided by the fact that in most cases market measures are readily available for valuing water. Where markets for water exist, the price of water will greatly simplify valuation challenges. And even where water markets are thin or non-existent, information on the cost of pumps and pumping will provide a useful valuation benchmark.

Third, many of the benefits of CGIAR efforts to promote water conservation – e.g., through zero tillage or other NRM-based crop management techniques designed to improve water use efficiency – will show up as on-site production effects. Analysts of these sorts of technologies need to carefully disentangle these on-site effects from off-site environmental impacts associated with reduced water withdrawals.

Finally, to the extent that the on-site production effects of groundwater depletion persist through time, intergenerational impacts may well arise. On-site production effects that felt by future resource users do represent an externality, and thus pose many of the same analytical challenges required to evaluate (spatial) external effects. Consideration of inter-generational impacts additionally requires attention to the bequest motives and rates of time preference of current resource users.

With regard to extensive rainfed agricultural systems, water conservation technologies will in some locations produce off-site environmental impacts. Positive impacts include greater recharge of downstream aquifers and enhanced water retention in upstream areas (World Bank 2007a). Negative impacts may include reductions in downstream water availability; for example, some recent evidence points toward forestry and agroforestry projects reducing downstream water availability due to increases in canopy interception and evapotranspiration (Jackson, et al. 2007). The bulk of the impacts of such technologies will likely take the form of on-site production effects, however.

Of potentially more interest are off-site environmental impacts associated with bringing croplands under irrigation that had previously been farmed using relatively low-input systems. Conversion of arid and semi-arid areas environments may lead to significant depletion of surface or groundwater resources if and when plant consumptive uses exceed recharge rates (Howell 2001). Attention to this off-site environmental impact is thus an important element in assessing the true net benefits of conversion of rainfed systems to irrigated systems.

Agro-chemicals

The literature on negative impacts of pesticides (or the positive impacts of reduced pesticide use) extends well beyond the examples cited in the earlier review of CGIAR-related EIA studies. Human health impacts occupy a central role in much of that literature, but so too do impacts on flora and non-human fauna (Mullen, et al. 1997; Cuyno, et al. 2001; Brethour and Weersink 2001). This research generally computes environmental impact quotients (EIQs) to proxy for aggregate environmental risk associated with pesticide use, and combines these with stated preference measures of willingness to pay for lessening those environmental risks.

One issue related to use of indicators such as the EIQ is that some of the component elements of the indicators – especially relative leaching potential and surface loss potential – would appear to be highly variable at higher spatial scales. Thus, model-based fate and transport studies of potential pollutants are a necessary complement to analysis of on-farm pesticide (Ducrot, Hutson, and Wagenet 1998). Note, however, that the predictive efficiency of soil and hydrology models declines rapidly at scales beyond the farm level (Roy, et al. 2003). Recognition of this probably explains why the authors of the CIP and IRRI studies of pesticide impacts were careful to caution against generalizing their finding of limited off-site environmental impacts.

Finally, it bears noting that the work conducted by CIP and IRRI in the 1990s on pesticide use and its impacts was preceded by earlier Center-endorsed recommendations that involved significant pesticide use. That the later research led to revision of earlier recommendations is not terribly unusual. And other factors, including large subsidies on chemical pesticides, overly aggressive promotion of chemical use, and inattention to health and safety guidelines, were also important contributors to overuse of pesticides and their attendant negative environmental consequences (Templeton and Jamora 2007). Nonetheless, it does raise an important attribution issue – namely, the need to account for the Centers' roles in the overuse of pesticides that the later research helped to ameliorate.

Livestock

Livestock are a major contributor to global greenhouse gases (especially methane), and therefore to climate change. Extensive livestock grazing activities can also have large scale negative environmental impacts, such as their contribution to desertification in West Africa. Note, however, that the "new rangeland ecology" literature provides evidence that extensive livestock systems can have positive environmental impacts if managed appropriately and – crucially – so long as land is sufficiently available to allow pastoralists to practice transhumance (Behnke, Scoones, and Kerven 1993).

Particularly in highly urbanized countries of Asia and Latin America, proliferation of intensive livestock systems has accompanied rising demand for livestock, fish and poultry products (World Bank 2007a). Some CGIAR efforts in breeding and management, particularly by ILRI and Worldfish, have facilitated this process. The social benefits of these activities – in the form of greater protein supply and more diversified diets – are to some extent countered by associated negative off-site environmental consequences of intensification. These include pollution due to waste runoff and greater disease transmission among animals and from animals to humans (e.g., avian flu).

Biodiversity

Assessment of the impacts of the CGIAR's crop genetic improvement research on biodiversity and biodiversity loss requires analysis at a global scale. The two key elements of the problem include the loss of biodiversity on lands converted from non-agricultural to agricultural uses, and the loss of genetic diversity of specific crop varieties due to decreases in the number of different varieties grown (as improved varieties continue to supplant landraces).

Forest conversion due to expansion of the agricultural frontier is a primary source of biodiversity loss – particularly in the Amazon, Southeast Asia, and West Africa, but in other hotspots around the world as well (World Bank 2007b). Assessing biodiversity loss at a global scale poses severe measurement and valuation challenges. Moreover, development of appropriate counterfactuals hinges on projections of highly uncertain future outcomes regarding the uses which might have been made of lost genetic resources.

Interest in the loss of crop genetic diversity associated with widespread adoption of modern varieties has existed for some time, but continues to be an area of inquiry for which there exist as many questions as answers when it comes to estimating the benefits of being able to address unforeseen future problems (Smale 2006; Koo, Pardey and Wright 2004). Here the development of an appropriate counterfactual represents a particularly difficult challenge, hinging as it does on projections of future yield losses and/or disease outbreaks whose reversal would be compromised by inadequate stocks of genetic resources (either *in situ* or *ex situ*). A research effort recently commissioned by SPIA seeks to analyze the implications of widespread diffusion of the CGIAR's crop genetic improvement work for both *in situ* and *ex situ* conservation of genetic resources; this will provide a more comprehensive assessment of these issues in the near future.

Climate Change

As with biodiversity, assessing the contribution of CGIAR research to climate change requires analysis at a global scale. Impacts of agriculture on climate change tend to be associated with specific practices (e.g., use of mechanical technologies that burn fossil fuels); with release of carbon into the atmosphere due to disturbance of soil carbon stocks; and with conversion of land (particularly forested land) to agricultural uses, with attendant declines in carbon sequestration.

Measurement of the physical contribution of a specific practices to emissions of greenhouse gases (or reduction thereof) is generally fairly clear cut. So long as the spatial extent and distribution of particular activities (e.g., use of some fossil fuel burning technology) is fairly well understood, scaling up of plot- or farm-level activities to a broader spatial units should be straightforward. In contrast, the other two agriculture-related sources of impact on climate change – soil carbon losses and land conversion – exhibit considerably greater spatial heterogeneity, and thus pose much more severe scaling challenges. Indeed the geographic variability of different land uses with respect to both soil carbon and forest loss is the core reason why the ASB program used "sentinel sites" in attempting to assess agriculture's contribution to global warming (SPIA 2006).

Valuation represents a distinct challenge in attempting to quantify the impacts of agricultural innovations and attendant land use changes on global climate change. One frequently hears reference to using the price of carbon in fledgling carbon markets, such as the Chicago Climate Exchange (www.chicagoclimatex.com), as a means of valuing net additions to (or subtractions from) atmospheric carbon resulting from agriculture . However, as presently constituted these markets are very thin. Moreover, given that trade on these markets is largely driven by government policies (e.g. U.S. wetland regulations), it is by no means clear that observed carbon prices are, as of yet, a particularly good indicator of aggregate social demand and supply of carbon. This may well change, however, as carbon trading becomes more ubiquitous worldwide.

Finally, it is worth noting that the counterfactual for assessing the impact of agricultural innovations on climate change might include significantly increased poverty and malnourishment (due to higher food prices that would have pertained absent those innovations). In other words, the relevant comparisons may well include two undesirable outcomes. And of course, an additional complication lies in the fact that many of the most critical impacts associated with climate change relate to future outcomes that are highly uncertain.

Environmental Impact Assessment: Key Elements

As depicted in the lower portion of Figure 2, assessing the environmental impact of CGIAR activities requires attention to five key elements: biophysical measurement, attribution, scaling, valuation, and counterfactual development. This section describes issues associated with each of these elements of the assessment process and approaches to dealing with them, then offers two examples of how those issues might be addressed in the context of specific environmental impact assessment case studies.

Biophysical measurement

A substantial body of work reflects on how best to quantify changes to stocks of agro-ecosystem assets and associated flows of ecosystem services. For example, the Shiferaw, Freeman, and Swinton (2005) volume on NRM impact assessment cited earlier contains separate chapters summarizing the uses of biophysical indicators and simulation models to analyze changes in on-site soil quality, water quantity and quality, and ecosystem services (including *in situ* biodiversity and land cover) attributable to agricultural production activities (Pathak, et al. 2005; Sahrawat, et al. 2005; Wani et al. 2005).

Wani, et al. (2005) provide a set of biophysical indicators that are commonly used to track or predict changes in ecological conditions. These are reproduced in Table 3. While by no means a comprehensive listing of all indicators employed by natural scientists, the indicators listed in Table 3 provide a feel for the multiplicity of potential environmental impacts, as well as the substantial amount – and variety – of data required to measure them. Agricultural scientists and ecologists commonly use these indicators to inform judgments about environmental impacts that follow from changes in production practices or use of a new technology. Some studies focus on one or a few key indicators such as soil loss (National Research Council 1993), nitrogen availability (Rego and Rao 2000), runoff rates (Pathak, et al. 2004), or soil salinity dynamics (Forkutsa, et al. 2009).

Other studies have developed integrated indicators of soil and/or water quality that encompass a variety of specific performance measures related to productivity, off-site environmental, and health impacts (Arshad and Martin 2002; Sanchez et al. 2003). These integrated indicators are essentially weighted averages of several sub-indicators; their accuracy depends fundamentally on the suitability of choices made regarding which sub-indicators to include and the specific weights applied to them.[12] Choice of appropriate indicators will vary substantially, depending on the particular

[12] In particular, exclusion of a potentially important sub-indicator amounts to assigning a weight of zero to it, which in turn can significantly bias in assessment of biophysical impacts (Paul Vlek, pers. comm).

environmental variable(s) of interest and the type of agricultural (or other) activity affecting it. For example, in some systems nutrient availability might be a dominant issue,[13] whereas in other systems it might be soil structure or water holding capacity.

A wide variety of nutrient balance and hydrologic simulation models exist for tracking changes in soil and water quality indicators over time and over space. Models such as the Erosion-Productivity Impact Calculator (EPIC), the Chemical, Runoff and Erosion from Agricultural Management Systems (CREAMS), and the Water Erosion Prediction Project (WEPP) are but a few of the more commonly used models. Constructed using long-term data from multiple locations, models such as these require substantial calibration in order to tailor them to specific locations (Pathak, et al. 2005). They may be embedded as sub-processes within larger crop production models,[14] or within bio-economic models that seek to integrate man-made alterations to the natural resource base and behavioral responses to them.

Several aspects of biophysical measurement have ramifications for environmental impact assessment. First, a significant amount of site specificity characterizes measurement of soil and water quality indicators. Both indicators and the models that simulate their evolution therefore require substantial calibration even to conduct plot-level analysis. A variety of techniques noted earlier are available for these purposes; these include GIS and digital elevation models, reflectance spectroscopy, remote sensing, and use of satellite imagery (in combination with field inspections). Scaling up plot-level results requires a substantial amount of additional, spatially referenced data.

Second, interactions occur among different media. For example, use of insecticides may contaminate groundwater, impact human health, compromise certain wildlife species, and disrupt populations of beneficial predators (Atkinson, et. al. 2004). Simulation of several important environmental outcomes – soil erosion in particular, but also nutrient and pesticide runoff – require modeling soil and hydrologic dynamics jointly to encompass the full range of spatial and temporal effects (Matthews 2006). Development and implementation of a measurement framework capable of synthesizing these multiple interactions requires the services of a relatively broad mix of specialists from multiple disciplines, which will generally add to the cost and time requirements for conducting environmental impact assessment studies, and add to projects' logistical and organizational complexity as well.

[13] Additionally, where nutrient loss is a critical issue, it is important to consider the source of nutrient loss. For example, Drechsel, et al., (2005) note that nutrient loss through crop removal tends to have much more profound impacts on crop production than nutrient loss due to soil erosion.

[14] Wani, et al. (2005, pg. 108) provides a list of ten crop simulation models that employ different approaches to evaluating and projecting the effects of various crop management strategies on long-term productivity, soil quality, and other ecosystem services.

Third, the predictive efficiency of soil and hydrologic models declines substantially at large spatial scales of analysis. Extant models are generally best-suited to analyzing on-site production effects (Roy, et al. 2003). This in no way precludes their use for assessing off-site environmental impacts; indeed, there will in most cases be little alternative to modeling when it comes to quantifying impacts at the meso-level or beyond.[15] It does nonetheless reinforce the need for careful sampling and repeated field verification to be part of the process whereby specific (plot-level) results are scaled up.

Finally, in addition to understanding the origin of environmental impacts, biophysical measurements need to be taken at receiving sites as well. While it is fairly straightforward to measure arrivals of pollutants or other negative externalities at a particular location, establishing causality between those negatives and a particular upstream land use or management regime can be a difficult challenge that may require "expert assessment" (as in the case of the IFPRI's Costa Rica work) or some combination of spatial modeling and physical measurement (as in ICRAF's Trans-Vic study) .

Scale

A primary interest of the CGIAR in assessing the impact of its work is to satisfy donor demands for evidence of substantial returns on their large investments (Walker et al. 2008). While individual Centers may have an interest in impact assessment at a somewhat lesser (farmer or "meso") scales for purposes of evaluating specific research products or programs, there is a clear imperative at the System level to illuminate impacts that are large and are widespread. However, as has been noted at various points in the preceding discussion, a specific management practice or technology can have markedly different biophysical impacts in different locations. This in turn complicates the extent to which particular observed or projected environmental outcomes can be upscaled.

There would appear to be a few basic approaches to addressing the scaling issue. One would involve taking biophysical measurements at multiple locations at different points in time as a means of determining how environmental impacts might vary across different agroecological zones. Monitoring "sentinel sites" over a period of time, as was done by the ASB program, is an example of a coordinated effort to tracking environmental changes on a large scale. This would seem to be an approach best suited to understanding aggregate impacts felt globally such as those related to global climate change or biodiversity.

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[15] In some cases, large-scale environmental impacts may be measured using changes in indicator elements or species. For example, using remote sensing to measure the presence of cesium-137 in the soil – the result of atmospheric testing of nuclear weapons in the 1950s and 1960s – soil scientists have been able to quantify net soil movements over a long period of time (Ritchie and McHenry 1990).

An alternative approach to measuring off-site impacts at a fairly coarse spatial scale is to combine recent innovations in monitoring environmental changes (e..g, reflectance spectroscopy, remote sensing, and use of satellite imagery) with GIS-based spatial modeling techniques. ICRAF's research on agriculture-environment interactions in the Lake Victoria basin (discussed earlier) is an example of this second approach to measuring off-site environmental impacts at a fairly coarse spatial scale. A similar approach to scaling was used by Imbernon (1999) in his investigation of land-use changes in the Kenyan highland over the period 1958-1985. These methods would seem best suited to aggregating impacts on soil and water quality up to a watershed or basin level scale.

Models that explicitly integrate economic and biophysical outcomes can be used to address the scaling issue. Antle et al. (2001) argue that (biophysical) crop growth models alone cannot provide accurate predictions of environmental outcomes beyond a very small scale precisely because those models do not factor in behavioral responses of farmers to economic forces that are themselves affected by the biophysical outcomes. Hence, they call for "fully integrated" models in which a unified set of biophysical and economic drivers jointly (rather than separately) influence biophysical and economic outcomes.

So-called agent-based models represent a means of achieving this level of integration at a meso-level. These models consist of a number of "agents," representing different types of households, livestock and landscapes, who are effectively connected by a set of submodels simulating biological, agronomic and economic processes. For example, Matthews (2006) develops an agent- based model to evaluate potential soil fertility enhancing interventions in Nepal. His model combines simulated water balances, nutrient dynamics, and organic matter decomposition with responses of households to both economic and environmental variables. Berger (2001) develops an agent-based model that integrates economic and hydrologic components within a spatial framework to analyze potential impacts of water-saving irrigation methods in rural Chile. And Le, Park and Vlek (2010) develop an agent-based "land use dynamic simulator" for central Vietnam in order to assess the co-evolutions of human and landscape systems in response to forest protection zoning, agrochemical subsidies, and agricultural extension. All three of these examples were *ex ante* analyses assessing what possible outcomes might emerge under specific policy regimes or technology adoption scenarios. Note, however, that these models could be used for *ex post* analysis by predicting what outcomes would have occurred had realized policy regimes or technology adoption scenarios not taken place – i.e., as a means of projecting relevant counterfactuals.

Finally, there are a few extant examples of large modeling efforts that attempt to synthesize economic and environmental impacts at a geo-regional level as well. The SEAMLESS model developed by a large team of researchers at Wageningen Agricultural University combines a large number of (farm-level) biophysical models with aggregative economic models (like GTAP) to engage in *ex ante* assessment of agricultural and agri-environmental policies in the EU (van Ittersum, et al. 2008). Researchers from Wageningen also were involved in a long-term project that produced an integrated model of economic and biophysical sustainability tradeoffs in the Northern Atlantic Zone of Costa Rica (Bouman, et al. 1998). That effort combined linear programming models of economic surplus maximization and "technical coefficient generators" for livestock and cropping activities with a GIS-based spatial modeling to assess land use changes accompanying a variety of policy shocks.[16]

Attribution

Attributing outcomes to specific research activities is a tricky issue in impact assessment work. Assessment of positive (anticipated) environmental impacts of CGIAR technologies faces issues of apportioning benefits among technology generators that are common to standard epIA work (Walker et al. 2008). Note, however, that in the case of NRM projects, the development of extension capacity and promotion of local property rights institution assumes a much more important role than is the case for other CGIAR research, notably crop genetic improvement work (Zilberman and Waibel 2007). In a sense, this augments Centers' attribution shares due to their greater role in facilitating institutional development and outreach activities.

With regard to assessment of (unanticipated) negative environmental impacts of CGIAR technologies, attribution difficulties are compounded by the large set of underlying drivers that determine environmental outcomes accompanying adoption. Many of these are quite unrelated to technology generation process.[17] One salient example is pricing policies for water, electricity or pumping equipment that hold the cost of irrigation water well below its true economic value. Subsidized irrigation water is commonly implicated as a primary driver of rapid groundwater depletion in well-

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[16] As a cautionary note, these models strike this observer as being so large and containing so many "moving parts" that describing the model's inner workings becomes a nearly impossible challenge. As is commonly the case with aggregative models, the difficulty of clearly communicating where model predictions come from escalates in proportion to the scale of analysis or the complexity of interactions considered. While this latter observation in no way impinges on the scientific integrity of model-based predictions, it is nonetheless relevant to their ultimate influence on policy makers and research managers.

[17] I assume here that CGIAR technologies are largely immune from induced innovation scenarios whereby policy and technology generation processes co-mingle (e.g., de Janvry 1973).

irrigated areas (Pingali, Hossain, and Gerpacio 1997), and as a driver of salinity and/or waterlogging in canal irrigated production environments (Umali 1993). However, the intensity of irrigation on individual plots and the growth of total cropped area under irrigation are also related to the greater returns to irrigation of the improved varieties of the crops being irrigated, and are quite distinct from shifts in demand for water due to water pricing policies. Thus, some "culpability" for irrigation-induced land degradation also falls on the generators of technology such as CGIAR commodity Centers.

Other social and economic policies, population pressures, and property rights institutions are all examples of drivers of environmental outcomes that are frequently exogenous to the research process. The critical role of these outside forces highlight the desirability of integrated models that synthesize biophysical relationships and behavioral responses to them. The bio-economic modeling efforts in Ethiopia noted earlier represent steps in that direction, as do the agent-based models discussed above. Explicitly incorporating population density, fertilizer subsidies and land tax policies into those models allowed the analysts to separate the contributions of those exogenous drivers from the contributions of specific farming practices. Doing so typically requires an estimate of the elasticity of environmental indicator(s) with respect to specific policy or socio-demographic variables.

Valuation

As was noted previously, a number of impact assessment studies conducted within the CGIAR have stopped short of measuring environmental impacts precisely because of an inability to assign monetary values to non-marketed environmental services. This is particularly evident among impact assessments of NRM projects found in the Waibel and Zilberman (2007) volume. But cost-benefit studies of crop genetic improvement have generally neglected incorporating (negative) environmental impacts for this reason as well (Hazell 2009).

As has been already noted, not all environmental services are non-marketed. For example, nutrients that are lost can be replaced (in the form of inorganic fertilizers) and remediation services for saline or waterlogged lands provide a benchmark for the costs of poor irrigation practices. While not perfect substitutes for lost environmental services, these may nonetheless inform assessment of on-site production effects to a reasonable degree. Moreover, in some cases it may be possible to employ hedonic analyses of land price changes to value changes in environmental services.[18]

However, when it comes to assessing off-site environmental impacts, non-market valuation will be required in most circumstances in order to assign monetary values to externalities. Bennett (2009) lays out in some detail a host of approaches for valuing non-marketed environmental goods and services. While still not common in developing country settings, a growing number of studies use stated preference non-market valuation techniques. These include applications to include valuing tropical rainforest preservation (Rolfe, Bennett, and Louviere 2000), conversion of cropland to forest and grassland (Wang, et al. 2007), and wetland restoration (Do and Bennett 2010).[19] Given the rising interest in environmental impact assessment within the CGIAR, there is little doubt that these sorts of valuation exercises will continue to proliferate.

Counterfactual development

A recurring theme in the discussion above has been that different environmental impacts occur at different geographic scales and across different media. This poses interesting challenges for the development of appropriate counterfactuals for environmental impact assessment. Figure 3 illustrates this with reference to a hypothetical productivity-enhancing innovation that mitigates soil losses (say, due to erosion control) at the plot level.[20] This is indicated in the top portion of chart by a lower rate of soil loss associated with the new technology than the traditional technology. Here, the relevant (plot-level) counterfactual lies in the distance between the two solid lines (and <u>not</u> between the baseline soil depth and that of the traditional technology).

The bottom portion of Figure 3 depicts two alternative outcomes at a larger (landscape-level) scale with regard to area cultivated, assuming that the new technology leads to greater productivity (e.g., if soil losses lead to lower fertility). The lower left graph depicts a scenario in which non-adopters bring more area under cultivation in response to declining productivity, while the new technology "freezes" the area under cultivation at the baseline level. In this case, the new technology

[18] Inferring the shadow value of environmental services from bioeconomic models of resource use is another possibility.

[19] In addition to these published articles, a recent edited volume contains a variety of choice modeling applications in developing countries (Bennett and Birol 2010).

[20] This chart was developed by Tim Kelley for a presentation at the Assessment of Environmental Impact of CGIAR Research: Results and Synthesis Workshop organized by SPIA.

produces environmental benefits at both the plot level and at the landscape level relative to what would have been the case absent technological change.

In contrast, the lower right graph depicts a scenario in which non-adopters continue to bring more area under cultivation in response to declining productivity; but now adopters of the new technology expand the area under cultivation to an even greater extent than non-adopters due to the greater profitability of the new technology *vis-à-vis* the traditional technology. In this case, the new technology produces environmental benefits at the plot level but engenders greater environmental damages (associated with land conversion) at the landscape level.

Whichever of these two landscape-scale outcomes is more realistic would be up to the analyst to decide, depending on the specific context. And of course, a host of other factors that might support or limit land expansion – such as land and labor availability – would need to be considered as well. The important point here, though, is that there is a considerable degree of complexity (and uncertainty) associated with determining the relevant comparison.

Other aspects of environmental impact assessment also pose special challenges for the development of appropriate counterfactuals against which actual outcomes must be compared. Several of these have been mentioned, but bear repeating here. First, some future or potential outcomes – especially those related to biodiversity loss and global climate change – are highly uncertain. Second, the importance of complementary property rights and extension institutions to successful diffusion of NRM packages increases the number of elements of counterfactual scenarios that need to be projected. Third, with respect to possible assessment of negative impacts of specific technologies that were adopted in the past, baseline data on environmental assets at the time of initial adoption may well be unavailable. Remedying this situation would require analysts to come up with creative solutions to the problem of (*ex ante*) projection of impacts from a starting point in the past.

Operationalizing the Framework: Two Examples

Box 1 and Box 2 provide two examples of how the various issues discussed above might be approached in the context of specific environmental impact assessment case studies – zero tillage in the Indo-Gangetic Plain (Box 1) and biological control of water hyacinth in West Africa (Box 2). These are by no means intended to be definitive descriptions of the full suite of activities required to pursue environmental impact assessment; rather they are meant to illustrate the sorts of issues that arise and possible means of handling them.

Each potential case study would pose different challenges. In the case of zero tillage, biophysical measurement of reduction in greenhouse gases (due to reduced tractor use) on a per hectare basis would be a relatively straightforward, as would scaling up those imputed reductions over a large area of adoption. On the other hand, measurement of the negative impacts of increased burning of crop residues on local air quality would much more complex undertaking, given spatial variability in weather patterns. Likewise, valuation of greenhouse gas reductions would be facilitated by the existence of carbon markets (the thinness of those markets notwithstanding), whereas valuation of alterations in air quality (due to increased burning) would be far less straightforward.

Ex post assessment of the environmental impacts of biological control of water hyacinth would be a complicated task, in no small part because the environmental benefits take the form of avoided negative consequences had chemical control of the weed taken place. Thus, establishment of an appropriate counterfactual would require projecting the degree of use of various herbicides in different locations where biological control was in fact adopted. This difficulty is in addition to the difficult biophysical measurement and scaling issues associated with the fate and transport of herbicides in different locations, and their impact on flora, fauna, and human health.

Key Lessons and Implications for Moving Forward

This paper has reviewed the current state of knowledge regarding environmental impacts of CGIAR research and laid out a set of issues that need to be addressed in order to pursue meaningful environmental impact assessment in the future. The review of CGIAR research revealed a very thin record accomplishment in the area. This is not entirely surprising: since its inception the overwhelming orientation of the CGIAR as a whole, and its member Centers individually, has been to stimulate production of mandated commodities and to promote policies supporting that goal.[21] Be that as it may, given the CGIAR's now-explicit focus on environmental outcomes as part of its most recent "reinvention," it is clear that environmental impact assessment will become an important element of the future research conducted under the auspices of CGIAR Centers.

[21] Another way of looking at this is that the CGIAR's strong focus on poverty alleviation via enhanced productivity has led to environmental outcomes being overlooked. Even NRM research within the System has strong production orientation. For example, in the introduction to their volume on the subject Waibel and Zilberman (2007) list five objectives of NRM research. Of these, two relate to increasing productivity, two relate to enhancing environmental amenities, and one relates to policy.

A number of important themes emerged from the discussion of critical issues to be addressed in pursuing environmental impact assessment of CGIAR research. These included the following:

- A distinction needs to be made between what have been termed here on-site production effects and off-site environmental impacts. The former generally will be reflected in "standard" *ex post* economic impact assessment. The latter, however, have largely been overlooked by past assessment work and pose distinct challenges both in terms of biophysical measurement and non-market valuation.

- Environmental outcomes from agricultural practices may be positive or negative. The former are generally anticipated consequences of research activities, whereas the latter tend to be unanticipated. Importantly, the benchmark data on environmental stocks and flows required for before-after comparisons of will generally be unavailable for assessing unanticipated negative impacts of existing technologies.

- Environmental impacts will be felt by a variety of different agents, both consumers of environmental goods and producers for whom environmental goods are inputs. The multiplicity of agents that are impacted and the variety of pathways by which those impacts are transmitted, increase the number of measurement and valuation challenges faced by analysts.

- Environmental impacts vary significantly by type of agricultural system (intensive versus extensive, irrigated versus rainfed) and by the scale over which those impacts are generally felt. Principle off-site impacts associated with intensive systems tend to reflect improper management of nutrients, agro-chemicals, and (in irrigated areas) water resources, whereas the primary impacts associated with extensive systems have to do with conversion of lands to agricultural uses.

- Impacts on biodiversity and climate change are global in scale. These pose special challenges with respect to valuation, biophysical measurement, and development of counterfactuals insofar as they hinge on projections of highly uncertain future events.

- A specific management practice or technology can have markedly different biophysical impacts in different locations, so repeated measurement of environmental indicators from a variety of locations is necessary. So too are modeling efforts that reflect this spatial variability, in order to reliably upscale observed or projected environmental outcomes.

Against the array of formidable challenges embedded in these observations, the preceding discussion has also identified a number of tools with the potential to begin addressing these challenges. On the natural science front, a large body of knowledge exists for identifying indicators needed to measure changes in both stocks of environmental goods and flows of ecosystem services emanating from them. Likewise, a variety of models exist for tracking changes in these indicators resulting from various external shocks associated with agricultural technologies and the policy milieu in which they exist (with the caveat that the predictive efficiency of these models declines as the scale of analysis increases). On the economics front, continuing advances are being made in our ability to conduct non-market valuation of environmental goods and services, as evidenced by the growing body of such studies in developing country contexts. Moreover, examples exist of models that synthesize economic and biophysical outcomes in a unified way – albeit mainly at a small scale (e.g,, farm, village or micro-watershed).

In sum, the necessary tools exist for serious pursuit of environmental impact assessment as a mainstream activity of the "new" CGIAR in which Environment for People is now a core objective . What is needed for moving forward, then, is a substantial commitment of organizational, financial, and human resources to the process. Four imperatives stand out in this regard relating to the Systemwide deployment of resources.

First, because environmental impact assessment (EIA) is a complex and costly undertaking, it is not feasible to build an EIA design into each and every new research project (or to subject every completed project to an *ex post* EIA). Rather, there is a need to prioritize which CGIAR projects to subject to this sort of evaluation. A sensible approach in this regard is to focus first and foremost on technologies, practices or policies with (a) the largest aggregate economic impacts, since for the most part these will be the projects affecting the largest number of individuals over the widest geographic area; and (b) the most profound aggregate environmental effects (positive or negative).

In terms of *ex post* assessment of existing CGIAR research outcomes, this approach to prioritization militates toward concentrating more EIA efforts on past crop genetic improvement, pest management, and policy research outcomes, and less on NRM research outputs and outcomes. This is because in general NRM research products generally have been adopted over a relatively limited geographic and demographic scale *vis-à-vis* other types of CGIAR research products (Renkow and Byerlee 2010) –

zero tillage adoption in the rice wheat zone of South Asia being a notable exception.[22] Beyond current SPIA research initiatives into the environmental impacts of past CGIAR crop genetic improvement research on genetic diversity and land use, examples of attractive targets for e*x post* EIA studies include (a) the negative impacts due to increased use of mono-cropping and agro-chemicals of Green Revolution technologies; (b) the positive impacts of biological control of harmful pests.

This approach to prioritization also has implications for deciding on whether or not to incorporate an EIA component into the research design of new projects. One clear theme that has been emphasized throughout this paper is the importance of benchmark measurements of environmental variables against which to gauge *ex post* biophysical outcomes. Hence, research managers making project design decisions will require realistic *ex ante* projection of (a) the likelihood of substantial private economic benefits – to adopters of technologies or practices, or to affected individuals for influential policy research – produced by specific projects; and (b) the likelihood that these substantial private economic benefits affect large numbers of individuals. Where these likelihoods are projected to be large, adoption may be expected to be widespread and hence EIA will be a more efficient use of scarce research resources.

Second, there is a need to build environmental monitoring and valuation strategies into project design. Benchmark measurements taken prior to projects' initiation are critical to gaining an *ex post* understanding of the environmental outcomes attributable to technological change, as well as for facilitating appropriate counterfactual development. And given that environmental outcomes typically unfold slowly, measurements will need to be taken over an extended time frame. Likewise, tackling the valuation problem will require considerable up-front planning in terms of survey design and other data collection activities. Creative uses of data on prices of environmental services for which markets exist will be useful in this regard, although these will likely be primarily useful for investigating on-site production effects. Measurement of externalities will generally require use of non-market valuation techniques.

Third, it is clear that a considerable amount of financial resources will have to be devoted to vigorously pursuing environmental impact assessment as a core element of the CGIAR's portfolio of evaluative activities. Tracking environmental outcomes requires investigators to take measurements of biophysical variables at multiple

[22] That said, there may be significant payoffs to EIAs of smaller-scale NRM projects – such as is being pursued in current SPIA-sponsored NRM case studies – if those case studies result in improving techniques for addressing methodological challenges related to biophysical measurement and the aggregation (upscaling) of those measurements spatially.

locations at different points in time. Incorporating environmental impact assessment as standard component of project design also will likely mean increasing the size of research teams due to the highly interdisciplinary nature of the work required. The activities needed to generate and analyze the requisite data for this task are time consuming, logistically complex, and require substantial human resources. All of these things add significantly to project costs. Importantly, these expenses should represent additions to existing research costs. It would be highly undesirable for research activities oriented toward understanding environmental outcomes to in any way compromise the System's core mission of enhancing agricultural productivity (or evaluation of those productivity impacts).

Finally, some changes in the human capital base on which the CGIAR draws would appear warranted. Existing manpower at specific Centers may not have the expertise to pursue some of the analytical tasks that need to be undertaken. For example, most economists within the System have considerably more expertise in the areas of agricultural production and marketing than they do in environmental economics issues (and in particular, non-market valuation). While some re-tooling might be feasible, augmenting existing staff resources to include environmental economists would seem to be inevitable. Alternatively, there is no doubt scope for partnering with institutions and individuals outside of the CGIAR that have a comparative advantage in research on environmental . Such "outsourcing" of research tasks is not unprecedented, and may in many circumstances represent a more efficient approach to pursuit of a System-wide research agenda in which environmental issues are more prominent.

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Figure 1. Economic, Environmental, and Social Impacts of Agriculture

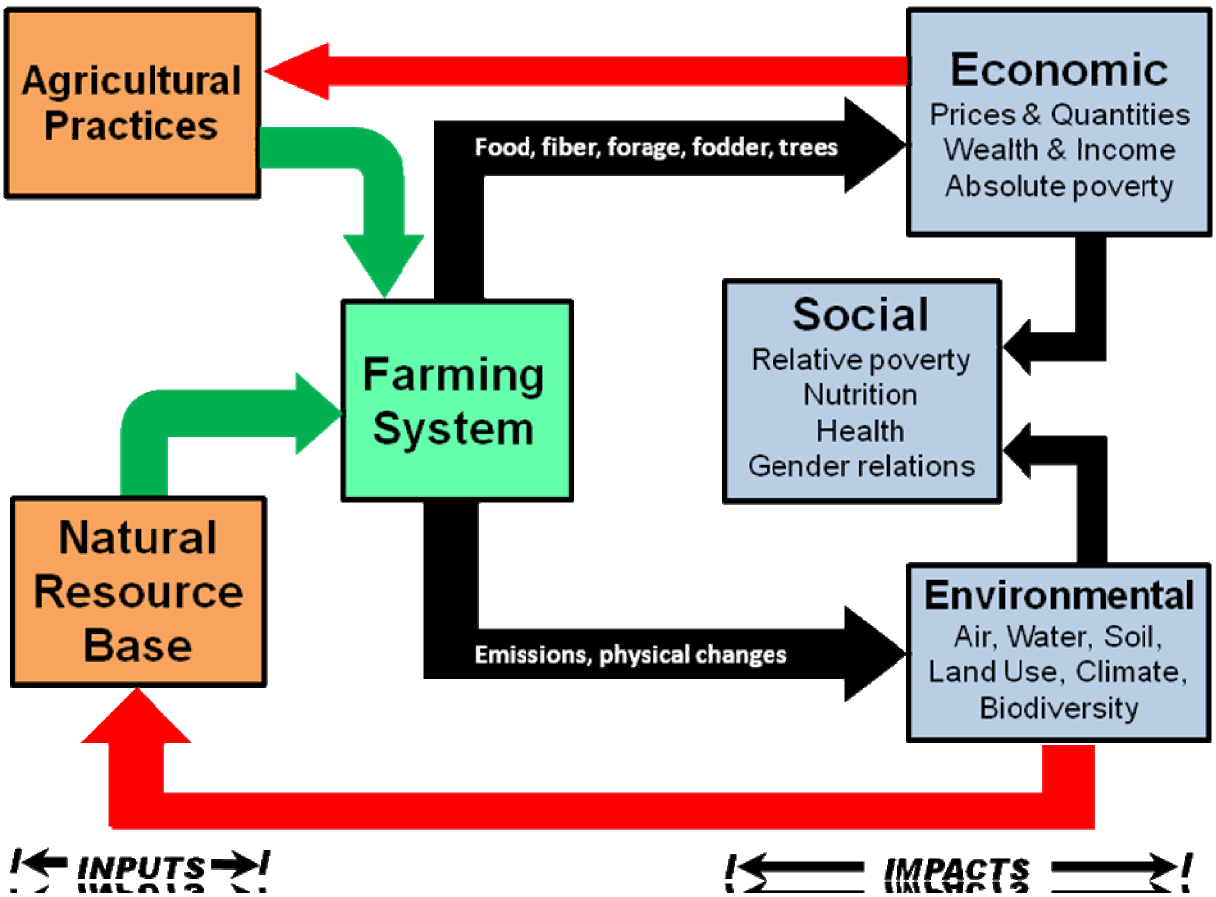

Figure 2. Pathway from Research and Extension to Economic and Environmental Impact Assessment

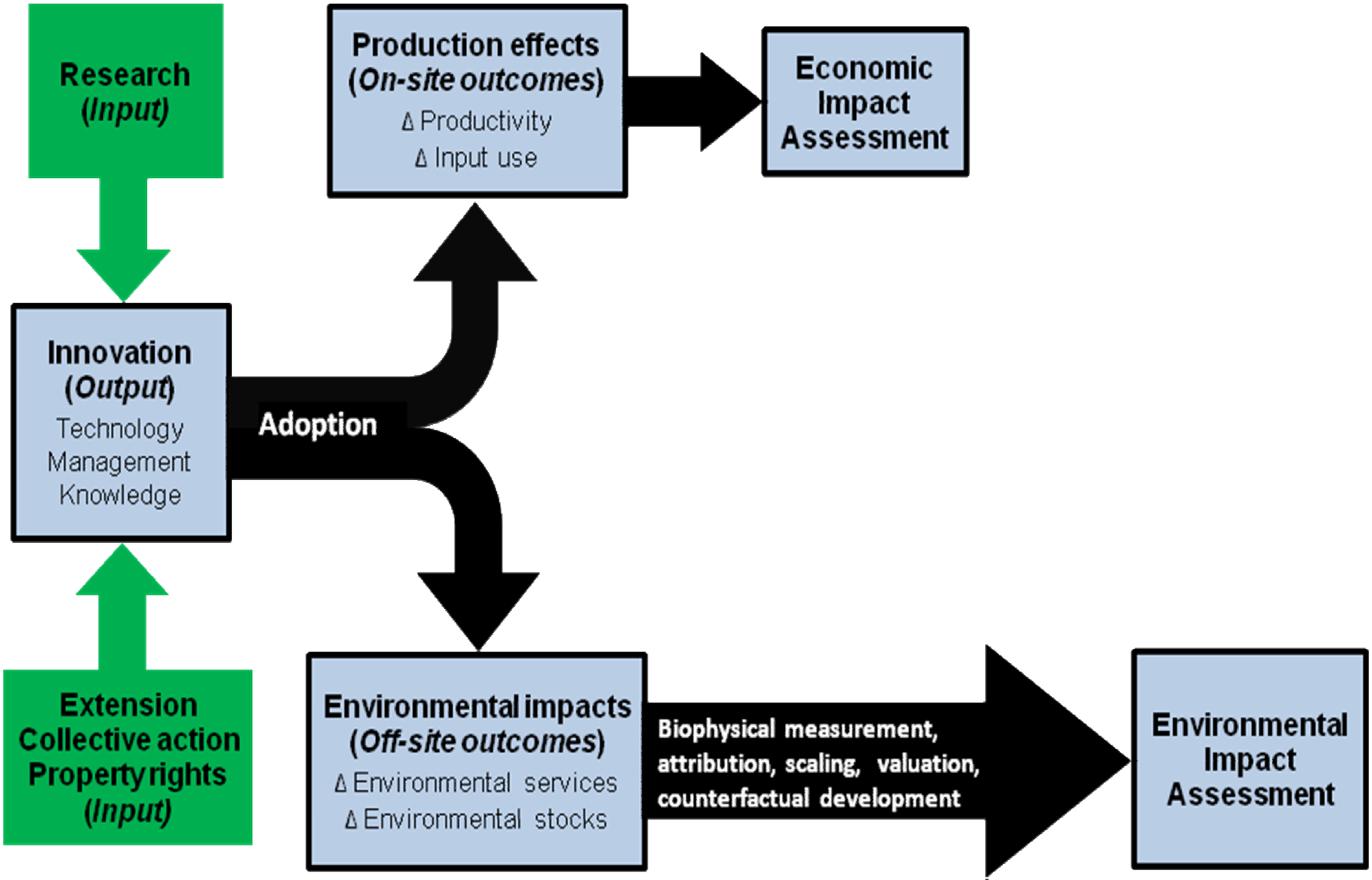

Figure 3. Appropriate Counterfactuals at Different Scales

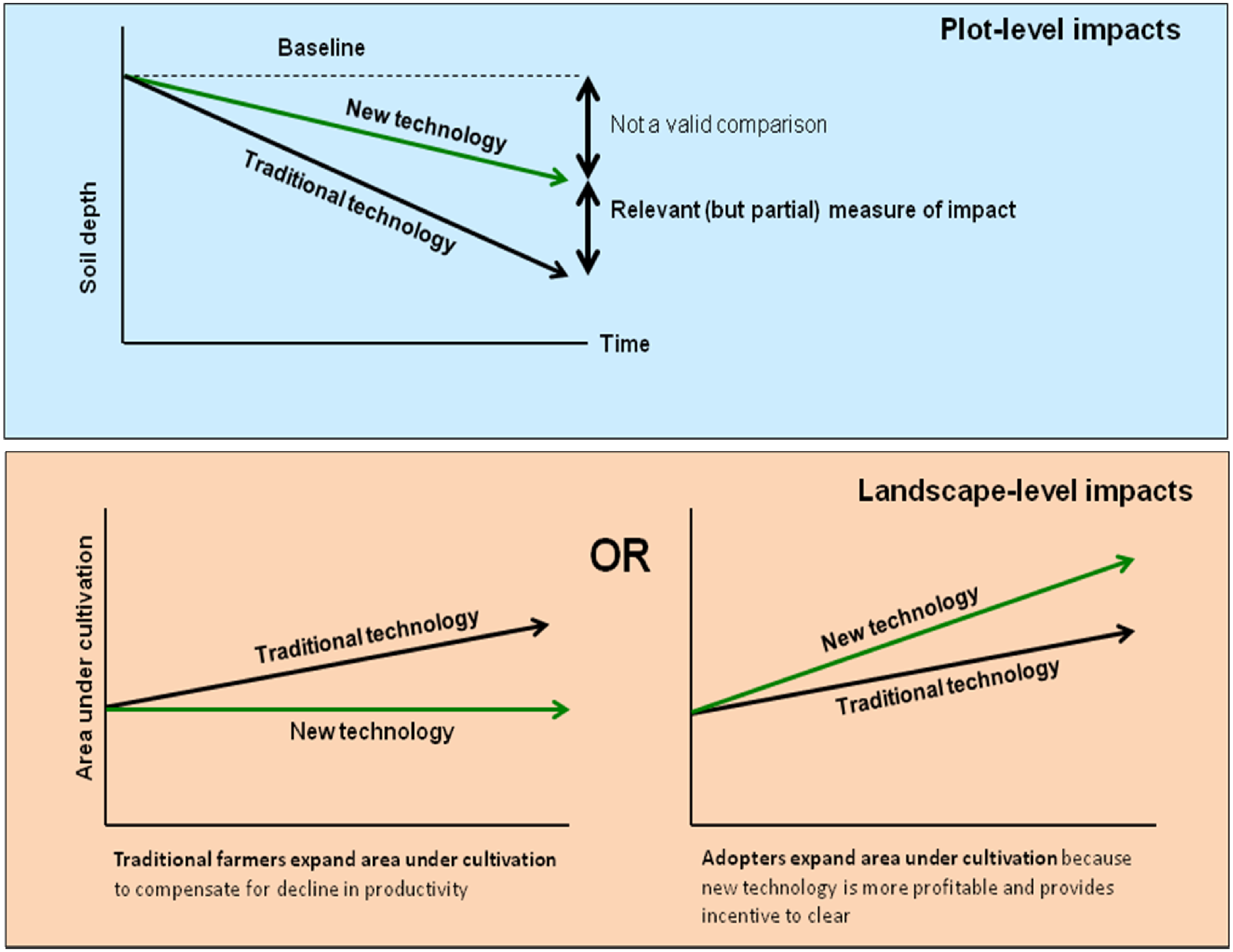

Table 1. CGIAR Studies Relevant to Environmental Impact Assessment

Center (Timing)	Location (Scale)	Focus[a] (*ex post* or *ex ante*)	Key Findings with Respect to Environmental Impacts
CIFOR[1] (2000-2006)	Indonesia (National)	Impacts of CIFOR's research on the political economy of the pulp and paper sector and fiber sourcing practices (*ex post*)	- Improved sustainability of pulp production and regulation of pulp sector - Averted loss of between 76,000 and 212,000 ha of natural forest (135,000 ha under the main set of assumptions) - Net present value of benefits = \$19 million to \$583 million (\$133 million under main set of assumptions) vis-à-vis < \$500,000 investment costs
IRRI[2] (1989-1995)	Philippines (National)	Pesticide impacts on farmer health, ground and surface water contamination, and rice ecosystem function (*ex post*)	- Very large negative human health impacts, particularly to agricultural households; minimal productivity impacts from reduced pesticide use - Only small negative impacts on ecosystem function - High rate of return to research on nonchemical pest control methods
CIP[3] (1989-1998)	Ecuador (Watershed)	Pesticide impacts on farmer health, ground and surface water contamination, and potato ecosystem function (*ex post*)	- Very large negative human health impacts, particularly to agricultural households - Positive productivity impacts of pesticide use => tradeoff with health impacts - Little evidence that pesticide leaching poses a threat humans
ICRAF[4] (1999-2005)	Kenya (Basin)	Water and sediment yield of different land use systems. **No CGIAR product evaluated** (*ex post*)	- Evidence found for synergies (win-win), tradeoffs (win-lose), and poverty traps (lose-lose) with respect to agriculture-environment links - Substantial spatial variability in outcomes
CIMMYT[5] (1994-2007)	Indo-Gangetic Plain of India and Pakistan (Regional)	Zero-tillage in irrigated rice-wheat farming systems (*ex post*)	Positive: Modest water savings and improved irrigation efficiency (<u>wheat only</u>) in India but not Pakistan Positive: Reduced diesel consumption (~\$50 million annually) Positive: Reduced CO_2 emissions (~91 kg/ha) Negative: Air pollution due to burning non-basmati rice residues

Table 1. (continued)

Center (Timing)	Location (Scale)	Focus	Key Findings with Respect to Environmental Impacts
ICRISAT[6] (2005)	Ethiopia	Bioeconomic models of soil conservation technologies at the farm and village levels **No CGIAR product evaluated** (*ex ante)*	- Household level *ex ante* predictions suggest that conservation investments will only occur where land is scarce and labor plentiful - Village-level simulations suggest that removal of fertilizer subsidies will worsen land degradation, especially for poor households
CIAT[7] (2007-2008)	Amazon Basin and East Andean Slopes (Regional)	Water quantity & quality, local & global climate regulation, soils, biodiversity via consultation with various stakeholders **No CGIAR product evaluated** (*ex ante)*	- Rural inhabitants are most vulnerable to changes in environmental services provision. - Traditional and indigenous populations particularly vulnerable to changes in flows of environmental services - Recommends more biophysical, socio-economic and policy research
Worldfish[8] (1995-2004)	Malawi (National)	Integrated aquaculture/agriculture (IAA) systems (*ex post)*	- Describes (but does not quantify) IAA-related environmental outcomes related to species diversity and waste/by-product recycling
IWMI[9] (1995-2005)	Global	Irrigation management transfer (IMT) programs (*ex post)*	- Substantial contribution to knowledge from of IMT - Positive operational contribution of IMT to institutional effectiveness
CIP[10] (2006)	Peru (Watershed)	Pesticide use and environmental impact quotient (EIQ) (*ex post)*	- Substantial variability in EIQ across locations found. - Lack of correlation between EIQ and productivity suggests opportunities for reduction in pesticide use via greater use efficiency and/or IPM strategies.
IFPRI[11] (1999-2001)	Costa Rica (Watershed)	Monitoring system for integrating environmental, economic, and institutional outcomes from multiple land uses **No CGIAR product evaluated.** (*ex post)*	- Results "illustrate an approach" rather than being "definitive" - Method centers on computing a Payoff Matrix that includes direct impacts plus externalities created for different stakeholders/interests - Payoff matrix circumscribes potential Coasian solutions

a. Unless otherwise noted, all studies evaluated outcomes and/or impacts of CGIAR technology, management, or knowledge products

Sources: 1. Raitzer (2008); 2. Pingali and Roger (1995), Templeton and Jamora (2007); 3. Crissman, Antle, and Capalbo (1998); 4. Swallow, et al. (2009); 5. Erenstein, et al. (2007), Farooq, et al. (2007), Laxmi, et al. (2007a); 6. Shiferaw and Holden (2005), Holden and Lofgren (2005); 7.ESPA-AA (2008); 8. Dey, et al. (2007); 9. Giordano (2006); 10. Pradel, et al. (2009); 11. Hazell, et al. (2001).

Table 2. Environmental impacts by source and type of agriculture

	Scale of Impact			Type of Agriculture	
Type of impact	**On-site**	**Local**	**Global**	**Intensive**	**Extensive**
Land					
Salinization & waterlogging	x			x	
Nutrient depletion	x			x	x
Loss of organic matter (soil erosion)	x	x		x	
Conversion of non-agricultural lands*		x	x		x
Water					
Groundwater depletion		x		x	
Water conservation	x	x			x
Agrochemical pollution					
Human health	x	x		x	
Animal health	x	x		x	
Plant health	x	x		x	
Animal					
Animal wastes	x	x			
Animal diseases		x	x	x	
Common property pasture degradation		x			x
Biodiversity loss					
Local biodiversity		x		x	
In situ crop genetic diversity			x	x	
Conversion of non-agricultural lands*			x		x
Climate Change					
GHG emissions from ag. operations			x	x	
Release of soil carbon			x	x	x
Reduced C sequestration*			x		x

* Denotes impact linked to deforestation

Table 3. Biophysical Indicators

Criteria	Indicators
1.Biodiversity	• Species richness • Species diversity • Species risk index
2.Agro-biodiversity	• Index of surface percentage of crops • Crop agro-biodiversity factor • Genetic variability • Surface variability
3.Agroecosystem efficiency	• Productivity change • Cost-benefit ratio • Parity index
4.Environmental services	• Greenery cover/vegetation index • Carbon sequestered • Emissions of greenhouse gases • Land degradation/rehabilitation of degraded lands
5.Soil quality	• Soil physical indicators (e.g., bulk density, water infiltration rate, water holding capacity, water logging, soil loss) • Soil chemical indicators (e.g., soil pH, CEC, organic C, inorganic C, total and available N, P, and other nutrients, salinity) • Soil biological indicators (e.g., soil microbial biomass, soil respiration, soil enzymes, biomass N, diversity of microbial species)
6.Water availability and quality	• Quantity of fresh surface water available • Fluctuations in groundwater level • Quality of surface water and groundwater

Source: Wani, et al. (2005)

BOX 1: ASSESSING THE ENVIRONMENTAL IMPACTS OF ZERO TILLAGE ADOPTION IN THE INDO-GANGETIC PLAIN

Zero tillage (ZT) in the rice-wheat farming systems of the Indo-Gangetic plains represent the most profoundly influential natural resource management activity to date within the CGIAR in terms of the geographic scope of diffusion and the number of farmers affected. The Rice-Wheat Consortium – a network of national, regional and multi-lateral partners including CIMMYT and IRRI – has developed and promoted several resource-conserving crop management technologies, the most widely adopted of which is zero tillage (ZT). The key technological component of ZT is use of specialized seeding and fertilization machinery. The magnitude of increased farm profits attributable to these improvements has been well documented. To date, zero tillage is almost exclusively practiced on the wheat side of rice-wheat rotation.

Key off-site environmental impacts: (1) Reduced greenhouse gas (GHG) emissions due to reduced tractor use. (2) Air pollution due to burning of greater amounts of crop residues (i.e., the residues that are burnt instead of being tilled back into the soil).

Biophysical measurement: (1) Average per hectare reductions in tractor use (and hence GHG emissions) could be computed for a variety of representative farms in the region. (2) Per hectare increases in crop residues burnt could be computed for a variety representative farms; this information could then be combined with information on particulate matter and other pollutants produced per unit of residue burnt to estimate the contribution to air pollution.

Scaling: (1) For ex-post assessment of impacts on GHG emissions, the area under ZT provides the primary benchmark. Average per hectare reductions in tractor use could be applied to the aggregate area to compute total effects for the region. (2) Considerably more creativity would be required to infer how this increased burning would negatively impact (local) air quality over space, given the effects of wind and other climatic factors on the dispersal of pollutants. Small scale measurement of particulate matter and other air pollutants from the burning of a hectare's worth of crop residue would need to be incorporated into aggregative models of weather patterns and/or airflows. Seasonality of weather patterns would no doubt be an important complicating factor as well.

Valuation: (1) The value of reduced carbon emissions can be imputed from prices on Chicago Climate Exchange or some other carbon market operating through the Kyoto Protocol. Given the thinness of these markets and the large variability in these values, a range of possible carbon prices may need to be employed. (2) One approach to valuing the negative impacts of air pollution from burning residues would be to establish a value for time lost due to illness that is associated with elevated pollution levels. Another would be to employ stated preference methods to value the willingness to pay for air quality improvements.

Counterfactual: The appropriate counterfactual scenario for establishing the total environmental impacts of ZT is that conventional tillage would have been undertaken on all farms in the region.

Attribution: Laxmi, Erenstein, and Gupta (2007b) attributed CIMMYT's share of the economic gains from ZT by assuming that diffusion assuming that diffusion occurred more rapidly than would have been the case absent CIMMYT's involvement – i.e., that it would have followed the same (logistic) adoption curve, but with a lag of five years. A similar strategy would appear appropriate for inferring CIMMYT's contribution to net value of both the positive and negative environmental impacts.

BOX 2: ASSESSING THE ENVIRONMENTAL IMPACTS OF BIOLOGICAL CONTROL OF WATER HYACINTH

Water hyacinth is a fast-growing ornamental plant of South American origin that has become a highly damaging waterweed in tropical and subtropical regions worldwide. In the 1980s it became a major threat to West African creek and lagoon systems from which many individuals derived their livelihoods – primarily by hindering fishing and transport, and in some locations by interfering with water use for irrigation, drinking water, and electricity generation purposes (Alene, et al. 2005). IITA-led collaborative efforts on biological control of water hyacinth led to the release of three host-specific natural enemies that have greatly reduced the scope and magnitude of negative economic impacts of the water hyacinth problem. Successful use of biological control methods obviated the need for chemical and mechanical methods for water hyacinth mitigation. Research by De Groote et al. (2003) estimates the present value of total net economic benefits of the program in Southern Benin alone to be US $258 million (in 1994 dollars).

Key off-site environmental impacts: Avoided negative effects of chemical pesticides on the human health, flora, and fauna in waterways where biological control of water hyacinth has been undertaken.

Biophysical measurement: The key information need is to quantify the negative impacts of (avoided) chemical control methods on the health of humans, non-human fauna, and flora. This would require assembling information on which chemicals were (or are likely to have been) used in different geographic locations; the amount of chemical use in each location; the toxicity of the various chemicals to specific organisms; the spatial extent and duration of those toxic effects; and a "census" of number of humans, flora and fauna likely to be exposed to toxic chemicals (as well as the intensity of that exposure). Measuring observed negative ecological impacts of chemical control in places where chemicals have been used would be an important component of this exercise.

Scaling: Avoided negative impacts of chemical use in specific locations would need to aggregated across multiple areas in which biological control was employed. This could entail projecting the likelihood that specific chemical treatment regimes would have been employed in representative locations. To the extent that such an exercise is feasible, upscaling would then require aggregating the value of projected negative impacts.

Valuation: For human health effects, lost work time due to chemical exposure is one possible approach to valuation. Alternatively, stated preference methods could be employed to estimate perceived costs of exposure to harmful chemicals (e.g., willingness to pay for avoided negative health effects). For flora and fauna that are consumed by humans, their market prices would to some degree facilitate computing the value of foregone consumption (to the extent that tainted organisms are not consumed). Alternatively, the value of ecosystem services provided by organisms projected to be affected could be estimated using stated preference methods.

Counterfactual: One counterfactual scenario would be that in all areas in which biological control has taken place, chemical treatment would have been employed. This would require projecting which chemicals would have been used in different locations. A useful refinement to this would be to project whether mechanical harvesting of water hyacinth would have been used (and if so, to net this out from the total area likely to have been treated). Sensitivity analysis might be desirable to accommodate the possibility that some locations would have simply not dealt with water hyacinth problems at all. Finally, an appropriate counterfactual would have to accommodate the negative environmental impacts that (untreated) water hyacinth proliferation creates (e.g., increased incidence of malaria, reduced fish populations).

Attribution: Estimating the net benefits attributable to IITA would require some partitioning of total benefits among the many collaborators on this work – possibly in proportion to the share of overall project costs borne by IITA. As most of the cost of biological control is attributable to salaries (De Groote, et al. 2003), this would simplify matters insofar as only manpower allocations (and associated salaries) would be needed.