INDEPENDENT SCIENCE & PARTNERSHIP COUNCIL

CGIAR

Measuring the Environmental Impacts of Agricultural Research: Theory and Applications to CGIAR Research

October 2011

CONSULTATIVE GROUP ON INTERNATIONAL AGRICULTURAL RESEARCH

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CGIAR Independent Science and Partnership Council. 2011. Measuring the Environmental Impacts of Agricultural Research: Theory and Applications to CGIAR Research. Independent Science and Partnership Council Secretariat: Rome, Italy.

Contents

Acronyms and abbreviations

AgLU	Agriculture and Land-Use model
ASB	Alternatives to Slash and Burn
BCA	benefit-cost analysis
CET	constant elasticity of transformation
CGE	computable general equilibrium
CGIAR	Consultative Group on International Agricultural Research
CIAT	International Center for Tropical Agriculture
CIFOR	Center for International Forestry Research
CIMMYT	International Maize and Wheat Improvement Center
CIP	International Potato Center
CM	choice modeling
CO_2	carbon dioxide
CREAMS	Chemical, Runoff and Erosion from Agricultural Management Systems model
CRP	CGIAR Research Program
CVM	contingent valuation method
EIA	environmental impact assessment
EIQ	environmental impact quotient
EMBRAPA	Brazilian Agricultural Research Cooperation
epIA	*ex-post* impact assessment (see also definitions below)
FAO	Food and Agriculture Organization of the United Nations
FASOM	Forest and Agriculture Sector Optimization Model
GHG	greenhouse gas
GIS	geographical information system
GTAP	Global Trade Analysis Project
GTAP-AEZ	Global Trade Analysis Project - Agroecological zone model
ha	hectares
IAA	integrated aquaculture-agriculture
IARC	international agricultural research center
IBGE	Brazilian Institute of Geography and Statistics
ICAR	Indian Council of Agricultural Research
ICARDA	International Center for Agricultural Research in the Dry Areas
ICRAF	World Agroforestry Center
ICRISAT	International Crops Research Institute for the Semi-Arid Tropics
IFPRI	International Food Policy Research Institute
IITA	International Institute for Tropical Agriculture
IMPACT	International Model for Policy Analysis of Agricultural Commodities and Trade
IMT	irrigation management transfer
IPM	integrated pest management
IRRI	International Rice Research Institute
ISPC	Independent Science and Partnership Council
IWMI	International Water Management Institute
NARS	National Agricultural Research Systems
NGO	non-governmental organization
NRM	natural resource management
PRA	participatory rural appraisal
R&D	research and development
REDD	reduced emissions from deforestation and degradation
Rs	Indian Rupees
SEAMLESS	System for Environmental and Agricultural Modelling: Linking European Science and Society
SI	supplemental irrigation
SPIA	Standing Panel on Impact Assessment
SRF	Strategy and Results Framework
SYP	Syrian Pound

TFP	total factor productivity
WATSIM	World Agricultural Trade Simulation System
WEPP	Water Erosion Prediction Project
ZT	zero-tillage

***Ex-post* impact assessment (epIA)** is a specialized area of evaluation designed to identify and measure the consequences resulting from a program or project's earlier interventions. These take place after a program or project has generated the intervention being assessed, and after sufficient time has elapsed and experience have accumulated to assess the intervention's performance in terms of longer-term economic, social, and environmental consequences.

***Ex-ante* evaluations** provide a description and analysis of the expected outcomes, and likely impacts, of a program or project before these have actually been realized.

Acknowledgements

The authors would like to thank staff from the participating CGIAR Centers who worked on the four case studies synthesized in chapter 2, in particular: Aden Aw Hassan, Tamer El-Shater, Guy Hareau, Suresh Pal, I Sekhar, Kamel Shideed and Dennis Wilchens. The Standing Panel on Impact Assessment would also like to thank the following experts for their advice during this project: John Dixon, Jeff Sayer, Bekele Shiferaw and Paul Vlek, and four anonymous external reviewers for useful comments on earlier drafts of chapters 1 and 3.

Foreword

Agricultural research generates technologies and information that when adopted by end users, results in economic, social and environmental impacts. The economic impacts of CGIAR-generated technologies, especially in germplasm improvement, have been widely documented (Evenson and Gollin, 2003; Raitzer and Kelley, 2008), but a comparable effort in documenting other types of impacts is lacking (Renkow and Byerlee, 2010). The inclusion of environmental impacts in the evaluation of agricultural research, both positive and negative, would yield a more complete picture of the overall returns to investments in R&D, even if not all of these impacts are measured in monetary terms. If, as Waibel and Zilberman (2007) suggest, the overall returns for some kinds of research are underestimated, then more comprehensive documentation of impact is likely to have a positive effect in terms of enhancing donor confidence in the CGIAR as an effective mechanism for achieving broader development goals. Even when environmental consequences are negative, their documentation would provide a more credible and comprehensive impact assessment that may still show a positive net impact. A broadening of the focus of impact assessment is consistent with the findings of the Independent Review Panel of the CGIAR (2009) which recommended that "future *ex-ante* and *ex-post* impact assessment make an effort to accurately assess environmental, gender and other indirect consequences of agricultural research for development".

In late 2008, the CGIAR Standing Panel on Impact Assessment (SPIA) launched a study on environmental impact assessment in the CGIAR with the goal of increasing the availability of information relating to the environmental impacts from CGIAR research, positive and negative, intended and unintended. The study had three objectives:

1. Adapt, apply and evaluate emerging approaches to assessing *ex-post* environmental impacts of agricultural research with high relevance to the CGIAR's mandate.
2. Advance SPIA's guidelines for conducting *ex-post* impact assessment, with particular emphasis on environmental impacts, seeking where possible, to explicitly value environmental impacts in monetary terms in order to build on earlier economic impact assessments and thereby achieve a more comprehensive (integrated) assessment of impact.
3. Provide results of environmental impacts from a range of case studies reflecting different types of research within the CGIAR and different dimensions of the environment.

The study was conducted in three phases. Phase I (October 2008 to early 2009) entailed the commissioning and writing of two background review papers. The first of these, from Djurfeldt et al. (2009) gave an overview of a range of potential environmental (and social) variables that could usefully be assessed by the CGIAR in *ex-post* impact assessments. These authors also strongly recommended that *ex-post* impact assessment practice in the CGIAR be based on more rigorous counterfactual analysis (using experimental or quasi-experimental controls) than has been the case in the past. The second review paper by Bennett (2009) was grounded in the traditional territory of cost–benefit analysis, and argued that an extended cost–benefit analysis approach in which environmental impacts are valued in monetary terms, is the only theoretically consistent framework available for reconciling trade-offs between economic, social and environmental objectives. These can be seen as complementary perspectives, focusing on different aspects of the challenges facing researchers in carrying out environmental impact assessments, i.e., measurement and valuation.

Phase II (April 2009 to December 2010) was based on SPIA's call for proposals from the CGIAR Centers for case-studies focusing on the environmental assessment component of specific agricultural technologies or policies. Eleven proposals were received and externally reviewed by a panel of six experts. SPIA selected six case studies, one each from IWMI, World Agroforestry Center, CIAT, ICARDA, CIP and the Indian Council of Agricultural Research (ICAR, evaluating CIMMYT-related research). Jeff Bennett, Director of the Environmental Economics Hub at Australian National University and an expert in non-market valuation of environmental goods and services, acted as lead consultant, with the task of assisting SPIA in supporting the work of the case study leaders from each of the CGIAR Centers. Workshops in Rome in 2009 and Istanbul in 2010 also provided technical support to the case study leaders.

Phase III of the study (February 2010 to December 2010) was launched to enable SPIA to consolidate the literature to date on environmental *ex-post* impact assessment of international agricultural research. Two papers were initiated in this phase. Mitch Renkow, natural resource economist at North Carolina State University, was commissioned to carry out a comprehensive review and assessment of the available documented evidence of impacts on the environment from agricultural technologies and policies derived at least in part from CGIAR research. He was also asked to develop a framework and possible methods for understanding the environmental impacts of agricultural research, describing the lessons for future environmental impact assessment work in the 'new CGIAR'. Renkow's paper is presented as chapter 1 of this report. The last chapter of this report focuses on the hypothesis first put forward by Norman Borlaug in the early 1980s, that the Green Revolution saved land from being converted to agriculture by raising yields on the existing agricultural areas. There is now an extensive and complex literature on the issue and SPIA itself decided to review the theory behind this hypothesis, and the variety of empirical approaches used to assess its validity.

Main findings

Renkow's review (chapter 1 in this report) reveals a very thin record of accomplishment in environmental impact assessment in the CGIAR. Off-site environmental impacts of CGIAR research have largely been overlooked. The limited evidence available is focused on on-farm environmental production effects which, theoretically, should already be captured under a standard economic analysis via their impacts on productivity. Renkow found only a small number of cases in which even a partial analysis was carried out, the notable exceptions being for timber harvesting policies on deforestation, and for pesticide reduction policies (although in the latter case, the focus was mainly on human health impacts). There has been no work carried out to date that has successfully traced the entire impact assessment pathway from research investment through to measurement of off-site biophysical effects on ecosystem services, and on to the ultimate impacts on agents located in receiving sites. Almost the only comprehensive and broadly conceived environmental impacts had been conducted by CIFOR on impacts of forest policy research. Renkow provides a typology of environmental impacts that differentiates between the scale at which impacts occur (on-site, local, global) and by the type of agricultural system in which the impacts occur (extensive, intensive).

Bennett's paper in this volume (chapter 2) provides an overview of methods for non-market valuation of costs and benefits. It shows a way forward for incorporating environmental and social values into *ex-post* impact assessment's dominant methodology of economic cost-benefit analysis. The main content of chapter 2, however, is a summary of the results from four of the more comprehensive case studies from the set of six originally commissioned.

- The International Centre for Agricultural Research in the Dry Areas (ICARDA) case study examined the relationship between the introduction of supplemental irrigation of wheat in Syria on the extent of groundwater depletion and soil salinity accumulation. Using a range of valuation methods, the estimated environment-related benefits ranged from zero to over US$8.5 million per annum across the wheat growing areas of Syria. This variability is attributable to different assumptions about the fate of the water 'saved' under supplemental irrigation.
- The Indian Council for Agricultural Research (ICAR) case study estimated the

change in water availability from the introduction of zero tillage in rice-wheat systems in the Indo-Gangetic Plains, that farmers valued at US$142 per person when presented to them as a 'water saving strategy'. However, there was a concern that this estimated 'willingness to pay' was not well grounded by a definition of what the water saving strategy would actually deliver, and to whom.
- The International Water Management Institute (IWMI) study examined the impact on flora and fauna of a change in sluice gate operations in the Mekong delta – a change implemented by the Vietnamese government following research carried out by IWMI. On average, individual households living in the delta were willing to pay between US$39 and US$73 per annum for this change in sluice gate operations, and that between 12 and 15 percent of this is attributable to favorable changes in flora and fauna, yielding an aggregate value of up to US$200,000 per annum.
- The International Potato Center (CIP) case study tracked changes in potato diversity over time in Peru, and used choice modeling to explore the trade-offs farmers make between modern and native varieties. Full analysis of this data was not undertaken, but the extent to which farmers are willing to trade-off improved yield with reduced biodiversity could potentially be calculated.

The synthesis of the experience offered by Bennett emphases the positive issues in these case studies, and that "the integration of environmental impacts into the rubric of cost-benefit analysis is practical and capable of application". However, it also recognizes the limitations posed by the fact that case study leaders faced data availability constraints and had limited experience in methods of environmental economics. Two of the six commissioned studies remain as 'work in progress due to these limitations. It appears to be too early to judge the environmental impacts of alternative rubber production systems in the World Agroforestry Center's study. In addition, further work is required by CIAT to establish the environmental impacts and associated values of introducing improved bean varieties in Uganda to fill current knowledge gaps.

Chapter 3 provides a review of literature on agricultural technologies and land-use change, and results from new analyses carried out by Nelson Villoria at Purdue University using the Global Trade Analysis Project (GTAP) model. These modeling results suggest that in the absence of CGIAR-related investments in germplasm research, the total global cropped area at the start of the 21st century would have been some 20–25 million hectares larger. This result may represent the single most important environmental impact of the research carried out by the CGIAR, with implications for carbon emissions, biodiversity and water use. However, the effects of agricultural intensification on land-use change are complex, and much depends on the governance of forest and land resources that is not adequately incorporated into even the most sophisticated of these models.

Implications for impact assessment in the CGIAR

SPIA is pleased that the case studies reported here, for the most part, show that the conceptual framework developed to integrate environmental impacts into cost–benefit analysis is practical and capable of application. Each of the studies reported in chapter 3 produced results that provide qualified but useful conclusions to the Centers and to the CGIAR as a whole. They demonstrate the relative magnitudes of some of the environmental impacts resulting from Center investments, albeit on a limited scale and with some technical problems that have yet to be resolved. At the same time, they demonstrate the inadequacy of the effort of the Centers to separate the relationships between research investments and environmental outcomes, including feedbacks into farm productivity. The primary focus of the biophysical research effort in the relatively few cases reviewed has not been on the outcomes that are important as values for people. Rather they, apart from the IWMI case study, have concentrated on outcomes that are intermediate (for example, water quantity and quality rather than the richness of species that depend on the water). The studies reported can best be characterized as taking an initial step toward assessing environmental impacts, based largely on the more familiar traditional farming system and productivity work that has characterized CGIAR *ex-post* impact assessments.

A lack of clear incentives at the system level, combined with the high cost of constructing complex biophysical models and collecting good data on changes in environmental quality, has resulted in the CGIAR underinvesting in the models and datasets required for more integrated *ex-post* impact assessment. This study was supposed to overcome some of these constraints, but it is found that none of the Center-led case studies have been completely successful. In hindsight, a number of case study leaders reported that resource constraints were binding, particularly with competing demands on their time over the past two years – a time of reorganization and reform in the CGIAR system. SPIA recognizes that the funding for the case studies was limited and thinly spread.

Over the three years during which this work was carried out, there have been significant changes in the CGIAR system, with proposals developed for a set of CGIAR Research Programs (CRPs) organized under a single Strategy and Results Framework (SRF) for the CGIAR. In this SRF, there are four system-level outcomes: reducing rural poverty, increasing food security, improving nutrition and health, and sustainable management of natural resources. As Renkow's paper concluded: "The review of CGIAR research revealed a very thin record of accomplishment in the area [of environmental impact assessment]. 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. 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."

In order for the CGIAR system to monitor how its suite of new long-term programs helps support progress towards these four system-level outcomes, there is a need for greater and more coordinated investment across the CRPs in identifying appropriate indicators of environmental impacts for monitoring purposes, and collecting and analyzing those data for environmental impact assessment. However, obtaining the relevant environmental and agricultural data is only part of the challenge for *ex-post* impact assessment. *Ex-post* impact assessment will still need to bring these data into a coherent study. Such a study must feature a clear model of how agricultural technologies and environmental outcomes are linked (Jeff Bennett refers to this as an 'environmental production function'), constructing a valid counterfactual (or 'without technology') scenario, and then valuing the environmental changes attributable to technology adoption in monetary terms in a consistent extended cost–benefit analysis framework. The complexity of all of this is compounded by the need to estimate impacts at different scales, local, landscape, national and global.

In this study, we have made most progress on valuation, through the application of choice modeling and contingent valuation methods. We have made less progress on the measurement, modeling and data collection issues. Developing our understanding of environmental production functions of new technologies has far-reaching implications for how research is conducted in the CGIAR, and for the metrics that are used to evaluate the suitability of a technology for development and dissemination. Traditionally, research has focused on the metrics of 'productivity', and experiments have been designed to study the relationship between technology and productivity. For 'sustainability of natural resources' to be made fully consistent as one of the CGIAR high-level goals, indicators of that goal have to be included in the technology (or policy) evaluation stage (as a metric). These evaluation studies can then serve as the model for estimating the 'environmental production functions', at least at plot or farm level, and for guiding environmental monitoring at higher spatial scales (i.e. landscape, national, global). Renkow's framework provides a good starting point for this challenging work.

SPIA recognizes that in addition to a greater and more coherent and coordinated financial investment in environmental monitoring, there are also issues in the CGIAR relating to expertise and specialization needed among impact assessment practitioners. Typically, the job of coordinating impact assessments at a Center is held by an agricultural economist, often working with colleagues, typically also economists. Ways are needed to support collaboration across the CGIAR system in order to improve practices. The learning and capacity building that was started in the process of implementing these case studies will need to be maintained and expanded across the system. In addition, there is a role for SPIA and the Independent Science and Partnership Council (ISPC) in stimulating methodological innovations in modeling of agriculture-environment linkages, through the creation of links with advanced research institutes outside the CGIAR. This is the focus of the forthcoming CGIAR Science Forum in Beijing in October 2011 on the topic of the 'Agriculture-Environment Nexus', featuring relevant sessions on land-use change, metrics and monitoring, and sustainability science.

SPIA is grateful to the case study leaders at the respective CGIAR Centers for their commitment to this study. The experts that SPIA hired over the course of three years have all added considerably to this final report, namely Jeff Bennett, Mitch Renkow, Nelson Villoria, and Göran Djurfeldt. In addition, several resource people helped make the Rome and Istanbul workshops useful and productive, valued by the case study leaders and by SPIA, namely John Dixon, Jeff Sayer, Bekele Shiferaw and Paul Vlek. We hope this study helps further the understanding of donors and Center scientists of this important agenda for the CGIAR.

Derek Byerlee, Mywish Maredia, Bhavani Shankar, Timothy Kelley, James Stevenson

CGIAR Standing Panel on Impact Assessment (SPIA)

September 2011

1. Assessing the environmental impacts of CGIAR research: toward an analytical framework

Mitch Renkow[1]

Abstract

Farming systems produce a range of food, fiber, fodder, forage, fuel and other products that generate economic impacts at different scales at the same time. Farming systems also generate environmental impacts in the form of changes to the natural environment. Agricultural research carried out by the CGIAR often aims to achieve economic impacts, but there are also environmental impacts associated with changes attributable to the adoption of research-derived agricultural technologies or policies. These may be positive or negative, intended or unintended, and may be felt on-farm, locally or globally. This chapter provides a framework for assessing the environmental impacts of agricultural research and reviews the existing evidence, finding only a small number of cases in which a partial analysis has been carried out, the notable exceptions being for timber harvesting policies on deforestation, and for pesticide reduction policies (although in the latter case, the focus was mainly on human health impacts). There has been no work carried out to date that has successfully traced the entire impact assessment pathway from research investment through to measurement of off-site biophysical effects on ecosystem services, and on to the ultimate economic impacts on agents located in receiving sites. The challenges in carrying out this kind of work are considerable and these are reviewed from the perspective of impacts on land, water, agrochemicals, livestock, biodiversity and climate change. A framework based on careful determination of biophysical measurement, integration across scales, attribution to research, valuation of non-market costs and benefits, and valid counterfactual development, is offered as basis for moving forward with improving practice on environmental impact assessment in the CGIAR.

1 Professor, Department of Agricultural and Resource Economics, North Carolina State University, Raleigh, North Carolina, USA. Email: renkow@ncsu.edu.

1.1 Introduction

During nearly forty years of 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 followed escalating demands on the part of donors and CGIAR managers, for evidence that specific research investments have generated 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) technologies far outstrips that for natural resource management (NRM) and policy research (Renkow and Byerlee, 2010). Also, whereas a large body of evidence documents and quantifies direct and indirect effects of CGIAR research using economic surplus approaches (e.g. Evenson and Gollin, 2003; Raitzer, 2003), very few studies quantify social impacts (on poverty and gender issues) or environmental impacts.

Ideally, a unified analytical approach would jointly consider impacts across all three dimensions – economic, social, environmental. Achieving this 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, being the market prices for traded goods and services whose existence can be attributed to research outputs. Combining price and quantity data makes economic impact assessment a relatively straightforward (although not always 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 unresolved valuation problems. 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 also renders social and environmental impact assessment a much more difficult task.

At the outset, it is useful to distinguish more clearly between economic, environmental, and social impacts, and Bennett (2008) is followed here. *Economic* impacts 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 refer to flows of goods and services that are to a large extent not traded (and thus not valued) in markets. Social impacts refer to public goods associated with changes in health, education, gender relations, and relative poverty (and more generally, the size distribution of income and wealth). Environmental impacts refer to public goods associated with ecosystem services in all their various forms. These can be as inputs into production processes, consumption goods that directly confer wellbeing (e.g. via enjoyment of ambient environmental quality), 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 1.1 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 refer to the full range of genetic and management activities associated with crop and 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, fuel and other products that generate economic impacts at different scales at the same time. Farming systems also generate environmental impacts in the form of changes to the natural environment, physical (e.g. of soil structure) and chemical (e.g. emissions, pesticide runoff). Both economic and environmental impacts are dynamic in that they feedback into the agricultural practices of 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 the environment jointly give rise to social impacts – again, both contemporaneously and over time. Although not shown in Figure 1.1, these social impacts will in many circumstances alter economic and environmental conditions (with some delay), 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 (see the upper part of the flow chart in Figure 1.1). The goal of this chapter 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 flow chart in Figure 1.1).

The chapter is structured as follows. It begins by reviewing existing evidence on the environmental outcomes associated with agricultural technologies developed by CGIAR research, before introducing a set 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 et al., 1991). But the key point here is that widespread markets for agricultural goods and services provide 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.

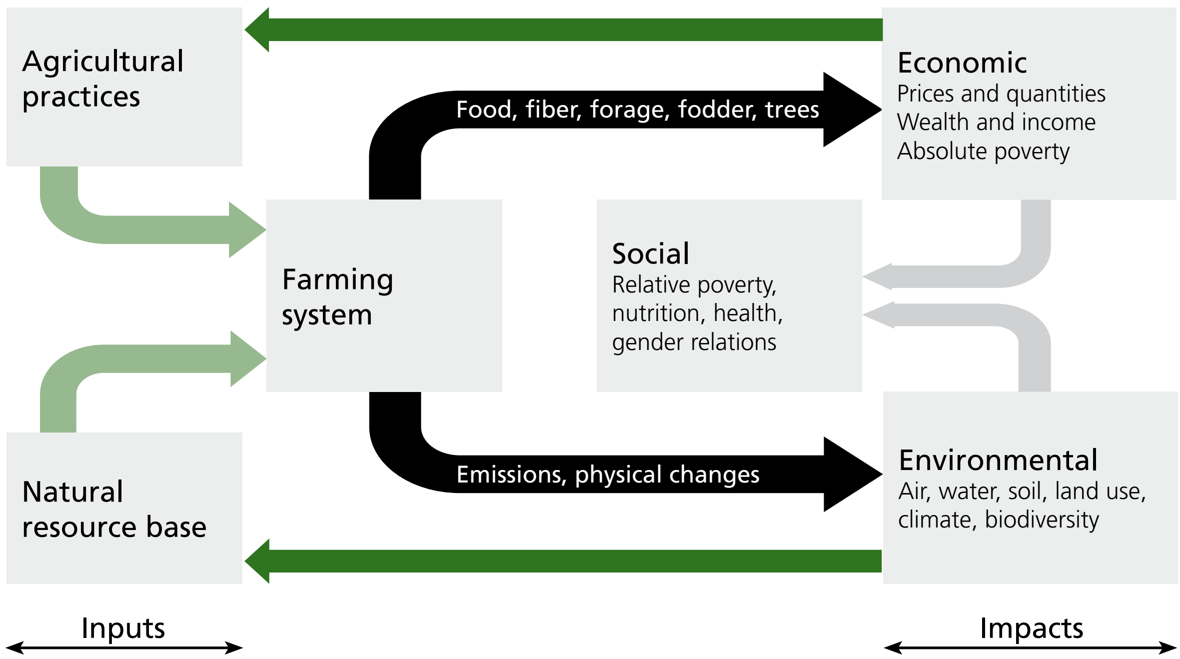

Figure 1.1. Economic, environmental, and social impacts of agriculture

definitions and concepts to establish a common vocabulary for use in the ensuing discussion. Next, a typology of environmental impacts is put forward that differentiates between the scales over which the impact is felt, and also the kind of agricultural system – intensive versus extensive – in which impacts occur. Elements that need to be addressed in order to adequately and meaningfully conduct the environmental impact assessment of various types of CGIAR research products are discussed, being biophysical measurement, scale, attribution, valuation and counterfactual development. Finally, observations are offered on steps that must be taken to facilitate environmental impact assessments becoming a more standard element of the CGIAR's self-evaluative activities.

1.2 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 true 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 and 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 1.1). Remarkably little has been done in the way of accurately tracing the entire chain of outputs, outcomes, and impacts of CGIAR research as it pertains to the natural environment. Moreover, the studies that have been conducted have tended to focus on positive outcomes, i.e. technologies or knowledge-based management regimes that redress some negative environmental externality. Importantly, no study could be found that directly tackles 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 companies. Those outputs produced *outcomes* that included changes in implementation of those policies by the Indonesian government due to pressure from various external watchdog groups like Friends of the Earth, World Wildlife Fund and other non-governmental organizations (NGOs) that 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.

Raitzer (2008) 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 measurements nor the modeling of biophysical effects were conducted, as the activity being evaluated was policy analysis rather than technology products.

Pesticide use

Two research programs that calculated the 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 et al., 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. Also, 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

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.

Table 1.1. CGIAR studies relevant to environmental impact assessments.

Center (Year)	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 = US$19 million to $583 million ($133 million under main set of assumptions) , compared to < $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; with minimal productivity impacts from reduced pesticide use • Only small negative impacts on ecosystem function • High rate of return to research on non-chemical pest control
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 leading to a tradeoff with health impacts • Little evidence that pesticide leaching poses a threat to 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 (wheat only) in India but not in Pakistan Positive: Reduced diesel consumption (~US$50 million annually) Positive: Reduced CO_2 emissions (~91 kg/ha) Negative: Air pollution due to burning non-basmati rice residues
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 and quality, local and 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 out-comes 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 environ-mental impact quotient (EIQ) (*ex-post*)	• Substantial variability found in EIQ across locations. • Lack of correlation between EIQ and productivity suggests opportunities for reduction in pesticide use via greater use efficiency and 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 impacts of CGIAR technology, management, or knowledge products.

Sources: 1. Raitzer (2008); 2. Pingali and Roger (1995), Templeton and Jamora (2007); 3. Crissman et al. (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).

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 US$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 cost–benefit analysis. Instead, they opted to present the results in the form of a 'tradeoff analysis', where 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 related to 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 scope of analysis and/or across different crops.

Agriculture and ecosystem services

The World Agroforestry Center was a central participant in the Trans-Vic research project, a multi-year, multi-institution activity that investigated agriculture-environment interactions in two watersheds in 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. This was 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 -reduced costs, explaining its spread in the region. Although this work mainly focused on agricultural profitability impacts, the research also documented environmental benefits coming from the wheat side of the system, being reduced diesel consumption (and the 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 component of the system generally was not 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 book (Shiferaw et al., 2005). Two case studies in that volume featured bioeconomic models of soil conservation technologies at the farm and village levels in Ethiopia (Shiferaw and Holden, 2005; Holden and Lofgren, 2005). These did not focus on measuring *ex-post* environmental impacts, nor did they examine technologies that were developed by ICRISAT. Rather, they were 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 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 (researchers, civil society organizations, government and NGOs), appears to have been mainly oriented toward documenting baseline conditions 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 the environmental impacts 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). It also found that substitution of fishpond sediments for inorganic nitrogenous fertilizers reduced fertilizer consumption by 50%. 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 to farmer organizations would improve management and make irrigated agriculture more productive and sustainable. However, although IWMI's self-assessment indicates that its efforts have made a substantial contribution to knowledge about the design and implementation of these programs, it falls well short of documenting *ex-post* whether or not the assumed environmental benefits of IMT were actually realized (Giordano, 2006).[4]

Greenhouse gas emissions

The global alternatives to slash-and-burn (ASB) program motivated research that investigated the net greenhouse gas (GHG) 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 program to assess the trade-offs between global environmental and private economic aspects of land-use systems in the humid tropics. The *ex-post* analysis indicated 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. Although 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) the trade-offs between agriculture and the environment.

Environmental indicators and monitoring systems

Two other Center-related research activities that deserve mention here are more oriented to monitoring environmental outcomes via development of indicators of environmental outcomes. CIP researchers studied the environmental and human

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

health risks associated with pesticide use through the development of an environmental impact quotient (EIQ). This 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, being 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 which the specific weights were imposed is unclear. The study's principle finding is that the correlation between EIQ and production outcomes is low, which 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 the 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 those involved in farming, forestry work, fishing and dairy production, 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 an 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. These are: IITA's work on biological control of insects and water hyacinth; the growing body of NRM research conducted within the CGIAR 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 in 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 also computed similarly large returns on investments in these programs (De Groote et al., 2003; Coulibaly et al., 2004).

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., 2005). 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 also likely to be large. This seems like a very promising direction for future impact assessment research.

NRM research represents a second general area of CGIAR activity where positive environmental impacts as yet have not been investigated. 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 without CGIAR crop genetic improvement activities, an additional 200 million hectares of cultivated land in developing countries would have been needed 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 the 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. Notably however, there has been no work to date that traces the entire impact assessment pathway from research investment through to the measurement of off-site biophysical effects on ecosystem services, and on to the ultimate economic impacts on agents located in 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.

Studies reviewed here do offer examples of analytical tools, however (including the following), that will need to be considered to satisfactorily pursue environmental impact assessments.

- The bioeconomic modeling work highlighted in Shiferaw et al. (2005).
- Discussions and implementation of environmental indicators found in Shiferaw et al. (2005), 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.
- Use of 'sentinel sites' for long-term monitoring of environmental impacts, developed under the aegis of the Alternatives to Slash and Burn program.
- Attention to quantifying trade-offs among various stakeholders whose actions impose negative externalities on one another, in IFPRI's Costa Rica work (Hazell et al., 2001).

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).

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, some of which should be constant in any meaningful statistical analysis, and that begin at a very early stage in the adoption/diffusion process. In addition, 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 also poses a distinct challenge. 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 interdisciplinary collaboration, the organization and administration of which can be challenging. The substantial field research required to pursue environmental impact assessment analysis is also costly, 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.

1.3 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 advance has been made in achieving that goal. Clearly, some intervening factors have severely constrained investigators' ability to pursue this line of inquiry.

In the next three sections, a set of issues are presented that need to be addressed in order to satisfactorily pursue meaningful assessment of the environmental impacts of CGIAR research. First, a set of definitions and concepts is introduced, to establish a common vocabulary for use in the discussion. Next, a typology of environmental impacts is offered that differentiates between the scales over which impacts are 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. Elements that need to be addressed in order to adequately and meaningfully conduct the environmental impact assessment of various types of CGIAR research products are then discussed. 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*, such as 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.

6 In order to begin bridging this gap, SPIA commissioned a set of six ex-post impact assessment case studies in 2009 to quantify the environmental impacts of specific Center research activities. These case studies encompassed 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 were scheduled to be finalized by the end of 2010, and the results are synthesized in chapter 2.

1.4 Environmental impact assessment: some definitions and concepts

Our concern here is with the impacts of agricultural activities on the natural resource base, in particular looking activities that are affected by the CGIAR's technology, management and knowledge products. 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 the purposes of this chapter, these types of on-site environmental impacts are referred to as 'production effects'. In general, our focus here is not 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.

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. For the purposes of this chapter, the term 'off-site environmental impacts' refers to alterations in the natural resource base that affect other (off-site) users of those natural resources. Correspondingly, the term 'environmental impact assessment' (EIA) is used here to encompass the suite of activities required to measure changes in off-site stocks and flows of environmental services accompanying the adoption of an agricultural innovation, and then to assign monetary values to those changes. Defined this way, EIAs account for the impacts on the natural resource base not already accounted for by standard *ex-post* economic impact assessment.[7]

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.

This dichotomy between off-site environmental impacts and on-site production effects is depicted in Figure 1.2, 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 Figure 1.2, 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 described elsewhere (e.g. 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 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 even at a small scale (e.g. farm-level). Furthermore, 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 since they are generally not traded in markets.

Three other general aspects of the assessment process merit mention here, before turning to a more detailed discussion of issues related to assessing environmental impacts of specific types of CGIAR research products. First, note that the depiction in Figure 1.2 of the pathway to environmental impact assessment explicitly includes extension (along with research) as an initiating input. Complementary investments in

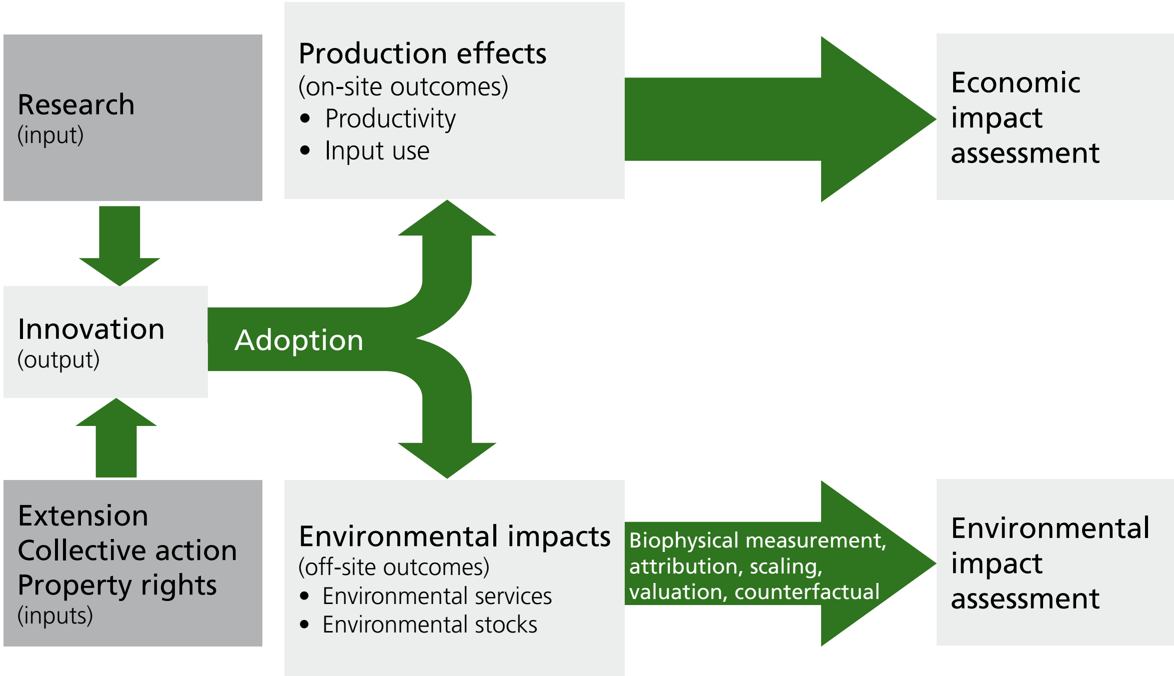

Figure 1.2. Pathway from research and extension to economic and environmental impact assessment

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 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. Agrochemical 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 positive environmental effects. Those resulting from a new technology or practice will in many cases be intended consequences of the research that generated it, so 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 are generally unintended outcomes. That negative outcomes tend not to be anticipated, which 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 and that can include the following.

- 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 runoff.

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.

- Non-agricultural producers affected by alterations to natural resources that are inputs into production process, e.g. fisherman whose livelihoods are affected by changes in waterways due to erosion-induced siltation.
- Agricultural producers, e.g. farmers whose livestock are negatively affected by pesticide residues, or whose costs of irrigation are increased due to groundwater depletion or siltation of canals.
- Non-local individuals for whom (non-use) options 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 also complicates the process of environmental impact assessment. 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.

1.5 Environmental impacts by source and type of agriculture

Table 1.2 presents a typology of environmental impacts of agriculture and the scale(s) over which those impacts are generally felt, i.e. on-site at the plot or farm level, locally at the village or watershed level, nationally or globally. The typology

Table 1.2. 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

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 in high rainfall areas that were most profoundly affected by new seed and fertilizer technologies that have traditionally been the mainstay of the CGIAR's commodity Centers, and the diffusion of which accelerated the intensification process in many of these areas. Environmental problems associated with intensive agricultural systems reflect the high demands that are placed upon the natural resource base by the intensification process. These include soil degradation due to continuous cropping, salinity problems and waterlogging associated with excessive and improperly administered irrigation, negative side effects from use of chemical inputs, and loss of *in situ* biodiversity.

Extensive agricultural systems on the other hand, tend to be found on land of lower agronomic potential, due to a variety of abiotic stresses, such as low or highly variable rainfall, fragile soils, limited fertility, etc. Increasing production in extensive systems often requires 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 agroforestry). 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 attendant implications for global climate change and loss of biodiversity. Other environmental problems in extensive system relate to cultivation of ecologically fragile 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 so 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 the 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 build-up in lakes and reservoirs, and increased risk of flooding (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).

An abundance of models have been developed and used to track soil and hydrologic dynamics. These tend to be applied on a short-term basis, however, and are best

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

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 combining techniques such as reflectance spectroscopy, remote sensing and use of satellite imagery are other 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 CGIAR crop genetic improvement is whether yield increases associated with improved varieties cause a reduction in land conversion as less land is needed to produce the same amount of food (the so-called 'Borlaug hypothesis'). Or, in contrast, does this lead to an expansion in the cultivated area as more farmers take 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 applications has contributed to over-pumping of groundwater and subsequent lowering of water tables.

Several salient points pertain to assessing the impacts and contribution of CGIAR technologies to groundwater depletion.[11] First, while there can be no doubt that modern varieties are a central to increased over-pumping, 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-exploitation of groundwater.

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 will greatly simplify valuation challenges. Or 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 such technologies need to carefully separate these on-site effects from off-site environmental impacts associated with reduced water withdrawals.

Finally, to the extent that on-site production effects of groundwater depletion persist over time, intergenerational impacts may well arise. On-site production effects felt by future resource users do represent an externality, and thus pose many of the same analytical challenges required to evaluate (spatial) external effects. In addition, considerations of inter-generational impacts require attention to the motives and preferences 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 in-

10 Note that even at the plot level, measuring soil loss is not straightforward. For example, the Universal Soil Loss Equationrequires 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.

creases in canopy interception and evapotranspiration (Jackson et al., 2007). The bulk of the impacts of such technologies, however, will likely take the form of on-site production effects.

Of potentially more interest are off-site environmental impacts associated with bringing cropland under irrigation that had previously been farmed using relatively low-input systems. Conversion of arid and semi-arid areas may lead to significant depletion of surface or groundwater resources if and when crop needs 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.

Agrochemicals

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 would appear to be highly variable at higher spatial scales, especially relative leaching potential and surface loss potential. Thus, model-based fate and transport studies of potential pollutants are a necessary complement to analyses of on-farm pesticide use (Ducrot et al., 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 on pesticide use and its impacts conducted by CIP and IRRI in the 1990s was preceded by earlier Center-endorsed recommendations that involved significant pesticide use. That the later research led to revisions of earlier recommendations is not unusual. Other factors, including large subsidies on chemical pesticides, overly aggressive promotion of chemical use, and lack of attention 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 roles of Centers in the overuse of pesticides that later research has helped to ameliorate.

Livestock

Livestock are a major contributor to global GHGs (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 literature on the 'new rangeland ecology' provides evidence that extensive livestock systems can have positive environmental impacts if managed appropriately, but crucially, only as long as land is sufficiently available to allow pastoralists to practice transhumance (Behnke et al., 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 local landraces.

Forest conversion due to expansion of the agricultural frontier is a primary source of biodiversity loss, particularly in the South America, Southeast Asia and West and Central Africa, but also in other hotspots around the world (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. However, it continues to be an area of inquiry for which there exists as many questions as answers when it comes to estimating the benefits of being able to address unforeseen future problems (Koo et al., 2004; Smale, 2006). 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. Research 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, and 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 such as the use of mechanical technologies that burn fossil fuels, the release of carbon into the atmosphere due to disturbance of soil carbon stocks, and the conversion of land (particularly forested land) to agricultural uses, with attendant declines in carbon sequestration.

Measurement of the physical contribution of specific practices to emissions of GHGs (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. There is frequent reference to using the price of carbon in fledgling carbon markets, such as the Chicago Climate Exchange 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. wetland regulations in the United States of America), 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 may have occurred without these innovations. In other words, the relevant comparisons may well include two undesirable outcomes. Also, 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.

1.6 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 agroecosystem assets and associated flows of ecosystem services. For example, the NRM impact assessment cited earlier (Shiferaw et al., 2005) 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 (Table 3). Whereas by no means a comprehensive listing of all indicators employed by natural scientists, the indicators in Table 1.3 give an indication of 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).

Table 1.3. Biophysical indicators.

Criteria	Indicators
1. Biodiversity	• Species richness • Species diversity • Species risk index
2. Agrobiodiversity	• Index of surface percentage of crops • Crop agrobiodiversity 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)

Other studies have developed integrated indicators of soil 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 the weighted averages of several sub-indicators, and 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

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 the assessment of biophysical impacts (Paul Vlek, Center for Development Research at the University of Bonn (ZEF-Bonn), personal Communication, 2010).

substantially, depending on the particular 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. These models were constructed using long-term data from multiple locations, and as such they 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 bioeconomic 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.

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, p108) provide 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.

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 the modeling of 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 assessments, and also add to projects' logistical and organizational complexity.

Third, the predictive efficiency of soil and hydrologic models declines substantially at large spatial scales of analysis. Extant models are generally better suited to analyzing on-site production effects (Roy et al., 2003), but 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 nonetheless reinforces the need for careful sampling and repeated field verification to be part of the process whereby specific (plot-level) results can be scaled up.

Finally, in addition to understanding the origin of environmental impacts, biophysical measurements need to be taken at receiving sites as well. Although it is fairly straightforward to measure arrivals of pollutants or other negative externalities at a particular location, establishing causality between those negative impacts 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

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 as a 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).

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). Individual Centers have an interest in impact assessment at lesser (farmer or 'meso') scales for the purposes of evaluating specific research products or programs, there is also a clear imperative at the System level to illuminate impacts that are large and 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 vary across different agroecological zones. Monitoring 'sentinel sites' over a period of time as in 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.

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 an investigation of land-use changes in the Kenyan highlands 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 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) developed 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 the responses of households to both economic and environmental variables. Berger (2001) developed an agent-based model that integrated economic and hydrologic components within a spatial framework to analyze potential impacts of water-saving irrigation methods in rural Chile. Le et al. (2010) developed an agent-based 'land-use dynamic simulator' for central Viet Nam in order to assess the co-evolution 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 also 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 also attempted to synthesize economic and environmental impacts at a geo-regional level. The SEAMLESS model developed by 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 agrienvironmental 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 trade-offs 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 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 complicated 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 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-irrigated areas (Pingali et al., 1997), and as a driver of salinity and 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 from the improved crop varieties 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 highlights the desirability of integrated models that synthesize biophysical relationships and behavioral responses. The bioeconomic modeling 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 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 indicators with respect to specific policy or socio-demographic variables.

Valuation

As was noted previously, a number of impact assessment studies conducted within

16 As a cautionary note, these models strike this author 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. Although this latter observation in no way impinges on the scientific integrity of modelbased predictions, it is nonetheless relevant to their ultimate influence on policy makers and research managers.

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

the CGIAR have stopped short of measuring environmental impacts precisely because of an inability to assign monetary values to non-market environmental services. This is particularly evident among impact assessments of NRM projects found in Waibel and Zilberman (2007). Cost–benefit studies of crop genetic improvement have also generally neglected incorporating (negative) environmental impacts for this same reason (Hazell, 2009).

As has been already noted, not all environmental services are non-market. For example, lost nutrients can be replaced in the form of inorganic fertilizers and other inputs, and remediation services for saline or waterlogged lands provide a benchmark for the costs of poor irrigation practices. Although not perfect substitutes for lost environmental services, these may nonetheless inform assessments 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) describes in detail a host of approaches for valuing non-market environmental goods and services. Whereas still not common in developing country settings, a growing number of studies use stated preference non-market valuation techniques. These include applications to include the valuing of tropical rainforest preservation (Rolfe et al, 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 types of valuation exercises will continue to proliferate.

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).

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 1.3 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 as compared to the traditional technology. Here, the relevant (plot-level) counterfactual lies in the distance between the two solid lines (and not between the baseline soil depth and that of the traditional technology).

The bottom portion of Figure 1.3 depicts two alternative outcomes at a larger (landscape-level) scale with regard to the 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 maintains the area under cultivation at the baseline level. In this case, the new technology produces environmental benefits at both the plot level and at the landscape level, relative to what would have been the case without any 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. However, 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 compared to 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.

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.

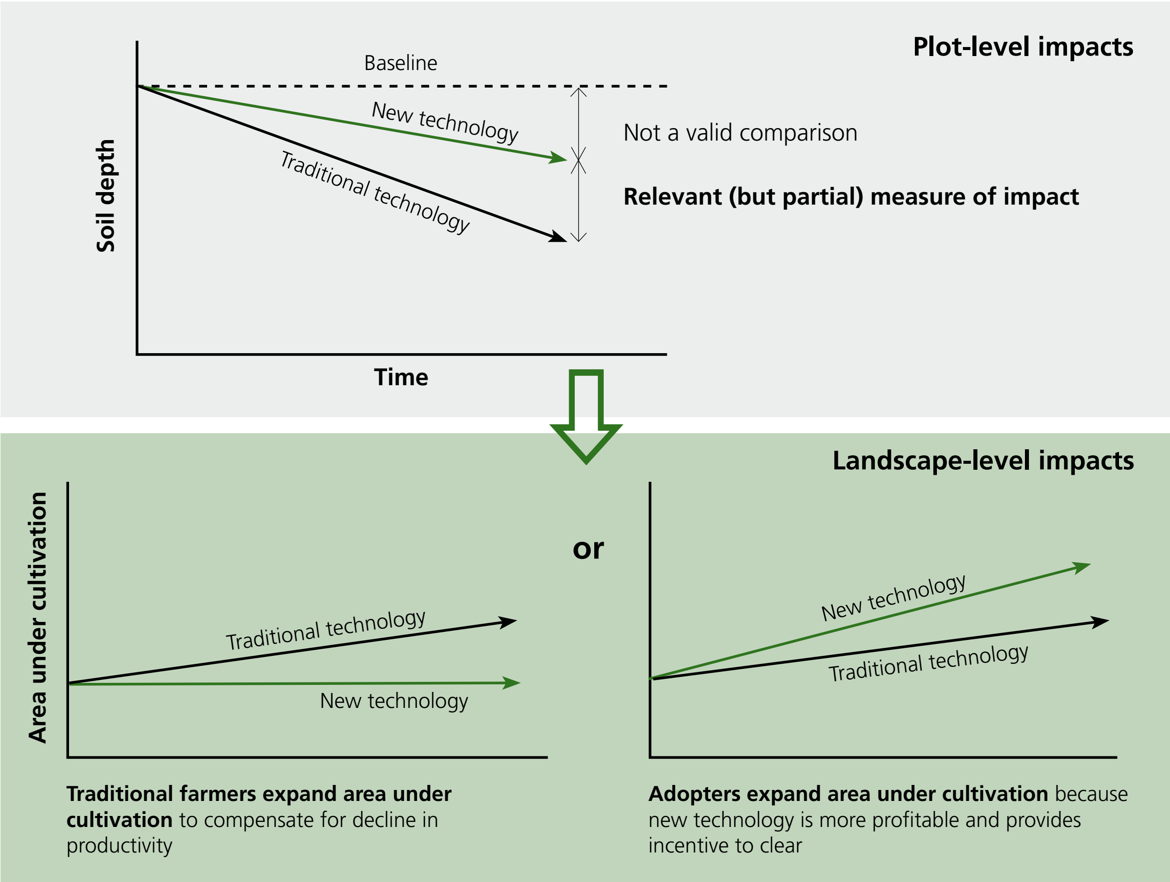

Figure 1.3. Appropriate counterfactuals at different scales

Whichever of these two landscape-scale outcomes is more realistic would be up to the analyst to decide, depending on the specific context. Of course, a host of other factors that might support or limit land expansion such as land and labor availability, would also need to be considered. 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 are worth repeating here. First, some future or potential outcomes are highly uncertain, especially those related to biodiversity loss and global climate change. 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 1.1 and Box 1.2 provide two examples of how the various issues discussed above might be approached in the context of specific environmental impact assessment case studies. These are zero tillage in the Indo-Gangetic Plain (Box 1.1) and biological control of water hyacinth in West Africa (Box 1.2). These are by no means intended to be definitive descriptions of the full suite of activities required to pursue environmental impact assessment, but they do illustrate the sorts of issues that arise and possible means of handling them.

Box 1.1. 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 of representative farms; and this information can 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 an area, 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 airflows. Seasonality of weather patterns would no doubt also be an important complicating factor.

Valuation: (1) The value of reduced carbon emissions can be imputed from prices on the Chicago Climate Exchange or other carbon markets 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 the 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 et al. (2007b) attributed CIMMYT's share of the economic gains from ZT by assuming that diffusion occurred more rapidly than would have been the case without CIMMYT's involvement , i.e. that it would have followed the same (logistic) adoption curve, but with a delay of five years. A similar strategy would appear appropriate for inferring CIMMYT's contribution to the net value of both the positive and negative environmental impacts.

Box 1.2. 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 people derived their livelihoods, primarily by hindering fishing and transport, sometimes interfering with water use for irrigation, drinking and electricity generation (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) estimated 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, fauna and flora. This requires assembling information on which chemicals were (or are likely to have been) used in different geographic locations; the amount of chemical use in each; the toxicity of the various chemicals to specific organisms; the spatial extent and duration of those toxic effects; and a 'census' of number of people, animals and plants 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 be 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 separate this 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 allocations of staff time (and associated salaries) would be needed.

Each potential case study would pose different challenges. In the case of zero tillage, biophysical measurement of reduction in GHGs 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 be a much more complex undertaking, given spatial variability in weather patterns. Likewise, valuation of GHG 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.

1.7 Key lessons and implications for moving forward

This chapter 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, which 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 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 all CGIAR Centers.

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 will generally 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 comparison 'before and after, 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. Principal off-site impacts associated with intensive

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 CGIAR System has a 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.

systems tend to reflect improper management of nutrients, agrochemicals 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 them. Regarding natural sciences, 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). Regarding economics, 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 levels).

In sum, the necessary tools exist for the 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 thus required for moving forward, is a substantial commitment of organizational, financial, and human resources to the process. Four imperatives stand out in this regard relating to the System-wide 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 are to be subject to this sort of evaluation. A sensible approach may be 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 tends 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 as compared to other types of CGIAR research products (Renkow and Byerlee, 2010), with 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 *ex-post* EIA studies include: (a) the negative impacts due to increased use of mono-cropping and agrochemicals of Green Revolution technologies; and (b) the positive impacts of biological control of harmful pests.

22 That said, there may be significant payoffs to EIAs of smaller-scale NRM projects, such as is being pursued in the 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.

This approach to prioritization also has implications for deciding whether or not to incorporate an EIA component into the research design of new projects. One clear theme that has been emphasized throughout this chapter 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 project 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. Also, given that environmental outcomes typically unfold slowly, measurements will need to be taken over an extended period of time. Likewise, tackling the valuation problem will require considerable advanced 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 useful primarily 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 locations and at different points in time. Incorporating environmental impact assessment as a standard component of project design is also likely to 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 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 CGIAR's core mission of enhancing agricultural productivity, and evaluation of those productivity impacts.

Finally, some changes in the human capital base on which the CGIAR draws would appear warranted. Existing manpower at certain Centers may not have the expertise to pursue some of the analytical tasks that need to be undertaken. For example, most economists within the CGIAR have considerably more expertise in areas of agricultural production and marketing than they do in environmental economics issues (and in particular, non-market valuation). Whereas some re-tooling might be feasible, augmenting existing staff resources to include environmental economists would seem to be inevitable. Alternatively, there is scope for partnering with institutions and individuals outside of the CGIAR that have a comparative advantage in research on environmental issues. Such 'outsourcing' of research tasks is not unprecedented, and may in many circumstances represent a more efficient approach in the pursuit of a System-wide research agenda in which environmental issues are more prominent.

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2. Advancing *ex-post* impact assessment of environmental impacts of CGIAR research: conceptual issues, applications and the way forward

Jeff Bennett[23]

Abstract

Four case studies of the non-market valuation of environmental impacts of agricultural research are reported, drawing on the methods outlined in chapter 1. The International Centre for Agricultural Research in the Dry Areas (ICARDA) estimated the impact of supplemental irrigation in wheat growing areas of Syria on the extent of groundwater depletion and soil salinity accumulation. The estimated benefits across the wheat growing areas of Syria ranged from zero to more than 423 million Syrian Pounds (SYP) (US$8.5 million) per annum. This variability is attributable to different assumptions about the fate of the water 'saved' under supplemental irrigation. The Indian Council for Agricultural Research (ICAR) valued environmental benefits from the introduction of zero tillage (ZT) in rice-wheat systems in the Indo-Gangetic Plains, valued at US$142 per person in the region when presented to them as a 'water saving strategy'. The International Water Management Institute (IWMI) examined the impact on flora and fauna of a change in sluice gate operations in the Mekong delta, implemented by the Vietnamese government following research carried out by IWMI. On average, individual households living in the region were willing to pay between US$39 and US$73 per annum for this change in operations, and that between 12% and 15% of this is attributable to favorable changes in flora and fauna, yielding an aggregate value of up to US$200,000 per annum. The International Potato Center (CIP) case study tracked changes in potato diversity over time in Peru, using choice modeling to explore the trade-offs farmers make between modern and native varieties. The full analysis of this data remains outstanding, but the potential extent to which farmers are willing to trade-off improved yield with reduced biodiversity is indicated. These cases demonstrate that tools exist to enable economic valuation of non-market costs and benefits of the impacts from agricultural research, but that a number of challenges remain. These relate primarily to expertise gaps within the CGIAR Centers and to the financial costs of getting accurate data. Lessons for mainstreaming and resourcing environmental impact assessment in the CGIAR are outlined.

2.1 Introduction

The Consultative Group on International Agricultural Research (CGIAR) has successfully generated improvements in the economic well-being of people, well quantified in a wide range of *ex-post* impact assessment studies. In contrast, relatively little attention has been given to quantifying the impacts of CGIAR research on the environment. Environmental impacts can occur on-farm, when natural resource inputs into agricultural production are altered as a result of the implementation of research findings. However, impacts can also arise if the environment affected by agriculture is used by people away from the farm where the research findings are enacted. On-farm and off-farm environmental impacts of CGIAR research may be beneficial to people. However, unintended harmful effects may also arise.

This gap in understanding the full range of impacts arising from CGIAR research presents an important challenge. Growing scientific and public recognition of the significance of environmental impacts of economic development in general, and agricultural research specifically, both positive and negative, necessitates their integration into the assessment process. Such integration will allow for a more complete appreciation of the role played by CGIAR research in the past, as well as providing better di-

23 Professor, Crawford School of Economics and Government, The Australian National University, Canberra ACT 0200, Australia. Email: jeff.bennett@anu.edu.au.

rection for future research investments. This is particularly pertinent given that the strategic objectives of the CGIAR include environmental sustainability, alongside poverty and food security. Working toward this will involve research specifically aimed at enhancing environmental conditions, as well as ensuring that the environmental impacts of productivity-focused research are recognized, whether they are positive or negative.

To address the challenge of incorporating environmental effects into *ex-post* impact assessments (epIA), the Standing Panel on Impact Assessment (SPIA) of the CGIAR used a two phase approach. Initially, two scoping studies were commissioned to provide a framework and give direction to the analysis. The first reviewed a range of quantitative and qualitative approaches to measuring environmental impacts from the perspective of non-economic social science and natural science disciplines (Djurfeldt et al., 2009). The second drew on economics as its disciplinary base to review the conceptual and methodological elements necessary to integrate environmental impacts into the benefit–cost analysis (BCA) framework for investment appraisal (Bennett, 2009). The BCA framework was argued to be superior to other integrative frameworks such as multi-criteria analysis and single element assessment tools such as 'ecological footprint' and 'food miles', because the alternatives lack the breadth of coverage and rigorous conceptual framework that is provided to BCA through its welfare economics base.

In the second phase, the BCA conceptual framework and its associated techniques were further developed, and a comprehensive review was produced, of the studies undertaken within the CGIAR Centers that have addressed environmental impacts (Renkow, 2010). The framework and techniques were then applied in a sequence of six studies commissioned by CGIAR Centers. These demonstrated the process of estimating environmental benefits and costs for inclusion in research investments epIAs. They focused on a wide range of environmental impacts including genetic diversity of potatoes in Peru, flora and fauna diversity in the Mekong River Delta, water resource quality and quantity in Syria and the Indo-Gangetic Plains, forest biodiversity in Indonesia, and ecosystem services in Uganda.

The purpose of this chapter is to provide a broad overview of some of the conceptual issues faced in the incorporation of environmental impacts into the epIA of CGIAR research investments, to review the impacts case studies conducted by four CGIAR Centers, and to consider the lessons yielded by those case studies. The chapter is therefore designed to be a point of reference for the broadening of CGIAR epIAs, so that environmental sustainability objectives can be specifically recognized in the organizational assessment and planning processes.

2.2 Conceptual issues

A key limitation to the CGIAR's ability to extend its epIA effort to include environmental impacts is the complexity of the task. Part of that involves a range of conceptual issues that are inherent to the process of integrating environmental impacts into the BCA framework. These are not unique to the evaluation of agricultural research however, but their resolution in developing country contexts does involve additional challenges for practitioners.

To understand the conceptual issues, it is firstly useful to set out the flow of inputs, outputs and outcomes involved in the integrated EIA process (Figure 2.1).

CGIAR research investments produce outputs that are focused either at the farm level or the 'macro' decision making level. For both types of research outputs, the resulting outcomes are generally experienced at the farm level. A two stage process is usually involved. First, the research effort delivers technologies or policies for adoption. Subsequently, the adoption of the technologies or policies causes impacts. These impacts are either experienced directly on-farm, or at the farm level via impacts on the environment. Whereas the 'environment' has been the primary focus of some CGIAR research projects (notably in the NRM stream), most evaluation work has been directed at on-farm outcomes through increases in productivity, and hence changes in the surpluses enjoyed by producers and consumers. This is the

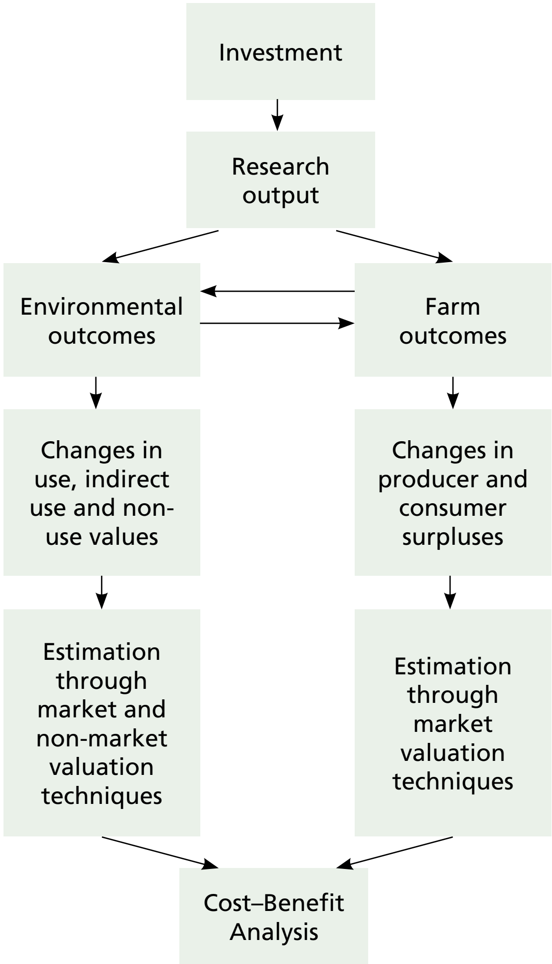

Figure 2.1. Integrating research inputs, outputs and outcomes

province of conventional epIA work where research-generates improvements in consumer and producer surpluses are compared against costs of research and its implementation.

However, environmental impacts are not limited to on-farm productivity. Research outputs that are directed at changing farming practices can have indirect impacts on the off-farm environment. For example, the introduction of zero tillage may reduce soil erosion and improve water quality for human consumption and ecosystem health in areas downstream from the farm.

Similarly, research that is targeted at the environment with the intention of improving farm productivity, may also have indirect impacts on people independent of farm operations or products. For example, measures to combat acidic soils may increase crop yields, but they may also bring about increased biodiversity in associated water bodies, or increased forest clearance as the profitability of agriculture is enhanced. Where endangered species are involved, the benefits of biodiversity enhancement or loss may be enjoyed or suffered by people who live far from the affected area.

It is also worth noting that such environmental benefits may be enjoyed by the local farming community too. Farmers are unlikely to be driven only by profits. The condition of their local environment may well improve their well-being. Those environmental conditions may involve factors such as the level of air and water pollution (including toxicity from farm chemicals) that may have impacts on their health as well as the aesthetics of their farm and local area. The values held by people for such non-farm environmental impacts are particularly challenging regarding their integration with the more conventional farm profit based BCAs of research investments. Specific challenges relate to both the biophysical and the valuation components of the EIA task.

Consider first the nature of the biophysical component. Here, the analyst must observe the environmental asset that is being impacted over time. In addition, a prediction of the condition of the asset must be generated under the counterfactual circumstance. The prediction is necessary as the counterfactual cannot be observed once the research induced change has occurred.[24] Both of these tasks require the impact of research to be teased apart from the impacts of all other factors. This necessitates a sound understanding of the full range of factors that influence environmental conditions, preferably in the format of a formal quantitative model. As there are frequently considerable delays in the occurrence of environmental impacts following the precipitating action, impact assessment

24 If the research has only caused a change in some geographical regions, then predictions of the counterfactual can be based on the observations from the unaffected areas. Comparability between affected and unaffected areas is required to ensure that only the research-induced changes are attributed.

that is undertaken *ex-post* of the research may involve environmental impacts that are yet to occur. Hence, even monitoring the condition of the environment to establish the 'with project' case may not be sufficient, with predictions of future condition being required.[25]

The complexities and data inadequacies implicit in the biophysical prediction stage of environmental impact assessment are such that techniques have been designed to facilitate the process. Foremost of these has been the development of indicators that aim to represent environmental impacts at a generalized level (Djurfeldt et al., 2008). Expert opinion can be used to develop these indicators either informally or through formal processes such as Bayesian Networks.

Establishing the links between research outputs and environmental outcomes is thus a key conceptual and empirical challenge. There are also large data requirements to establish the cause-effect relationships involved, as uncertainties abound. Attributing outcomes to the research input is complicated because so many factors play a role in determining environmental conditions. In addition, determining the geographical extent of the impacts can pose difficulties. Frequently, impacts are diffuse and can spread over wide areas and extrapolating the results derived from a small scale trial to full implementation may not be a simple linear process. Direct effects of research on the environment may yield more and more indirect effects as the scope of implementation increases over both time and space. Linear relationships between research effort and environmental outcomes are unlikely.

Knowledge of the measured and predicted[26] impacts of research investments on the biophysical environment is a necessary component of impact assessment but not sufficient on its own. In order to facilitate decision making regarding the trade-offs involved between costs of research investments and environmental (and other) impacts achieved, biophysical changes need to be converted into societal values, being the values that people place on environmental impacts.

A number of value frameworks have been developed to categorize environmental impacts. Importantly, these value frameworks are founded in the philosophical base of welfare economics. These are that the relevant values held by people ('anthropocentric') are interpreted in terms of the impacts on people's well-being (i.e. they are 'utilitarian'). Values that are so defined are then categorized by environmental economics into use, indirect use and non-use values. Use values involve people having direct contact with environmental assets, and include recreational and aesthetic use. Indirect uses include the provision of food and fiber and many of the 'ecosystem services' such as climate regulation, water supply and flood control. The non-use values relate to the enjoyment people have for knowing that species continue to exist and for protecting environmental assets for future generations. The Millennium Ecosystem Assessment (2005) assigns these values to different but readily reconcilable categories.

1. Supporting (nutrient cycling, soil formation)
2. Provisioning (food, fresh water, wood, fuel, fiber)
3. Regulating (climate, flood, disease, water purification)
4. Cultural (aesthetic, spiritual, recreation)

Supporting and regulating functions provide indirect use values, whereas provisioning and some cultural functions (e.g. aesthetics and recreation) generate use values. The spiritual part of the cultural function can be regarded as providing non-use values.

Importantly for valuation purposes, these values are either on-farm or off-farm and either marketed or non-marketed. Whether values are on-farm or off-farm determines the focus of the valuation task. Whether they are marketed or non-marketed determines the type of valuation techniques that are best suited to the task.

25 Conventional impact assessment that focuses on economic surplus changes resulting from research-induced changes also faces such issues. Costs and benefits to farmers and producers can be time-delayed and indirect.

26 Preferably with rigorous 'ground-truthing' of the predictions.

For example, the impacts of a research project that causes soil biota enhancement may be on food production. These impacts, caused by an improvement in supporting and provisioning the function of the environment would be use values on-farm (improved food crops provide profits to farmers) and marketed (farm production is bought and sold in markets). Another research investment managing wetlands in river systems may have an impact on the regulating function of the environment. Water quality downstream may be enhanced, generating benefits to people off-farm, and these benefits may not be marketed because the water involved is not bought and sold. If wetland species such as a rare migratory bird is protected because of the research outputs, a cultural, non-use value that is not marketed may be enjoyed by people living far from the research impacted site.

Market valuation, while far from simple in the context of environmental impacts in developing countries, has a well-established conceptual and application base. Surpluses to both producers and consumers of marketed good are estimated with reference to supply and demand information collected from market observations. Where the relevant markets are distorted by government interventions (both domestically and internationally through trade restrictions), 'shadow' prices (i.e. those that reflect the true economic value of inputs and outputs in the absence of policy distortions and market failures) may need to be calculated as the basis for valuations.

The values of non-marketed environmental goods and services are more challenging to estimate. Bennett (2009) provides an overview of the techniques available. They fall into two categories, revealed and stated preference techniques. Revealed preference techniques rely on specific relationships existing between the non-marketed environmental goods of interest, and marketed goods.

The hedonic pricing technique is a revealed preference technique that relies on the estimation of the relationship between the price of marketed goods (for example farming land) and the various characteristics of those goods, including non-marketed environmental attributes (such as soil quality, access to water, proximity to areas of timber for shelter, etc.). The impacts that changes in environmental attributes have on the marketed price of goods can be used to infer a monetary value for the non-marketed goods.

The production function technique also uses a relationship between marketed goods and non-marketed environmental attributes, but in production space. For instance, the impact of increased water availability resulting from a research output on a crop output could be estimated through the analysis of the production function that relates inputs (including water) to crop outputs. With the production effect estimated, multiplication by the price received for the crop output, yields the value of the additional water.

The travel cost method is widely used to estimate values associated with recreation and tourism visits to non-marketed environmental sites. Its application involves the estimation of the relationship between the frequency of visits to a site and the costs incurred by visitors. This relationship enables the simulation of the demand function for the site and hence the calculation of the surplus enjoyed by visitors.

All the revealed preference techniques rely on observing people's behavior and understanding the relationships between their actions in markets and environmental conditions. Where no such relationships exist or where the related markets are thinly traded, distorted by interventions, or do not offer sufficient data variation, stated preference studies may be useful sources of preference information. These techniques involve survey respondents being asked questions designed to quantify the strength of their preferences for the environmental goods at hand. Primary challenges in the application of stated preference techniques relate to respondents providing biased answers. In developing countries where monetary transactions are rare and where literacy levels are low, additional barriers to applying these techniques exist. There is also an issue associated with the extrapolation of results generated in surveys to the whole population of people affected by the change. Defining the scope of eventual impacts and the proportion of that population who hold non-market environmental values are complex tasks.

The contingent valuation method (CVM) is the most widely used stated preference technique. Survey respondents are asked if they would be willing to pay to secure a defined environmental improvement or to avoid an environmental harm. The question of payment can be framed in terms of voting in a referendum that would see taxes increased to secure an environmental gain. The proportion of respondents agreeing to pay varying amounts provides data sufficient to infer the average willingness to pay, which is interpreted as an estimate of benefit enjoyed.

A more recently developed stated preference method is choice modeling (CM), otherwise known as choice experiments. While similar to CVM in that survey respondents are provided information regarding hypothetical scenarios, CM involves asking multiple questions that present a number of alternative future environmental outcomes, each with differing associated costs. Outcomes are described in terms of a number of environmental attributes. Peoples' choices across the multiple questions provide information as to the trade-offs they are prepared to make across the outcomes. Given that one attribute is a cost, a monetary estimate of the unit values of each environmental attribute can be calculated from the choice data.

Despite these biophysical prediction and valuation hurdles, the environmental impact assessment framework that incorporates the concepts of benefit–cost analysis provides a sound platform for the analysis of the trade-offs that are inherent in resource allocation decisions, including investments in agricultural research. By estimating the societal values associated with the environmental changes brought about because of a research investment, they can be weighed up against the economic consequences of that investment. Thus, increases in farm income and the consequential improvements in well-being including hunger alleviation, can be weighed up against any associated environmental harm.

This approach to assessing trade-offs, because of its roots in welfare economics, takes an anthropocentric, utilitarian philosophical stance. It considers changes only valuable insofar as they affect people. It allows for trade-offs between environmental and economic outcomes, and does not either priority over the other, be they either economic (for example, raising farmers' incomes) or environmental (for example, preventing the extinction of a species). It does not give special consideration to the use of specific resources (such as 'virtual water' or 'carbon footprint' indices). Nor does it necessarily provide for equity or distributional concerns whereby values for specific groups of people within the current generation or across generations are considered to be of greater significance than others.[27] Hence, the conceptual framework proposed should be regarded as an information input into the assessment process rather than the determinant of a strict judgment.

2.3 Center case studies

Implementing the environmental impact assessment conceptual framework and the various techniques for estimating environmental impacts in monetary terms so that they can be incorporated into extended environmental cost–benefit analysis outlined in the previous sections, brings with it particular challenges. These challenges are in the application of both biophysical science and economics. They are particularly acute in the context of developing country applications. Case studies were commissioned from a number of CGIAR Centers in order to test the applicability of the concepts to specific research initiatives. In this section, four of these studies are reviewed (Table 2.1).[28]

27 Some initial experimentation has been conducted using choice modelling to estimate the relative importance people give to impacts that affect different groups in society (poor vs rich) and across time (current vs future generations). These experiments are detailed in Bennett (2009).

28 The four studies reviewed here are the most comprehensive and complete of those commissioned.

Table 2.1. CGIAR Center environmental impact assessment case studies.

Research focus	Center	Impact	Scope	Valuation method
Supplemental irrigation of wheat in Syria	ICARDA	Water Soil	On-farm	Production function Choice modeling
Zero tillage in rice-wheat systems in South Asia	ICAR/CIMMYT	Water GHG	On-farm Off-farm	Contingent valuation
Sluice gate management in the Mekong delta	IWMI	Flora Fauna	On-farm	Contingent valuation
Potato genetic diversity in Peru	CIP	Biodiversity	On-farm	Choice modeling

2.3.1 Supplemental irrigation of wheat in Syria[29]

Irrigation practices for wheat crops in Syria have been the subject of investments by the International Centre for Agricultural Research in Dry Areas (ICARDA) from 1986 to 1990. New supplementary irrigation (SI) technologies were developed and extended, involving changes in when, how and how much to irrigate. Existing studies (e.g. El-Shater, 2009) have demonstrated the relationship between SI, wheat yields and consequential economic impacts. However, broader environmental impacts on the amount of water available and the level of soil salinity had not been analyzed. The focus of the ICARDA case study was therefore to consider the relationship between the introduction of SI on the extent of groundwater depletion and soil salinity accumulation, and to estimate the values associated with these changes relative to a counterfactual defined by traditional irrigation practices. It represents an attempt to quantify the on-farm benefits arising from research that was focused at changing farm management, to impact on environmental assets (soil and water) on which farm viability is based.

The biophysical relationships between the introduction of SI and the availability of water and the extent of soil salinity were obtained from previous research. In addition, regression analysis was used to estimate a relationship between the quantity and quality (in terms of salt content) of irrigation water applied and the levels of soil salinity.

29 See Yigezu et al. (2010)

Data for the ICARDA study were collected in Februrary–March 2010 from a sample of 513 farm households across three governorates in Syria, and from a further 78 households in July 2010. First, the data were used to estimate a production function to understand the relationship between water inputs and wheat yield. This was then used to infer the marginal product of water in wheat production, and then, by multiplying the marginal product of water by the price of wheat, the marginal value product of water was estimated.

Second, to estimate the value of lowering salinity levels on farm, a choice modeling approach was adopted. Surveyed households were presented with a range of alternative parcels of land, and asked to decide if they would be willing to buy them. A ranking of the options was also requested. Each parcel was described in terms of its physical characteristics (including salinity level) and its cost.

Soil salinity was found to be affected by the amount of irrigation water applied. Furthermore, the hypothetical choices made by farmers regarding land purchases demonstrated a willingness to pay for low salinity land compared to high salinity land, amounting to 2.5 million SYP/ha (US$50,000/ha). This estimate was checked with reference to the production function estimation that showed a negative relationship between salinity levels and yield, converted to a value of 5,116 SYP/ha (US$102/ha). An explanation of the disparity between the results of the two valuation approaches was sought by the study in the differences between the quality of the land considered in the two applications, and the use of the land for crops in addition to wheat. However, the extent of the divergence suggests that value estimation issues may also have been involved. For instance, the two studies may have been directed at estimating different features, or biased results were generated from the stated preference application.

The value of water in wheat production was also estimated using the production function approach. The marginal value product was estimated at 6.14 SYP/m^3 (US$0.12/$m^3$). This estimate was then used to calculate the total value of water savings under alternative scenarios of water use:

1. all water 'saved' from the change to SI percolates back to the same aquifer, resulting in no water savings;
2. all water 'saved' from the change is lost to low quality aquifers, to evaporation or to weed transpiration; and,
3. part of the water saving is lost due to evaporation or weed transpiration.

The range of total estimated values was from zero to more than 423 million SYP (US$8.46 million) per annum across the wheat growing areas of Syria.

The ICARDA study concludes that the estimated environmental benefits supplement the economic benefits of yield improvements and pumping cost savings to offset the costs of change such as the capital costs of sprinkler system installation.

2.3.2 Zero tillage in rice-wheat systems in South Asia[30]

A related study carried out by the Indian Council for Agricultural Research (ICAR) and the Indian Agricultural Research Institute (IARI) considered the environmental impacts of the introduction of zero tillage (ZT) in rice-wheat systems in the Indo-Gangetic Plains. The two types of impacts assessed were the change to water availability and the reductions in GHG emissions. ZT has resulted in the elimination of pre-planting irrigation and reductions in the volumes of water applied to the crops after sowing. In addition, with less cultivation necessary under ZT compared to conventional tillage (which is taken to be the counter factual for the analysis), less diesel is used and fewer GHGs are emitted.

Like the ICARDA case study, this analysis considered environmental impacts on farm profitability that were caused by farm management changes, but it also considered off-farm environmental impacts in the form of reduced climate change risks resulting from reductions in GHG emissions.

Previous experimental studies have found that ZT has the potential to reduce water use by up to 36% on average. That amounts to between 140 m^3/ha and 340 m^3/ha depending on the use of associated technologies. The impacts on the groundwater source of this water saving will depend on farmers' decisions, individually and as a group. Farmers may choose to use the water 'savings' to irrigate summer pulse crops, or they may choose to allow the groundwater level to rise by not pumping as much water.

GHG emission reductions through lower tractor use have been shown to range from 52.4 kg/ha of CO_2 to 62.9 kg/ha. In addition, ZT involves the sequestration of carbon in the soil through stubble retention. However the ICAR study did not pursue this element of environmental impact to the valuation stage.

The valuation stage of the ICAR impact assessment involved the application of the contingent valuation method to the estimation of water saving benefits. A survey of 66 farmers asked if they would be willing to pay a stated amount to promote a 'water saving strategy' were also asked the maximum amount they were prepared to pay. The average willingness to pay was Rs 7100 (US$142).[31] A binary logit model was estimated to determine the factors that affected respondents' decisions to agree to make a payment. However, hypothesized independent variables were found to be not significant determinants. The concern is therefore that the willingness to pay values estimated were not well grounded by a

30 See Pal et al. (2010)

31 A weakness of this study was that respondents were not given a clear indication of just how much water would be 'saved' under the strategy for which they were asked to pay.

strong definition of what the water saving technology would deliver.

The conclusion drawn by the ICAR study is that the water 'saving' benefits from ZT act to supplement the yield increases the technology has been shown to generate, and further justify the research investments that have led to its more widespread use in the Indo-Gangetic plains.

2.3.3 Sluice gate management in the Mekong delta[32]

The International Water Management Institute (IWMI) carried out a research project at the request of the Government of Viet Nam that investigated ways of managing sluice gates in the Mekong River delta. This action was precipitated by conflict between shrimp farmers and rice growers over the impacts of the sluice gates on the relative flows of saline and fresh water. While fresh water flows are needed for rice growing, saltwater inflows are necessary for shrimp cultivation. Vulnerability to acidification of water through exposure of acid sulphate soils was also a concern to farmers.

The IWMI research developed a management plan for the sluice gates that provided an acceptable compromise between rice and shrimp cultivation, and allowed increased yields from both enterprise types. It also had impacts on the flora and fauna dependent on the waterways of the Mekong delta through improvements in the quality of the water in the system. The focus of the IWMI case study on an environmental impact assessment was on the net benefits of these flora and fauna changes for the local people of Bac Lieu Province. As such, its focus was on a research investment targeted at improving the operating environment for agriculture that had impacts beyond farm productivity for the local farm population.

The biophysical element of the EIA for the sluice gates had already been undertaken as part of the original research project. As that research was directed at predicting the environmental outcomes of alternative management scenarios, it was able to predict 'with and without' conditions in terms of water salinity levels. However, it was not able to predict the consequences for flora and fauna of those alternative salinity conditions. To do that, the IWMI study undertook a participatory rural appraisal (PRA) in which a sample of families were asked to provide their assessments of changes in soil acidity, water salinity and flora and fauna. The latter questions were specifically asked to determine the species that local residents deemed to be important and which had been altered with the changed operation of the sluice gates.[33]

The second phase of the data collection exercise involved a sample of 120 households who were again asked for their perceptions of environmental change, but were also asked for their willingness to pay for the continued operation of the sluice gates. This amounts to the application of the 'open-ended' version of the contingent valuation technique. In answering this question, respondents were expected to incorporate their values for both the economic net benefits and the environmental net benefits of the sluice gate operations. To separate these two types of values, respondents were also asked to apportion their willingness to pay between economic and environmental net benefits.

The study found that on average, respondent households were willing to pay between US$39 and US$73 per annum across the case study area. The proportion of this sum apportioned to flora and fauna changes ranged from 12% to 15%. This implies a willingness to pay for the flora and fauna changes of between US$5.56 and US$9.68, representing 0.3% and 0.4% of annual household income, respectively.

The conclusions drawn from this IWMI study are that these estimates of value are plausible and that extrapolated across the population of the area affected by the

32 See Wichelns et al. (2010).

33 Constructing the counterfactual in this case is particularly challenging. The understanding of local people regarding the impacts of the sluice gates may be limited, independent of other factors. Furthermore, the initial construction of the gates may have been a more significant influence on populations of plants and animals that would have taken several years to be fully felt. The approach taken to developing the 'with' and 'without' contexts was therefore one of necessity in the data-poor conditions.

sluice gates' operation, yield an aggregate value for the environmental improvements of up to US$200,000 per annum. These benefits are deemed to be in excess of the likely costs of sluice gate operation, so that the environmental benefits alone justify the operating costs. These net benefits would be relevant to a BCA of the research investment made by IWMI and its partners.

2.3.4 Potato genetic diversity in Peru[34]

The potato breeding efforts of CIP have generated improvements in the profit for farmers in the Andes. However, as higher yielding and more disease resistant varieties have been introduced, they have displaced native landraces and have so caused a reduction in local varietal biodiversity of potato species. This is a potential example of a CGIAR research investment causing an unintended negative consequence. The foregone value of genetic diversity may be a direct cost to farmers in terms of a reduction in the overall resilience of crops to future disease outbreaks.[35] Without diversity, crop losses may be greater in the event of a disease outbreak that affects the predominant high yielding variety. Farmers may also suffer a loss of cultural values associated with the continued availability of a wide range of varieties. Part of their culture may center on the availability of a range of varieties. Loss of diversity may also represent a cost to the wider community in the form of lost 'existence benefits' associated with biodiversity. Distant consumers and processors may also suffer reduced surpluses from the consumption and production of potatoes and potato products made from alternative (lost) varieties.

The loss to farmers of genetic diversity was explored by the CIP environmental impact assessment case study. This study was an attempt to explore the effects of research on farmer behavior that impacted on the environment (the genetic stock), and that in turn had a 'feedback loop' to farmers both in terms of profitability and non-marketed, cultural impacts.

34 See Hareau et al. (2010).

35 These costs are taken into account by farmers when they choose between alternative planting strategies. They trade off the benefits of yield improvement, against the risks of disease and other costs of planting only 'improved' varieties.

The biophysical element of the EIA was carried out using measures of biodiversity from the species to the genetic level. Measures were identified in terms of both displacement and loss using farmer surveys in two districts in the Peruvian Andes. First, an inventory of current diversity was carried out. This was followed by a 'memory banking' exercise, in which community elders were interviewed regarding their perceptions of change over the past 30 years. This exercise was substantiated by a literature and database review. A part of the exercise to construct the 'with and without' research scenarios was the use of participatory Geographical Information System exercises in which the community mapped out contemporary and past plantings.

The valuation element of the case study used choice modeling to explore the trade-offs farmers were making between modern and native varieties. The aim was to investigate the extent of the productivity sacrifice made by farmers to protect potato biodiversity. A sample of 85 households from the two case study sites were asked to make choices between alternative potato planting strategies. The outcomes of the alternative strategies were described in terms of: native potato cultivar diversity, area planted to native potato cultivars, expected yield of commercial varieties, and expected price of commercial varieties.

Planting choices in the conditional logit model used to explain the choices were found to be determined by all of the descriptors except area planted to native cultivars. Similar results were found for the two case studies despite their differences in terms of market access and climate. Put simply, the results suggest that farmers regard diversity, yield and price as positive factors influencing their planting choice. Hence, an increase in the yield of a commercial variety will encourage farmers to plant more of that variety. However, any resulting loss of biodiversity will be a cost because it too is valuable to the household. The results of the conditional logit model show the relative importance of pairs of attributes. From that analysis, the extent to which farmers are willing to trade off improved yield with reduced biodiversity can be elucidated (Table 2.2).

Table 2.2. Case study results.

Research focus	Sample size	Environmental impact	Value estimate (US$)
Supplemental irrigation of wheat in Syria	591	Water Soil	$102/ha 0.12/$m^3$
Zero tillage in rice-wheat systems in South Asia	66	Water	$142 per respondent
Sluice Gate management in the Mekong delta	120	Flora Fauna	$5.56 per respondent
Potato genetic diversity in Peru	85	Biodiversity	Not estimated[a]

a This study estimated the loss of native cultivar diversity as a result of the adoption of modern varieties.

2.3.5 Remaining challenges

The four Center studies demonstrating EIA within the BCA conceptual framework highlighted in this chapter have shown the capacity of existing Center impact assessment work to be extended to meet new challenges. In some cases, existing biophysical research on outcomes of research has been utilized as a platform for the valuation task, whilst in others more biophysical research has been necessary to link the research investment to environmental values. In all cases, new valuation work was required. Remaining challenges exist in both the biophysical and valuation elements in all of the studies.

The first apparent gap in the case studies is that none of them focused on off-farm environmental values.[36] All the studies considered the direct impacts on farm profitability of a research-induced change in farm inputs. Some of them analyzed research investments that changed farming practices (potato plantings, SI and ZT), while others looked at research projects that directly focused on the management of the environmental inputs (water in the Mekong delta). The end point was predominantly the impact on farm profitability. Even where non-marketed goods and services were targeted (e.g. flora and fauna in the Mekong delta), the beneficiaries considered were the farm households. This leaves the field of non-market valuation of off-farm effects of CGIAR research investments still open for exploration

Attribution of environmental impacts of research is problematic in most impact assessments, even those where economic outcomes are the focus. The attribution problem arises largely because of the complexities involved in understanding the relationship between the factor impacted by research, and the environmental outcomes. In part this is because there are multiple inputs to research efforts and assigning proportional responsibility is not straightforward. Furthermore, the separation of the impacts of research from other factors causing change raises significant issues. The studies reported here have largely left these aspects aside in rising to the valuation challenges. Even the IWMI study where biophysical modeling was at the heart of the original research work, it was necessary to seek local's perceptions of change as the basis for predicting the impact of the revised sluice gate operation. This is especially noteworthy as the original modeling was of factors such as water quality. The link between water quality and species richness was not established in the modeling, yet it is that which generates well-being for the local residents. Put simply, the existing modeling studies do not often target the factors that create or destroy value for people.

Similarly, the CIP study involved asking the village elders to judge the changes in varietal plantings over time. However, just how much of that change was due to CIP research was not quantified.

The biophysical linkages between research outputs and resource availability in both

36 The off-farm GHG emission impacts considered in the ICAR study were not taken through to the monetary valuation stage of analysis. Given the increasing presence of carbon markets internationally, a relatively straightforward extension to the ICAR study would be to use carbon prices to estimate the GHG reduction values arising from ZT.

the ICARDA and ICAR studies are also relatively tenuous. Whereas the relationships are defined between SI and water use in the ICARDA study, and ZT and water use in the ICAR analysis, the resultant impacts on the supply of water remain undefined. This is partly because of a lack of knowledge regarding the relevant aquifers but also because the responses of farmers to the availability of 'saved' water were not quantified. Although ample information is available on the condition of the aquifers over time, cause-effect relationships are harder to establish and hence the definition of the counterfactual is problematic.

The valuation elements of the four studies also leave unanswered questions. While the use of Choice Modeling in two of the studies (ICARDA and CIP) has 'opened the door' for this technique to be used in a range of related applications, further development are required to ensure unbiased results. For example, the comparison in the ICARDA study of the stated preference results against the production function analysis, showed a large difference that deserves greater investigation. In addition, the use of contingent valuation in the ICAR and IWMI studies demonstrated the flexibility of that technique, but also showed the need for refinement of the method for the specific circumstances of the cases involved. Incentive compatibility, i.e. ensuring that respondents have the incentive to tell the truth when answering these valuation questions, also requires further development, especially when the two cases used the 'open-ended' version of the technique without a provision rule. This combination of features is likely to be incentive incompatible. Communication of information in a Choice Modeling application is also challenging in developing countries where levels of literacy and numeracy are low. The CIP study, for example, encountered issues associated with language dialects in remote regions and involved the use of satellite images of the local potato farming areas to stimulate interest in answering the choice questions.

2.4 Lessons learnt

The studies commissioned for this EIA in the CGIAR system exercise have demonstrated that the conceptual framework developed to integrate environmental impacts into the rubric of BCA is practical and capable of application. Each of the four Center studies reviewed in this report produced results that provide qualified but useful conclusions to the Centers and to the CGIAR as a whole. They demonstrate the significance and relative magnitudes of some of the environmental impacts resulting from Center investments albeit on a limited scale, and with some yet to be resolved technical problems. The CIP study is particularly notable in this respect as it attempted to demonstrate the presence and magnitude of a cost associated with potato productivity enhancement research.

The overall conclusion to be drawn for CGIAR research is the importance of extending conventional epIA studies that are based on the BCA conceptual framework to encompass environmental impacts. As the IWMI case study of sluice gate management in the Mekong River delta demonstrated, even a partial analysis can show that environmental benefits alone may cover the costs of research investments. Similarly, knowledge of the extent of any environmental costs triggered by research investments is also useful for CGIAR policy formulation.

However, all the Center studies commissioned by SPIA ventured into conceptual and practical application areas that were unfamiliar to the researchers involved. The studies highlighted the need for capacity building amongst the evaluation staff of the Centers. They also demonstrated the inadequacy of biophysical research efforts in the Centers in being able to separate the relationships between research investments and environmental outcomes, including feedbacks into farm productivity. The primary focus of biophysical research in the relatively few cases reviewed was not on the outcomes that are important for people. Rather, apart from the IWMI case, they concentrated on outcomes that are intermediate, for example, water quantity and quality rather than the richness of species that depend on the water. The reported studies can best be characterized as taking tentative initial steps, based largely on the more familiar traditional farming system and productivity work that has characterized CGIAR epIAs.

The valuation exercises also showed the specialized nature of the skills required to undertake this type of work. The level of sophistication, especially in non-market valuation is growing rapidly as the field expands. With growing realization of the significance of environmental impacts in resource allocation decision making, there are increasing calls for environmental values to be estimated, and the validity of these estimates is critical to good policy making. Hence, research has been directed toward the development and refinement of non-market environmental valuation, and extended into developing country contexts. For CGIAR epIAs to take advantage of these new directions, the capacity building started in the case studies will need to be continuous and expanded across Centers other than those engaged in the current exercise.

2.5 Future EIA work within the CGIAR

2.5.1 Mainstreaming EIA

All CGIAR Centers are undertaking research that has environmental impacts. Such is the interconnected complexity of farming systems that it is impossible to contemplate an agriculturally focused research agenda without considering its effects on the environment within which it is set. Recognition of this interconnectivity has been given through the new CGIAR's 'Environment for the People' objective.

Those Centers which are focused specifically on natural resources, specifically Bioversity International, CIFOR, IWMI, World Agroforestry Centre and WorldFish, have mandates that specifically include environmental issues and are thus prime candidates for prioritizing EIA integration with epIA exercises. Biophysical research staff in these Centers are also more likely to have environmental science skills, so EIA work there thus offers potential as 'low hanging fruit'.

That is not to say that other Centers should not be encouraged to embark on the EIA integration pathway. As demonstrated by the Center case studies highlighted here, there are many opportunities outside the natural resource focused Centers. The interface between agriculture and the environment is arguably one of the most vexed policy issue facing governments in developed countries today, for example, irrigation extractions versus environmental flows, vegetation protection versus clearing for cropping and grazing, etc. The same sort of conflict between agricultural and environmental interests is becoming the focus of attention in developing countries. Governments in these countries are faced with growing pressure, some of which emanates from developed country lobby groups, consumers and governments, to prevent further environmental decline due to agricultural industries. The environment becomes, in these circumstances, a competing user of the scarce resources used in agriculture. With this resource use conflict in mind, more research is warranted to better understand the capacity for increasing productivity generated from the scarce resources currently engaged in agricultural production, to satisfy food demands from a growing population without compromising (or even enhancing) the environment. EIA would be a key component of this type of research given that the dual goals of agricultural production and environmental protection would be addressed.

Such 'mainstreaming' of EIA integration into CGIAR epIA exercises is likely to come at least in part as a result of donor pressure. With donor funds being sourced from developed countries where environmental protection pressures are building, so accounting for environmental impacts will become increasingly important. However, ensuring quality control in epIA studies across Centers, prior to their becoming available to donors, will be an important task for SPIA in the initial years of EIA integration into epIAs becoming more widely and better known. A part of this task will, in some cases, involve donors being made aware of developments in integrating environmental benefits and costs into more conventional epIA work. Explanations of the processes and techniques involved, their strengths and weaknesses will be required to enhance acceptance.

Mainstreaming does not mean that all CGIAR research investments will be subjected to epIAs incorporating EIA, just as not all investments undergo epIA review. What it does mean however, is that the CGIAR incorporates EIA as a regular feature of its prioritization of research investment evaluations.

2.5.2 Resourcing

For mainstreaming to be achieved, CGIAR research investments must include sufficient funding for developing models and monitoring programs that link research outcomes to environmental effects as well as productivity impacts. This will involve data collection (monitoring) as well as model building at a scale that will enable the cause-effect relationship to be extrapolated to the same scale as will eventually be involved. This necessitates serious investments in the validation of model predictions through land, water and other relevant resource surveys, as well as *ex-post* measurements of key environmental variables. Part of the resourcing issue involves ensuring that sufficient Center staff members have skills in environmental as well as agricultural science to enable adequate biophysical analysis.

However, as has been made clear in this report, biophysical analysis is a necessary but not sufficient component of the EIA process, where it has to fit into the BCA conceptual framework. Environmental effect value estimates are also required. Adequate time and resources need to be allocated to the task of valuing environmental impacts. The significance of these impacts in decision making is now widely recognized and relegating their valuation to 'add-on' status would neglect their potential to sway investment decisions. This will require that sufficient Center staff members have adequate skills in environmental economics, especially expertise in environmental valuation techniques. Skill in applying non-market valuation techniques is particularly pertinent in this respect. Stated preference technique applications in particular can be easily flawed. A poorly conducted non-market application can be detrimental to the specific investment decision being assessed but can also have wider ramifications for the reputation of the technique.

Having well-qualified Center staff available to undertake both biophysical and economic components of EIA is certainly desirable, but it may not be feasible or practical in all Centers. Instead, arrangements could be made to share key qualified staff across a number of Centers or strategic partnerships could be sought with Associated Research Institutes (ARIs) that have staff with the required skills.

2.5.3 Methodological issues

CGIAR Centers have the opportunity to become research leaders in environmental impact assessment as it is applied in developing countries, particularly in environmental valuation which to date is relatively little explored in this context. There are numerous methodological challenges to be faced, however, as set out below.

Biophysical modeling under risk and uncertainty is being developed in developed country contexts, where applications are scarce, and more experience in Bayesian Network Analysis for example, would be productive.

Establishing the links between biophysical analysis and economic valuation stages is especially important. To do this, it is necessary to ensure that biophysical monitoring and modeling is aimed at measuring and explaining the factors that are relevant to people as sources of either benefits or costs. The focus needs to be on the effects of research on people, not the environment *per se*. Under the BCA conceptual framework it is not 'the environment' that benefits or endures costs. Rather it is people who enjoy benefits when the environment is improved or experience costs when it is degraded.

Part of establishing the biophysical-economic value linkage is the definition of research investments in terms of their anticipated environmental impacts. This requires the demarcation of the types of environmental benefits and costs to be generated according to the classification of values, i.e. use, passive use, non-use, and non-market. The importance of this classification process is in developing an understanding of the types of values involved. This gives direction to the biophysical element of the EIA, i.e. what is the outcome that needs to be understood, as well as providing a base to the process of allocating value estimation techniques.

The valuation techniques used must be consistent with the categories of defined values. Revealed preference techniques should be used as the first option because of their solid foundation in observed behavior. Stated preference techniques are to be used when no relationship to marketed goods and services can be identified, or when related markets are thinly traded or heavily distorted by government intervention or monopoly behavior.

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3. Agricultural technology, global land use and deforestation: a review and new estimates of the impact of crop research

James Stevenson,[37] Derek Byerlee,[38] Nelson Villoria,[39] Tim Kelley,[40] Mywish Maredia[41]

Abstract

There is a rich literature tackling the impacts of the adoption of new agricultural technologies on land-use change. In this chapter, the multiple causal pathways of impact between adoption of yield-increasing technologies, land-use change in general and deforestation in particular are reviewed.

New estimates of the impact of crop germplasm improvement in the major mandate crops of the CGIAR between 1965 and 2004 on global land-use change are provided, based on simulations carried out using the Global Trade Analysis Project Agroecological Zone model (GTAP-AEZ): a multi-commodity, multi-regional computable general equilibrium (CGE) model linked to a global spatially explicit database on land use.

Although the model incorporates countervailing effects of technology on land-use changes through prices and land rents, the results support Borlaug's hypothesis that increases in cereal yields as a result of widespread adoption of CGIAR related crop germplasm have saved natural ecosystems from being converted to agriculture. However, the results of this study suggest that this effect is of a much smaller magnitude than Borlaug estimated using a simplistic approach based on a global food equation.

The GTAP-AEZ estimates suggest that the total crop area in 2004 would have been between 17.9 and 26.7 million hectares larger in a counterfactual world which had not benefited from crop germplasm improvement since 1965. Of these hectares, 12.0 to 17.7 million hectares would have been in developing countries.

One limitation of this modeling effort is that the counterfactual scenario simply models what would have happened in the absence of technological change based purely on market transactions, and cannot include potential political interventions (for example, higher rates of hunger or poverty in the counterfactual world spurring governments to intervene to bring a larger area into cultivation). Thus, these figures could be considered lower-bound estimates.

The results of additional simulations in GTAP on productivity shocks to soybean in Brazil and oil palm in Indonesia and Malaysia, two crops that are often associated with tropical deforestation, are also reported to illustrate the similarities and differences between broadly based productivity improvement in cereals and the more focused productivity gains in oilcrops grown on the forest margins.

The chapter concludes by suggesting how the CGIAR can best ensure it maximizes its potential positive impact on global land-use change through its technological and policy research.

3.1 Introduction

3.1.1 Why the relationship between agriculture and deforestation is important

The competition for global agricultural land and forest resources is high on the development agenda as a result of climate change, rising commodity prices and rising land prices. Land-cover change is the third most important human-induced cause of carbon emissions globally and the second most im-

37 CGIAR Independent Science and Partnership Council Secretariat, United Nations Food and Agriculture Organization, Rome. Email: james.stevenson@fao.org.

38 Chair, CGIAR Standing Panel on Impact Assessment.

39 Department of Agricultural Economics, Purdue University.

40 CGIAR Independent Science and Partnership Council Secretariat, United Nations Food and Agriculture Organization, Rome.

41 Member, CGIAR Standing Panel on Impact Assessment and Department of Agricultural, Food and Resource Economics, Michigan State University.

portant in developing countries (World Bank, 2010). In turn, agricultural expansion (especially for commercial agriculture) is the single most important determinant of tropical deforestation. Between 1980 and 2000, 83% of all new agricultural land in the tropics came from either intact forests (55%) or disturbed forests (28%) (Gibbs et al., 2010).

Many have argued that agricultural research to increase yields is critical to saving the world's remaining forests and, in doing so, limiting GHG emissions (Burney et al., 2010) and losses of biodiversity (Green et al., 2005; Phalan et al., 2011). Technological change that improves productivity on existing agricultural land saves natural ecosystems (including forests) from being converted to agriculture. This is commonly known as the Borlaug hypothesis after Norman Borlaug (2007), who claimed that the intensification of agriculture between 1950 and 2000, partly as a result of the technological change made possible by the Green Revolution, had saved land from being brought into agricultural production.

However, the relationship between adoption of new technologies and land use is complex. Increases in productivity from new technologies also increase the profitability of agriculture in comparison with alternative land uses (such as forest and pasture) thereby encouraging expansion of the agricultural land frontier. Several case studies in Angelsen and Kaimowitz (2001) support these types of land-use effects resulting from technological change.

It is not possible to sort out these relationships directly in empirical studies, since the counterfactual cannot be observed. Moreover, the pathways through which technological change effects land-use change are manifested through markets for agricultural outputs and the factors of production. For these reasons, the impacts of technical change can only be estimated econometrically or through models.

The CGIAR is a major source of technologies for food crops, and its impacts on productivity have been well documented (Renkow and Byerlee, 2010). However, impacts of the CGIAR system on the environment have received little attention. The land-use effects of technological change may represent the single most important source of environmental impacts of the work of the CGIAR globally (Renkow, 2010). Earlier studies have argued that CGIAR-led agricultural technologies have significantly reduced agricultural expansion (over what otherwise would have emerged), and in doing so potentially saved forests (Nelson and Maredia, 2001; Evenson and Rosegrant, 2003).

3.1.2 Rationale and organization of the chapter

In section 3.2 we open the discussion and examine the theoretical basis of the relationship between increased agricultural productivity and changes in land-use globally and for developing countries, briefly reviewing earlier modeling and estimation efforts. In section 3.3 we provide estimates of land-saving effects of technological change based on a CGE model. Section 3.4 begins with an analysis of what is happening on the agriculture-forest frontier as documented in global statistics and research studies from the last 10 years, and then asks to what extent new technologies or institutional and policy contexts, especially governance of land and forest resources, are contributing to deforestation.

This chapter thus takes a two-pronged approach to address these questions with respect to the CGIAR, enabling us to provide an informed view of the relationship between CGIAR research and land-use change. From the perspective of sections 3.2 and 3.3 we are starting with agriculture, and technical change in crop agriculture in particular, and working forwards to examine methods for assessing the contribution of agricultural technology to agricultural expansion and, further, to the extent of encroachment into forests. In section 3.4, we work in the other direction, starting from known hotspots of deforestation (the large forest-rich countries of Brazil and Indonesia) and working back to look at causal factors, trying to tease out the circumstances under which technological change may have played a role. This two-pronged strategy is clarified in Figure 3.1 below, showing that, whichever end we start from, there will always be some uncertainty regarding the causal connections in the middle of this causal chain. Nonethe-

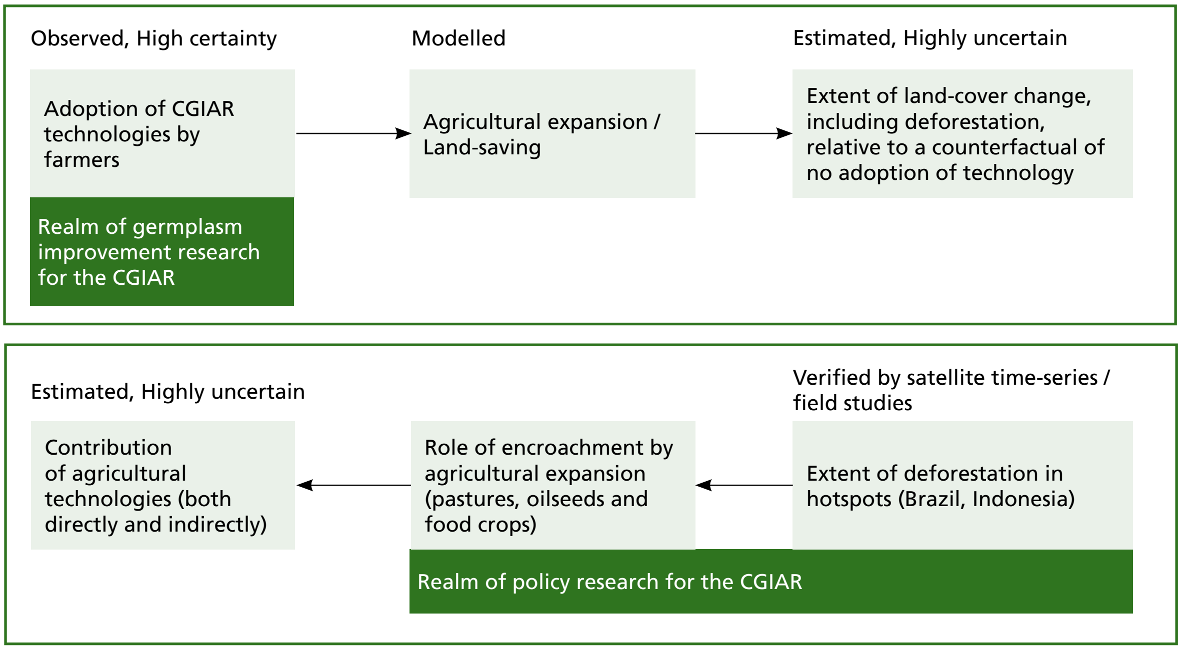

Figure 3.1. Description of rationale for sections 3.2 and 3.3 (top) and section 3.4 (bottom) in terms of the chain logic underlying the reviews and the links to CGIAR research

less, these complementary approaches help us understand the issues in detail.

Sections 3.2 and 3.3 focus on the well-documented adoption of agricultural technologies in the Green Revolution, and asks whether this caused agricultural expansion or saved land from being brought into production. Section 3.4 focuses on tracing back the causal factors underlying land-use change associated with particular crops in the two global deforestation hotspots: Brazil and Indonesia. We conclude in section 3.5 by suggesting how the CGIAR can most effectively contribute in future to the twin goals of maximizing agricultural productivity and minimizing forest loss.

3.2 Alternative perspectives on agricultural intensification and land-use change

3.2.1 Land-saving effects: the Borlaug hypothesis

Norman Borlaug's response to environmental critiques of the Green Revolution is summarized thus:

> *"If the global cereal yields of 1950 still prevailed in 2000, we would have needed nearly 1.2 billion[42] more hectares of the same quality, instead of the 660 million hectares used, to achieve 2000's global harvest. Moreover, had environmentally fragile land been brought into agricultural production, the soil erosion, loss of forests and grasslands, reduction in biodiversity, and extinction of wildlife species would have been disastrous."* Borlaug (2007)

Borlaug argues two related points here. The first argument is that increases in agricultural yields have saved new agricultural lands from being brought into production. The second is that the 'saved land' provides valuable ecosystem services by maintaining natural areas. In this chapter, we primarily address the first of these hypotheses (how much land has been saved by new technologies for food crops), although we touch on the encroachment factor of agricultural expansion (how much of the averted land expansion would have been into forests versus other less ecologically valuable habitats, such as grasslands).

42 We believe this to be an error in the text of the Science magazine article. We think that it should read: "… we would have needed nearly 1.2 billion hectares in total, of the same quality, instead of the 660 million hectares used". This suggests an estimated land saving of between 500 and 600 million hectares, not 1.2 billion. Unfortunately, it is not possible to check this with the author now.

Estimates based on the global food equation

A simple identity often referred to as the global food equation, links global population (N), food consumption and production (q), land area (L), and agricultural yield (q/L), with demand on the left hand side and supply on the right hand side (Angelsen, 2010):

$$\frac{Nq}{N} \equiv \frac{q}{L} \times L \qquad (1)$$

Borlaug's estimates noted above involve simple calculations using this identity: if yields do not change but population increases, then more land is required to feed everyone at the same level (also bearing in mind rising per capita consumption). The variables for this identity for cereals, which includes the world's major food staples for the period 1961–2008[43] are given in Table 3.1. During this period, global population more than doubled and per capita consumption increased by 20%. The increase in cereal production to meet this increase in demand has overwhelmingly come from an increase in yields. Area harvested increased by only 7%.

The argument that in the absence of the observed 140% increase in cereal yields between the 1960s and 2000s, the area under cereals would have expanded by a similar percentage is based on a number of assumptions. First, population growth and economic growth are assumed exogenous to agricultural productivity.[44] Theory (and some evidence) might suggest that higher agricultural productivity may reduce both human birth rates and death rates, leading to ambiguity. The causal contribution of agricultural productivity growth to economic growth at early stages of a country's development is widely recognized (Christiaensen et al., 2010; Valdes and Foster, 2010), even if it is difficult to establish empirically (Gollin, 2010).

More importantly, the estimates of land savings based on the above identity do not consider changes in demand as a result of changes in the supply. In the absence of the observed yield increases, prices would have increased and curtailed at least a proportion of the per capita consumption increases observed in developing countries with, of course, negative implications for the number of people suffering from malnutrition.

In addition, on the supply side, higher output prices would induce a supply response by farmers. Keeney and Hertel (2009) note that this is an important area of uncertainty in the models that attempt to estimate the impact of biofuels on land use in response to higher prices. Farmers may boost supply by increasing cropping intensity or using more capital (e.g. irrigation) and labor (e.g. weeding).

Given that yields and consumption are endogenous to the agricultural system, simple calculations such as those performed by Borlaug and many others since, tend to overestimate the extent of land savings relative to a more realistic counterfactual.

Nelson and Maredia (2001) attempt an approximation to a counterfactual by applying a coefficient of substitution between yield and area of 1:0.5 on the supply side based on Evenson (unpublished estimates). According to this calculation, over the 30 years from the 1960s to the 1990s, area in seven CGIAR-mandated crops (wheat, rice, maize, pulses, barley, cassava and sorghum) increased by 75 million hectares at the time that yields approximately doubled. Application of the above coefficient suggests that land in production would have been 230 million hectares higher than observed. Note that the Nelson and Maredia (2001) and Borlaug's (2007) estimates of land savings correspond to yield growth from all the sources and is not restricted to only the effects of yield increments from crop genetic improvement (which represents only about a third of total yield gains observed from the 1960s to the 1990s). If yield increases from crop

43 This is the longest range for which the data are available through http://data.un.org/.

44 More sophisticated approaches to modeling these impacts (using partial equilibrium or computable general equilibrium, or CGE, models) make similar to assumptions about population, although it is possible to make population endogenous if a dynamic CGE approach is used. With regards assumptions about the links to economic growth, the key advantage of CGEs is the ability to make the impact on the rest of the economy endogenous.

Table 3.1. Variables of the global food and land equation between 1961 and 2008.

	1961–1963	2006–2008	% increase
Demand side			
Population (billions)	3.13	6.62	111.6
Cereal consumption per capita (kg/capita/ year – food, feed and other uses)	294.3	358.3	21.8
Supply side			
Area harvested (million ha of cereals)	653.7	697.2	6.7
Cereals yield (t/ha)	1.4	3.4	141.5

germplasm improvement only are considered, along with the effects on changes in cropping intensity, global estimates for land savings to achieve the 1990s level of production would be around 85 million hectares (Maredia, 2003, unpublished report). However, these studies do not account for the impacts of food price increases on consumption demand, substitution effects on other crops, or impacts through factor markets.

Estimates based on global partial equilibrium modeling

Better economic modeling approaches are needed to account for various market effects of technical change. For the CGIAR, Evenson and Rosegrant (2003) conducted a comprehensive modeling analysis based on the findings of a major initiative that estimated the adoption and impact of crop germplasm improvement in developing countries (Evenson and Gollin, 2003). They compared the observed level of technology in developing country agriculture in 2000 (referred to as the 'base case') with a counterfactual case of no crop germplasm improvement since 1965. In this counterfactual scenario, developed countries still benefited from crop germplasm improvement consistent with their historical record for the period.

Evenson and Rosegrant (2003) used the International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT), a multi-market, multi-country model with 17 crop commodities[45] (very close to those covered by Nelson and Maredia) and 35 countries or regions. In IMPACT,[46] crop supply and demand factors determine the market-clearing prices, quantities supplied and consumed, and the trade volumes.

In IMPACT, domestic crop production is determined by area and yield response functions. Harvested area is specified as a response to the crop's own price, the prices of other competing crops, the projected rate of exogenous (non-price) growth trends in harvested area, and water (eq. 2). The projected exogenous trend in harvested area captures changes in area resulting from factors other than direct crop price effects, such as expansion through population pressure and contraction from soil degradation or conversion of land to non-agricultural uses. Yield is a function of the commodity price, the prices of labor and capital, water, and a projected non-price exogenous trend factor. The trend factor reflects productivity growth driven by technology improvements, including the research outputs of the CGIAR (eq. 3). Annual production of commodity *i* in country *n* is then estimated as the product of its area and yield (YC). On the demand side, in IMPACT domestic demand for a commodity is the sum of its demand for food, feed and other uses, and demand for food is a function of standard factors such as per capita income and population.

45 The number of commodities and country groups featured in IMPACT has changed over time. The 2002 version (Rosegrant et al., 2002) features 36 countries and 16 commodities: beef, pork, poultry, sheep and goat, eggs, milk, maize, other coarse grains (barley, millet, oats, rye, sorghum), rice, wheat, cassava (and other tubers), potatoes, sweet potatoes and yams, meals (e.g. copra cake, groundnut cake), oils (vegetable oils and products, animal fats and products), and soybeans. Evenson and Rosegrant (2003) report that the version they used contained 17 commodities across 35 countries.

46 The equations here are for IMPACT 2005 version. Subsequently, additions were made to the model to incorporate supply and demand schedules for water (Rosegrant et al., 2008).

Area response:

$$AC_{tni} = \alpha_{tni} \times (PS_{tni})^{\delta}{}_{iin} \times \prod_{j \neq i}(PS_{tni})^{\delta}{}_{ijn} \times (1 + gA_{tni}) \quad (2)$$

Yield response:

$$YC_{tni} = \beta_{tni} \times (PS_{tni})^{\gamma}{}_{iin} \times \prod_{k}(PF_{tnk})^{\gamma}{}_{ikn} \times (1 + gCY_{tni}) \quad (3)$$

where AC = crop area; YC = crop yield; QS = quantity produced; PS = effective producer price; PF = price of factor or input k (for example labor and capital); Π = product operator; i,j = commodity indices specific for crops; k = inputs such as labor and capital; n = country index; t = time index; gA = growth rate of crop area; gCY = growth rate of crop yield; ε = area price elasticity; γ = yield price elasticity; α = crop area intercept; β = crop yield intercept.

Evenson and Rosegrant (2003) estimated that crop area in 2000 was 2.8–4.6% less than would be the case for the counterfactual with no crop germplasm improvement in developing countries over the period. Land-saving estimates were higher for rice (7.5–9.4%), one of the focus crops of the Green Revolution in Asia, than for other staple crops.

A range of 3–4% of agricultural land saved between 1965 and 2000 corresponded to 9–12 million hectares in developed countries and 15–20 million hectares in developing countries. These estimates of a total land saving effect from crop germplasm improvement of 24–32 million hectares between 1965 and 2000 are an order of magnitude lower than those of Nelson and Maredia (2001), but are still significant from the perspective of potentially averted deforestation, biodiversity loss and GHG emissions. Evenson and Rosegrant (2003) also use assumptions to apportion part of the land saving to CGIAR crop germplasm improvement research.

The IMPACT model provides a greater degree of economic realism than the estimates based on the global food equation, and represents an intermediate case between the perfectly inelastic global food equation and the perfectly elastic von Thunen or open economy case (outlined in the following section).

There are still, however, many restrictive assumptions associated with the model. First, IMPACT is only a partial equilibrium model for the agricultural sector – it does not compute equilibria for other markets, thereby missing an entire pathway of impacts via impacts on non-farm incomes and their feedback to the agricultural sector via product and factor markets (labor and capital). Second, the model does not include a land market and lacks any explicit link to the physical realm of existing land cover. This means that we cannot estimate the ‘encroachment factor’ – the extent to which the additional hectares required under lower-yielding technologies in a counterfactual world would have come from forest, rather than from grazing land or other land cover with lower value to society than forests. Modeled using IMPACT, crop germplasm improvement can only ever save land because there is no mechanism for modeling land competition between crop and non-crop uses and, even among crops, the coverage is only partial.

Hertel (2010) made an important refinement to modeling global land supply and demand based by incorporating the major drivers into a single simple model as follows:

$$q_L^* = \left[(\Delta_A^D + \Delta_L^S - \Delta_L^D) \,/\, (1 + \frac{\eta_A^{S,I}}{\eta_A^{S,E}} + \frac{\eta_A^D}{\eta_A^{S,E}} \right] - \Delta_L^S \quad (4)$$

where q_L^* is the percentage change in long-run land supply and demand in equilibrium;

Δ_A^D is an exogenous shift in commodity demand (e.g. from biofuels mandates);

Δ_L^S is an exogenous shift in land supply (e.g. from urbanisation);

Δ_L^D is exogenous growth in agricultural yields (e.g. from prior investments in agricultural research);

$\eta_A^{S,I}$ is an elasticity describing the potential for intensification of agriculture (i.e. the supply response to prices at the intensive margin);

$\eta_A^{S,E}$ is an elasticity describing the potential for land expansion (i.e. the supply response to prices at the extensive margin, reflecting land scarcity and governance factors protecting natural areas from encroachment); and η_A^D is the demand response to prices (describing the potential for endogenous demand reduction).

The main contribution of this equation is to recognize that agriculture can expand at the extensive margin or at the intensive margin with the overall effects on land use determined by the ratio of the two. The first ratio in the equation $\frac{\eta_A^{S,I}}{\eta_A^{S,E}}$ captures the incentives to expand at the intensive margin. Land scarcity, reflected in a high value for the ratio $\frac{\eta_A^{S,I}}{\eta_A^{S,E}}$ encourages intensification. Conversely, where land is readily available and relatively cheap, expansion occurs. The second ratio of relevance above is the ratio $\frac{\eta_A^{D}}{\eta_A^{S,E}}$ indicating that response is least where demand is highly inelastic (recall Δ_A^D is negative) relative to the elasticity at the extensive margin.

In the simplest of cases, which corresponds to the Borlaug hypothesis, with no yield response to price and no demand response (ie. $\eta_A^{S,I} = \eta_A^D = 0$), a yield shock Δ_L^D, results in a proportional change in area of the opposite sign. With $\eta_A^{S,I} = \eta_A^D = \eta_A^{S,E}$ the impacts of a positive yield shock are equally distributed between supply reduction at the intensive margin, supply reduction at the extensive margin, and higher demand (Hertel, 2010).

Elasticities of land expansion with respect to prices $\eta_A^{S,E}$ are especially scarce. Hertel (2010) cites low estimates of 0.025 to 0.033 globally, but ranging up to 0.9 for Brazil (Barr et al, 2010). Likewise, there are few estimates of the elasticities of intensification $\eta_A^{S,I}$ outside of developed countries. They are likely to change over time, depending on exploitable yield gaps. In the United States, Hertel (2010) finds that $\eta_A^{S,I}$ for maize has fallen from 0.7 in the post War period to 0.2 recently.

The contribution of this model is to show how an endogenous response to demand shocks within the food and farming system can buffer agricultural land-use change, acting as a shock absorber (via higher land prices encouraging intensification) limiting the extent of demand-led rises in food prices. In section 3.3, we empirically model these relationships in a CGE model. However, by using a more complex model, we can include an additional dynamic in the relationship between exogenous technological change and land-use (which was not incorporated in Hertel's simple model) – the land rent effect, discussed below.

3.2.2 The land rent effect: Jevon's paradox

When innovations (such as new agricultural technologies) that improve productivity or reduce costs for producers are adopted, they increase producers' profits, at least in the short run. Making agriculture more profitable relative to other land-uses at the margin encourages agricultural area expansion. Therefore it is an apparent paradox that the adoption of a technology that ostensibly saves land (i.e. increases yield) could under some circumstances lead to total area expansion.

The general principle of this paradox, a net increase in use of an input when a technology is introduced that increases the efficiency of the same input, was highlighted by Jevons (1865; cited in Alcott, 2005) in the context of aggregate coal use and its relationship to new blast-furnace technologies:

> *"Economy multiplies the value and efficiency of our chief material...[and] renders the employment of coal more profitable, and thus the present demand for coal is increased. . .. [If] the quantity of coal used in a blast-furnace, for instance, be diminished in comparison with the yield, the profits of the trade will increase, new capital will be attracted, the price of pig iron will fall, but the demand for it increases and eventually the greater number of furnaces will more than make up for the diminished consumption of each"*

In the context of agricultural technologies, a result consistent with Jevon's paradox would be where yields (i.e. land efficiency) increase and there is increased expansion of agricultural land. The work by Angelsen and Kaimowitz (2001) incorporates a number of local and national level case-studies, many of which find this kind of result.

Models that make land rents endogenous can simulate how the returns to alternative land-uses, such as agriculture and forest, vary under different scenarios. In a context where agriculture is well integrated in markets, and assuming two kinds of land-use (agriculture and forest) we can model land rents as a function of distance to a forest frontier. In a von Thunen model (after von Thunen, 1826; in Angelsen 2007) land rent from agricultural activities (r) is modeled as a function of distance (d) from a central market and can be computed as a residual as follows (Angelsen, 2007):[47]

$$r(d) = py - wl - qk - c - vd \qquad (5)$$

where yield is given by y and this output is sold in a central market at price p; labor (l) and capital (k) required per hectare are at prices w (wage) and q (annual costs of capital); the cost of defending property rights is given as c; and transport costs per km are denoted by v and the distance from the center as d.

Assuming perfect markets and homogenous land quality, then agriculture expands on the frontier until there are zero returns to land (i.e. r = 0). That is the distance from the center at equilibrium is given by:

$$d = (py - wl - qk - c) / v \qquad (6)$$

Assuming perfectly elastic demand, p is unchanged, and new agricultural technologies that raise yields will tend to promote expansion of the frontier by boosting output (y). New agricultural technologies that reduce costs promote expansion by reducing labor or capital requirements per unit output (l and k respectively). If demand is less than perfectly elastic, both effects could be overruled by reductions in p either locally or globally through trade. The insights gained from a land rent perspective on the kinds of factors that inhibit or promote the expansion of agricultural area are summarized in Table 3.2 below.[48]

The size of the supply shift and the elasticity of demand are the two crucial determinants of the commodity price effect from technological change. The size of the supply shift will be determined by the market share of the adopters and the average treatment effect size of the technology on reducing the per-unit cost of production.

The impact of new technologies on agricultural expansion may also be transmitted through capital and labor markets if the technological change is factor biased. Technological change that increases labor use per hectare has the potential to constrain agricultural expansion through impacts on local labor markets, particularly in forest-rich regions with high underemployment of labor, where deforestation induced by population growth may be tempered by in-

Table 3.2. Factors promoting and limiting expansion of agriculture at the frontier.

Factors promoting expansion	Factors limiting expansion
Higher output prices from increased demand or low supply (*increase in p*)	Lower output prices from lower demand or high supply (*decrease in p*)
Lower wages or opportunity costs of labor (*decrease in w*)	Higher wages or opportunity costs of labor (*increase in w*)
Lower cost of defending property rights (*decrease in c*)	Higher cost of defending property rights (*increase in c*)
Technologies that increase yield (*increase in y*)	Higher values for alternative forestry uses (*reduces d at which r = 0*)
Technologies that save inputs (*decrease in l or k*)	
Reduced access costs (*decrease in v*)	Higher access costs (*increase in v*)
Lower costs of capital (*decrease in q*)	Higher costs of capital (*increase in q*)

Source: Angelsen and Kaimowitz (2001)

47 This model assumes well functioning markets and profit maximization. More restrictive models are reviewed in Angelsen (1999).

48 Note that c can be negative in the case of forests being cleared to establish property rights on untitled land – a situation that is common in Brazil.

creased employment. The process of technological change from traditional varieties to modern varieties in the case of the Green Revolution was generally labor intensive, at least in the early stages.

The empirical case-studies included in the volume by Angelsen and Kaimowitz (2001) are built around one or more of the above impact pathways. Many studies at the local level present tropical agricultural technologies in a bad light. Yield increases for cash crops are not tempered by price reductions, boosting land rents at agriculture–forest frontiers and encouraging agricultural expansion. Clearly, when elastic demand is combined with spatial bias in favor of adoption at the frontier, we would expect technological change to cause further deforestation. In those cases, the increased profitability effect on demand for land may dominate the output price effect and lead to greater agricultural expansion. However, in the case of staple crops like rice, wheat, maize and cassava, at a global rather than local level, the impact of widely adopted technologies on land-cover change is likely to be mainly via a suppression of agricultural prices, relative to the counterfactual, that should result in a net land saving in the aggregate.

Two factors combine to constitute the impact on output prices: output market share, and the elasticity of demand for the good in question. For staple crops the elasticity of demand is usually in the range -0.2 to 0.4 (i.e. inelastic), but this varies by crop and by country. It is useful to think of the inverse of the elasticity of demand (1/*e*) which tells us how much an increase in supply (from technological change in agriculture) will reduce the price. The market share of the sector experiencing technological change – defined as the proportion of the total market supply that comes from adopters of the technology – scales the effect. This rationale applies whether we are studying an increased supply from a given country selling into global markets, or from one sector (e.g. uplands) of a dual agricultural sector selling into a national market. Following Angelsen (2007) we can write:

$$\frac{1}{e_1} = \frac{\delta p}{\delta x_1}(x_1/p) = \frac{\delta p}{\delta x_1}(x_1/p)(x/x) = \frac{\delta p}{\delta x_1}(x/p)(x_1/x) = \frac{1}{e}(x_1/x) \quad (7)$$

where e is elasticity of demand in the market, x_1/x is the market share of sector (country) 1 in a national (global) market. There is also the important scaling effect contributed by the land supply function in the countries in question (as we have seen from Hertel's model discussed previously); for countries where agriculture already dominates land-use, the potential for further impacts via land rents is much more limited.

In summary, we have outlined three different conceptual models – the global food equation; partial equilibrium models (e.g. IMPACT); and the von Thunen / open economy models. Table 3.3 demonstrates

Table 3.3. Overview of the main models used to estimate land savings.

Model	Main features	Impacts
Global food equation	• Perfectly inelastic supply (i.e. strongest possible price suppression from increased supply) • No link to economic growth • Does not capture land rent effect	Always finds land saving effects from yield-increasing technologies
Partial equilibrium model (e.g. IMPACT)	• Intermediate elasticity of supply • Economic growth endogenous to the model • Does not capture land rent effect	Always finds land saving effects but will be of lower magnitude
General equilibrium model (e.g. GTAP-AEZ)	• Intermediate elasticity of supply • Economic growth endogenous to the model • Does capture land rent effect	All outcomes are possible, depending on the parameters used
von Thunen / open economy model	• Perfectly elastic supply (i.e. no price suppression from increased supply; unlimited demand) • No link to economic growth • Exclusive focus on land rents – all benefits from productivity increases go to producers	Always finds land expansion effects from yield increases (as per table 2)

how these models are based on different assumptions – primarily about the elasticity of demand for crops – and cover different types of impact. A global equilibrium model is included here for comparison, as it is the tool we use in this chapter, and is introduced in the following section.

3.3 Estimating the impact of long-run technological change in agriculture using a global computable general equilibrium model

3.3.1 The Global Trade Analysis Project model

For a more comprehensive model we turn to a global model that includes land rent effects and impacts on land use via factor markets. The Global Trade Analysis Project (GTAP) model is a multi-commodity, multi-regional CGE model based on national or regional input-output tables. Villoria (2011) uses GTAP-AEZ, a version of GTAP which is linked to a global spatially explicit database on land use. The foundations of these data are the global datasets for agricultural productivity from Monfreda et al. (2008) and forests from Sohngen et al. (2009). Lee et al. (2005) used these data to develop a land-use and land-cover database that offers a consistent global characterization of land in crops, pastures and forestry, taking into account biophysical growing conditions. Agroecological zones represent six different lengths of growing period (6 x 60 day intervals) spread over three different climatic zones (tropical, temperate and boreal).

Derived demand for land

The basic production function in the GTAP-AEZ framework is given in Figure 3.2 below, where it can be seen that output is a function of all intermediate inputs and a value-added composite. These factors of production substitute for one another with the ease of substitution governed by the parameter δ_T. As with the standard GTAP model, value-added is a composite of skilled and unskilled labor, capital, land and natural resources (in the case of the extraction sectors). The ease with which these factors substitute for each other is governed by δVA and this determines the demand for land. The substitutability of the value-added components in the production of crops implies that producers can substitute capital and labor for land to increase output. So it is possible to increase production using the same amount of land by employing more of the non-land factors. In other words, the yields are endogenous.

The land input is an aggregation of the diverse agroecological zones. For this we

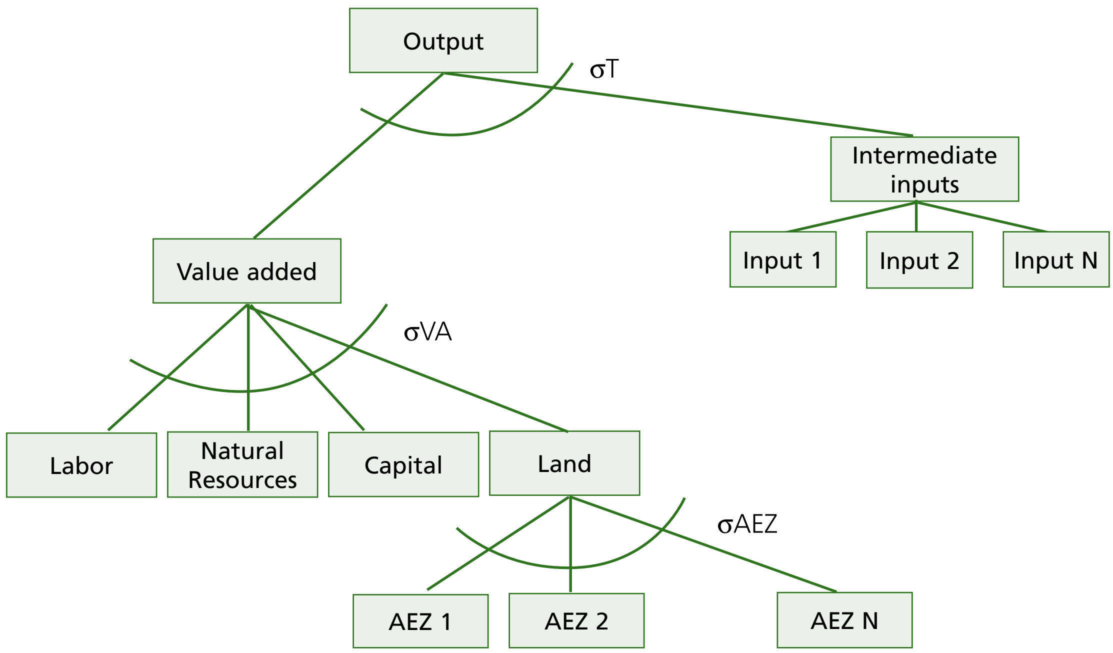

Figure 3.2. Land demand in GTAP-AEZ

assume that the same products produced in the same region must share a common price since they are perfect substitutes in use. If, as we assume, production functions for each crop and within a given region are similar across agroecological zones, and the firms face the same prices for non-land factors, then land rents in comparable activities must move together (even if they do not share the same initial level). In this case, from the point of view of land markets, the returns to land on different agroecological zones employed in the production of the same product must move together. This suggests a very high elasticity of substitution, δAEZ, between agroecological zones in the crop-specific national production function specification.

Land supply
The GTAP-AEZ framework used for this work introduces land competition directly into land supply via a two-tiered structure such as that used by Keeney and Hertel (2009), shown in Figure 3.3. In the upper tier, crops compete with each other for land within a given agroecological Zone. In the lower tier, crops as a whole compete with grazing and forestry for land within a given agroecological zone. In addition, different agroecological zones can be substituted in the production of any single agricultural or forest product.

Calibration of the constant elasticity of transformation (CET) of land supply functions in the model is based on the available

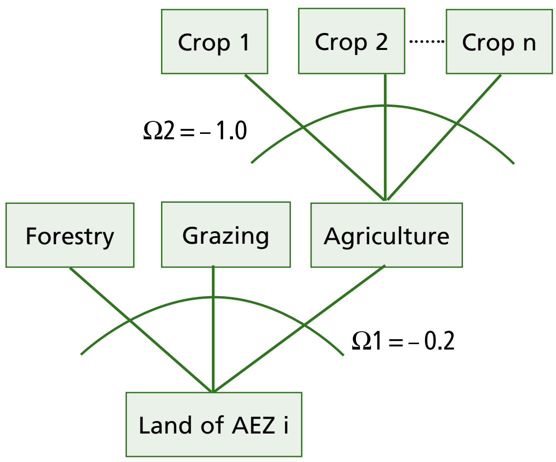

Figure 3.3. Land supply in GTAP-AEZ

econometric evidence. Recent evidence for the United States from Choi (2004) indicates that the elasticity of land supply to forestry averages about 0.25 (i.e. a 1% increase in the land rents for forests relative to the land rents of competing uses increases land supply to forests by 0.25%, provided that forests have an infinitesimal share of total rents). Accordingly, we set the CET parameter at the bottom of this supply tree equal to -0.25. This places the maximum forest land supply elasticity at 0.25. In agroecological zones where the forest land share is dominant, the supply elasticity will be much smaller, as would be expected. At the top of the supply tree, where land is supplied to individual crops, we employ the elasticity from the standard GTAP model. The GTAP model uses a CET value of -1.0, based on econometric evidence for land supplies to United States crop sectors, which suggests an upper bound of 1.0 on this elasticity.

GTAP-AEZ potentially offers some advantages over the IMPACT model discussed earlier.

- The crop coverage is complete in GTAP, although they are aggregated into only five categories complicating the inclusion of specific CGIAR crops. Eighteen agroecological zones are defined, several of which may occur within a country.
- In GTAP-AEZ the land rent effect is incorporated, which then allows us to model the net effect of land saving minus increased expansion, while also crudely modeling land supply through a CET between crop, pasture and forest lands. However, this greater ambition also results in further problems of restrictive assumptions which will need to be addressed in future research.
- GTAP-AEZ uses historical patterns of trade (the 'Armington assumption') between pairs of countries to influence where expansion and contraction of agricultural area takes place. It is possible to assess effects across different crops and different global regions, but the main results reported here are for aggregate results across all developing countries.
- GTAP-AEZ, as a CGE model, allows for general equilibrium effects through not only product markets, but also labor and capital markets.

3.3.2 GTAP-AEZ simulations of impacts on price and crop harvested area

Villoria (2011) uses factor-neutral productivity 'shocks' in GTAP-AEZ to replicate the simulations carried out by Evenson and Rosegrant using IMPACT to estimate the land-use impacts of crop germplasm improvement in developing countries since 1965. This is achieved by removing total productivity gains attributable to crop germplasm improvement in the CGIAR focus crops in developing countries (as documented in the book by Evenson and Gollin, 2003) in a 'back-casting' experiment. These negative total factor productivity (TFP) shocks to crop productivity allow us to track the main price, production, land-use and trade effects in a counterfactual world that did not benefit from these productivity gains in agriculture.

Since GTAP includes cassava, beans, lentils and potatoes in more aggregate groupings (vegetables), and maize, barley, sorghum and millet in another more aggregate grouping (coarse cereals), we show only results for two cereals – rice and wheat – and for the category of coarse grains as a whole. For simplicity, only crop germplasm improvement effects are presented in Table 3.4, although results assuming crop germplasm improvement synergies with other yield-changing factors were also computed by Villoria (2011). Overall price effects for rice, coarse grains and maize (as one of the main coarse grains) are large and quite comparable in the two models (Table 4), despite the fundamental differences in underlying data, product definitions and modeling assumptions.

In general, when the crops are taken in isolation, GTAP predicts a much larger area effect for the same scenario modeled by Evenson and Rosegrant (2003) of 6–8% expansion (corresponding to 52–75 million hectares) under the 1965 counterfactual, compared with 2–4% estimated in IMPACT. However, after accounting for the effects of crop substitution between those food crops that received the TFP shock and those that did not, the GTAP model estimates the global land savings effect across all crops of 1.5–2.2%, partly in developing countries (0.9–1.5%) where yields are reduced, and partly in developed countries (0.5–0.9%) through price and trade effects.

3.3.3 GTAP-AEZ estimates of the impact on land-cover change

Additional cropland in GTAP-AEZ can be obtained through conversion of pastures or forests. Table 3.5 shows that the model estimates the additional land (about 2.2% globally, upper bound estimate) would have relatively more impact on forests than on pastures. Although these estimates are in terms of productivity weighted land area, they indicate an expansion in cropland of between 17.9 and 26.7 million hectares, of which 12.0–17.7 million hectares would have been in developing

Table 3.4. Comparison of results from the Evenson and Rosegrant (2003) and Villoria (2011) estimations.

	Evenson and Rosegrant (2003) Model: IMPACT Time period: 1965–2000 TFP shock to CGIAR crops:[a] – 0.72% per year (weighted average); – 32.2 total TFP shock)			**Villoria (2011)** Model: GTAP-AEZ Time period: 1965–2004 Yield shock to CGIAR crops:a – 0.72% per year (weighted average); – 32.2 total TFP shock		
	Wheat	**Rice**	**Maize**	**Wheat**	**Rice**	**Coarse grains[b]**
Prices (% change, lower bound)	+ 29	+ 80	+ 23	+ 29	+ 68	+ 20
Area (% change in global harvested area, lower bound)	+ 3.2	+ 7.5	+ 1.1	+ 9.4	+ 20	+ 8

a CGIAR crops represented in this weighted TFP shock include: wheat, rice, maize, sorghum, millet, barley, dry beans, lentils, cassava and potato.

b GTAP-AEZ model includes the following CGIAR crops under this grouping: maize, barley, millet and sorghum.

countries.[49] This estimate is similar to that of Evenson and Rosegrant (2003) of 24 to 32 million hectares globally, although their estimate does not take into account the area effects on other crops (and thus may not be strictly comparable with the GTAP model estimates). Note that the CGIAR can only claim a portion of this saving, since crop germplasm improvement is the result of both CGIAR and national system investments in crop improvement. An alternative counterfactual relating to assumptions about the specific contribution of the International Agricultural Research Centers of the CGIAR to crop germplasm improvement is presented in Villoria (2011).

The simulation results from GTAP-AEZ demonstrate that for the staple food crops, as expected, the Borlaug hypothesis prevails – land is saved as a result of the global crop germplasm improvement and subsequent increases in yield that have taken place since 1965. These estimates are orders of magnitude lower than predicted by the simple global food equation (discussed in section 3.1.1) that does not take account of feedback loops through prices of products and land. These lower net land-saving effects may still represent a significant positive impact of agricultural research on the environment. However, the overall effects on land saving are dwarfed by the effects of crop germplasm improvement on food prices. In the absence of crop germplasm improvement in developing countries, increases in food prices of the order predicted by both GTAP and IMPACT would have serious implications for poverty reduction and malnutrition.

3.3.4 Limitations

Overall each generation of estimates of land savings from CGIAR crop intensification has incorporated additional impact pathways through more complex modeling. GTAP-AEZ is one of a number of global economic models of land-use change (reviewed by Hertel et al., 2009) but most others such as IMPACT (Rosegrant, 2002), WATSIM (Kuhn, 2003), AgLU (Sands and Leimbach, 2003) and FASOM (Adams et al., 1996; USEPA, 2005) are partial equilibrium models that do not consider impacts through economy-wide effects, and most importantly for this study, through land market effects.

Table 3.5. Percentage change in land cover assuming no crop germplasm improvement-related productivity gains in CGIAR crops since 1965, GTAP-AEZ estimates.

	Cropland	Forests	Pasture
Developing countries	1.52	– 0.86	– 0.66
Developed countries	0.87	– 0.51	– 0.36

Very important policy questions regarding the land-use change impacts of alternative policies are being asked of models such as GTAP-AEZ. The effects of biofuel mandates on land use are a particularly prominent example, with much of the recent literature on land-use change being devoted to the question of whether biofuels actually deliver net benefits in terms of GHG emissions when indirect land-use change from higher prices is factored in (e.g. Searchinger, 2008; and Keeney and Hertel, 2009 which critiques the Searchinger paper).

Nonetheless, the introduction of land heterogeneity (agroecological zones), pasture and forest land use, and land markets into CGE models is a relatively new enterprise. As such, most of the modeling assumptions need to be validated against observed data. Two assumptions are particularly critical. From the perspective of the demand for land, GTAP-AEZ assumes that there is only one national production function for each crop. From the supply side, GTAP-AEZ assumes a CET functional form to determine the transformation of land across different uses in crops, pastures and forests, which may not be realistic. There are few empirical estimates of these elasticities and they are likely to vary across factor endowments and institutional settings. Thus, this assumption should be improved through new empirical evidence to estimate elasticities in different contexts, as well as further advancement in the modeling.

49 About 9–13 million hectares of these would be in Asia, 2–3 million hectares in sub-Saharan Africa, 1–2 million hectares in Latin America, and the remainder in the Middle-East and North Africa region.

The assumptions can be improved through new empirical evidence on particular elasticities in different contexts, as well as further advances in modeling. At the moment, our theoretical understanding of land-use change issues is somewhat ahead of our abilities to make empirical estimates. However, no matter how good the state of the art of modeling becomes, we will always be constrained by the inevitably high degree of uncertainty that we have about the extent to which a model can generate a sound counterfactual back-cast over such a long time period.

Decomposing the impacts on production and trade of the productivity shock to developing country agriculture,[50] GTAP-AEZ estimates that developing countries would have imported 111% more wheat. Recall that crop germplasm improvement occurs at its historical rate in developed countries – though in fact several studies have documented the significant positive spillover effects of CGIAR research on developed countries (e.g. Pardey et al., 1996). This raises the question of whether this counterfactual scenario would ever have been allowed to play out in practice. It is valid to ask whether the purely economic counterfactuals presented here are ever likely to have occurred from a political perspective: government policy will often exert more influence on outcomes, particularly when related to food concerns, than economic rationality might dictate based on open trade models.

Since policy environment is not integrated into the GTAP-AEZ modeling exercise, the counterfactual world (i.e. the world that is not benefiting from crop germplasm improvement in key food security crops) predicted by the model would have more people living in poverty and more people going hungry. This predicted equilibrium state of increased poverty and hunger without the benefits of crop germplasm improvement research assumes no government actions to increase food production, especially by clearing land. The inability of the GTAP-AEZ model to account for policy response suggests that the land saving effects predicted (18–27 million hectares globally) are lower bound estimates of the true effect.[51]

3.4 Expansion at the agriculture–forest frontier

Agriculture competes for land with forest, other kinds of natural ecosystems and urban areas. Most countries have followed a development path that has resulted in significant loss of forest cover from their initial endowment, with conversion to agriculture the main driver of deforestation and land-cover change. Since 1850, 600 million hectares of forest and 470 million hectares of savannah have been converted to agriculture, and yet many developing countries are far from a discernible turning point in their land-use transitions (Geist, 2001). The land-use transition theory (Mather, 1992; Grainger, 1995; Mather and Needle, 1998) describes the long-run reduction in the percentage of land area under forest experienced by every country in which there was a majority of forest cover before human settlement.

Some have argued that deforestation is an inevitable feature of national development if a country is to experience a growth in population and living standards (Grainger et al., 2003). However, the multiple failures of markets, institutions and policies that drive deforestation result in great inefficiencies of resource use and inequities in development outcomes. Research that aims to improve forest policy and governance,

50 Remember that developed countries in this counterfactual still benefit from the level of crop germplasm improvement we observe historically.

51 This discussion about modelling a realistic counterfactual also has implications for estimating other types of impacts of crop germplasm improvement research. For example, poverty impacts of technological change are often modeled through effects on food prices. The claim being made in such analysis is that without productivity gains (i.e. the counterfactual world), food prices would have been very high, resulting in lower real incomes and consequently increased poverty. This is the same counterfactual assumption of the GTAP-AEZ model that ignores policy response by government. Thus one should keep in mind that the criticism that the counterfactual world predicted by the GTAP model is based on unrealistic assumption of no policy response also applies to other models used by economists to estimate poverty impacts of research. In other words, if we believe that a counterfactual scenario of government inaction would not have existed and governments would have done all they could to curtail food prices in the absence of productivity increases, then one cannot claim both a higher level of land saving as well as higher poverty reduction as a result of the same type of research.

including a significant body carried out by CGIAR centers, can help to address these failures, despite little documentation of such research having a significant impact (although Raitzer's 2008 study is an important exception). A 'forest transition' (Chomitz, 2007) takes place in those countries where a long-run trend of deforestation is stabilized and then reversed through net increases in forest area as a result of afforestation or reforestation.

Modern informational tools allow more precise measures of areas deforested, as well as the causes. Using satellite imagery, Gibbs et al. (2010) found that the total agricultural area in tropical countries increased by more than 75 million hectares during 1980s and 1990s. Of this expansion, more than half (55%) occurred by clearing intact, natural forest, and a further 27% came from expansion into 'disturbed forest.' Moreover, deforestation has been concentrated in a few countries. Hansen et al. (2008), again using satellite imagery, estimate that almost half (48%) of all humid tropical forest clearing from 2000 to 2004 occurred in Brazil, followed by 12% in Indonesia. Both countries are considered 'hotspots' for tropical deforestation. Rising interest in global GHG emissions has put a spotlight on the role of agriculture in tropical deforestation (Burney et al., 2010; West et al., 2010).

Studies at lower levels of aggregation also support the role of agriculture in deforestation. In an authoritative meta-analysis of 152 studies at the sub-national level on the causes of deforestation, agricultural expansion was identified as a proximate cause in almost all (96%) of cases (Geist and Lambin, 2001). Very little deforestation occurs without agricultural expansion (for crops and cattle), although there are usually a number of simultaneous causes operating together which constitute a limited number of 'syndromes' from around the world.

- Loss of land productivity on sensitive areas following inappropriate use.
- Deforestation on forest frontiers by weak states for geopolitical reasons or to promote interest groups.
- The transition from communal to private land ownership in developing regions.
- Policy interventions that drive modifications of landscapes and ecosystems.
- Delayed and ineffective social responses to deteriorating environmental situations, combined with absence of political will to mitigate damage and to alter the trajectory of change.

From the 1960s to the 1980s, tropical deforestation for agriculture was driven by population growth as a growing number of farmers pushed further into the frontier in search of land to meet subsistence needs (Rudel et al., 2009). Subsequently, a slowing in the global population growth rate and rapid integration of the global economy has meant that expansion of commercial agriculture is now recognized as the main driver of deforestation (Lambin et al, 2001; Nepstad et al., 2006; Rudel et al., 2009a). This expansion is in turn driven by urbanization, rising incomes and increased trade flows (deFries et al., 2010). Africa is the only region where agricultural expansion is still primarily driven by population growth (Chomitz, 2007).

Although agricultural expansion may be the proximate cause of deforestation, meta-analyses of over 140 studies have identified three groups of factors as the primary drivers: (1) commodity prices, (2) construction of roads, and (3) low wages or high unemployment (Angelsen, 2010). These factors are in turn strongly influenced by property rights and governance of forest resources (Chomitz, 2007).

Although agriculture is important to deforestation, the corollary does not hold. From 1985 to 2004, crop and livestock production in developing countries grew by 3.3–3.4% per annum, whereas gross annual deforestation (1990–2005) represents only approximately 0.3% of total agricultural area (Angelsen, 2010). This suggests that the vast majority of the increase in agricultural output came from sources other than simple expansion into forests.

3.4.1 Recent market-led commodity expansion

A handful of commodities have been associated with recent land expansion. If we consider the 10 most important commodities of the past 20 years in terms of area expansion in developing countries only, oilseeds (led by soybeans and oil palm) and cereals (rice, wheat and maize) dominate. Table 6 shows that average yields have increased across the board, whether the crop is increasing or decreasing in area, which suggests that other factors are more important in determining land-use change than technology (Rudel et al., 2009b). Beyond crop area expansion, the expansion of pastures for cattle has been significant, growing by 12 million hectares in Brazil alone over the past 20 years. Plantation forestry has also been a significant factor.

These changes in crop composition mostly reflect shifts in diets towards vegetable oils and livestock products driven by high, sustained rates of economic growth in countries like China over the past two decades. A major proportion of soybeans and maize production is destined for animal feed. Moreover, growing proportions of global soybean, maize, rapeseed, oil palm and sugar cane production are being diverted to fulfill government biofuel mandates (e.g. in the United States, European Union and Brazil). Some of this expansion is taking place directly in developing countries, to grow biofuel crops, such as the rapid expansion of sugar cane (mostly in Brazil). But indirect effects of expansion of biofuel production in developed countries such as maize in the United States are likely to be significant (Hertel et al, 2010). Such expansion can cause knock-on effects or 'indirect land-use change' via impacts on commodity prices and the subsequent changes in land rents for these crops at the forest frontier, via higher prices for specific crops.

Over the past 20 years, the expansion of three commodities in particular – pastures, soybean, and oil palm – has intersected with tropical deforestation. Pastures are not included in Table 3.6 as the data on them held on FAOSTAT are of poor quality. However, were these data to be included, they suggest that in the region of 300 million hectares of pastures and meadows have been established in developing countries since 1990 (FAOSTAT data, using the same ranges and definitions as Table 3.6 below).

In this section we examine three commodity–country combinations with respect to evidence on the relative roles of policies and governance versus technologies as drivers of expansion at the forest margin. The three commodities and countries most often 'blamed' are: pastures/cattle in Brazil, soybeans in Brazil, and oil palm in Indonesia and Malaysia. Area growth on the frontier associated with these commodities has attracted much research on which this

Table 3.6. Top ten expanded and contracted crops, 1990–2007 (globally and for developing countries only),[a] and associated change in global yields.

Top 10 expanded crops						
Globally				Developing countries only		
Rank	Crop	Change in harvested area (M ha)	Change in yields (%)	Rank	Crop	Change in harvested area (M ha)
1	Soybeans[b]	36.9	27.8	1	Soybeans	30.9
2	Maize	23.9	35.1	2	Maize	18.9
3	Rapeseed	11.1	30.8	3	Wheat	12.2
4	Rice, paddy	9.4	20.1	4	Rice, paddy	10.6
5	Oil palm fruit	7.8	43.8	5	Oil palm fruit	7.8
6	Sunflower seed	6.9	– 2.4	6	Cow peas, dry	5.7
7	Cow peas, dry	5.7	16.3	7	Sugar cane	5.4
8	Sugar cane	5.5	14.2	8	Potatoes	4.3
9	Cassava	2.9	23.0	9	Seed cotton	3.3
10	Olives	2.9	18.1	10	Cassava	2.9

a The table gives a comparison of 3-year rolling averages for 1989–1991 versus 2006–2008. Developing countries are defined as all countries except for the United States, Canada, Europe and wealthy countries in Asia.

b In the case of soybeans, the 1990–2008 period contains two distinct decades of very different dynamics. The majority of the observed yield increases occurred in 1990–2000 (a 19% average yield increase in the decade globally; a 44% average yield increase in Brazil), with comparatively little yield growth subsequently (6% increase globally in 2000–2008; 8.5% yield increase in Brazil in 2000–2008). In the case of Brazil, the soybean area increase shows the opposite trend (22% increase in area harvested in 1990–2000; 56% increase in area harvested in 2000–2008).

review draws. These changes are also highly relevant to the CGIAR which has through CIAT (pastures) and CIFOR carried out considerable research on the expansion of these commodities (Sheil et al., 2009; Barona et al., 2010; Pacheco et al., 2011).

Pastures in Brazil

Pastures have expanded by over 40 million hectares in Brazil since 1970 as the cattle herd has more than doubled to over 200 million head to meet rising domestic and international markets. As a consequence of the rise to becoming the world's largest meat exporter, Brazil has been the world leader in tropical deforestation, with an average of about 2 million hectares per year cleared between 1996 and 2005 (Nepstad et al., 2009).

The frontier to the Brazilian Legal Amazon[52] since the 1970s has been open for claiming by ranchers, and from the 1970s to the early 2000s this process was supported by successive Brazilian governments. This support was either explicit in the form of government settlement programs designed to colonize the Amazon, or implicit through weak protection to the region's forests.

Under these circumstances it is not surprising that the beef production system in Brazil has historically been very extensive with extremely poor productivity. There have been few incentives to encourage improved pasture adoption and intensification of grazing. Indeed incentives support expansion at the extensive margin. The Brazilian constitution authorizes reassignment of private lands to squatters if the land is not placed into productive use and, in practice, forest lands are not recognized as undergoing productive use (Araujo et al., 2009). The lowest-cost means of securing the land through productive use is conversion to pasture for livestock. Historically, the overriding concern of ranchers has been to stock a minimum number of cattle to ensure a level of property rights over public lands that have been appropriated, in the absence of suitable mechanisms for ensuring title and a functioning land market (e.g. Merry et al., 2008).

52 The Brazilian Legal Amazon consists of the following states: Acre, Amapá, Amazonas, Goias (north of 13oS), Rondônia, Roraima, Pará, Maranhão (west of 44oW), Tocantins and Mato Grosso.

In all, the evidence reviewed here suggests that improved pastures have not been a driver of deforestation in Brazil to date, as the economic and policy environment has favored extensive production, and this has dominated any technological impacts on land rents. However, there is recent evidence that intensification is now taking place. Pacheco and Poccard-Chapuis (2009) examined changes between two agricultural censuses in 1995/96 and 2006. The growth in beef production during this period was composed of a simultaneous extensification (expansion of the area of cultivated pasture) and intensification (increase in the stocking rate) for the Legal Amazon as follows:

1995/96: 51 million hectares x 0.70 head / hectare = 35.7 million head of cattle
2006: 61 million hectares x 0.92 head / hectare = 56.1 million head of cattle

Pacheco and Poccard-Chapuis examined the counterfactual to the observed intensification process, pointing out that if the stocking ratio had stayed at the level of 1995/96, an additional 20 million hectares of pasture would have been needed to produce the same number of cattle. What is not clear is the extent to which improved pastures have supported this increase in stocking density, and whether improved pastures may also have increased the rate of expansion by raising land rents.

Four factors now favor intensification of livestock in addition to any role for improved pastures. First, soybean has much higher gross margins per hectare than livestock ranching (as outlined below), so is out-competing the more extensive cattle operations in areas where crop production is viable. Second, commercial feedstuffs from by-products from industrial agricultural processing have become commercially available and allow for supplemental feeding at low cost. Third, there is a market-pull effect as both the soy and beef industry attempt to exclude products from newly deforested land from their export supply chains (Nepstad et al., 2009), partly as a response to significant consumer-awareness campaigns by NGOs such as Greenpeace. Finally, forest governance in the Amazon is improving, in tandem with significantly improved satellite monitoring of new forest clearance, leading to a gradual closing of the forest frontier. With better protection of the Amazon, total area under pasture shows a slight decrease in the 2000s (IBGE - Instituto Brasileiro de Geografia e Estatística website). Deforestation in the Amazon has also slowed dramatically in the period 2005–2010 although pasture remains the main source of new deforestation.

Soybean in Brazil

Brazil is now the world's second largest producer of soybean. Production in Brazil has nearly quadrupled since 1980, and the country now accounts for nearly a third of the world's soybean exports (ranking second behind the United States). Although these figures are impressive, the abundance of Brazilian soybeans has come at the expense of pastures as well as millions of hectares of natural vegetation lost over the course of the past three decades. Soybean area in Brazil rose from 8.8 million hectares in 1980 to 21.3 million hectares in 2008. Since 1990, the fastest-growing area for soybean production has been in the Cerrado, a frontier area of natural savannah and woodlands. In Mato Grosso, the largest state in the Cerrado, cropland expansion (mainly for soybean) into forested areas contributed an average of 17% of the total direct forest loss between 2000 and 2004 (Morton et al., 2006). Most soybean replaced pastures which were directly responsible for over 60% of the area deforested (Figure 3.4).

There has been a gradual move northward of this soybean production frontier, from the Cerrado to the fringes of the Amazon (Barona et al., 2010). The sequencing of deforestation in the highly mechanized soy industry in Brazil, depicted by the diagram from Morton below, is a simplification of a carefully organized agronomic sequence that maximizes the value from newly cleared land. First, the vegetation is cleared, then the highest quality timber is sold for lumber and the remaining biomass burnt off during the dry season (Cassman, 2005). A rice crop that can tolerate acidic soil conditions often follows, and soybean is then planted after soil amendments have been applied. The total transition time is often of the range of 5 to 10 years, so while it appears that soybean replaces pastures, it is really the gradual replacement of forest with soybean.

In addition to the gradual, staged process of deforestation by soy expansion at the frontier, there is also an indirect impact from soybean expanding into pasture area, and ranchers then moving to the frontier and creating new pasture areas. This hypothesized 'displacement deforestation' (Barona et al., 2010) is an example of indirect land-use change from soybean expansion, and is analogous to the phenomenon that has gained a lot of attention in recent years with regards to the indirect land-use impacts of biofuels (Searchinger, 2008; Hertel et al., 2010).

Measuring indirect land-use change is challenging given the spatial displacement of causality (Babcock, 2009). Arima et al. (2011) is the first example of a novel combination of GIS and spatial statistics (the spatial Durbin model – SDM; LeSage and Pace, 2009) that links frontier deforestation to the expansion of soybean production in a settled agricultural area away from the frontier. Using data from 2003–2008 from

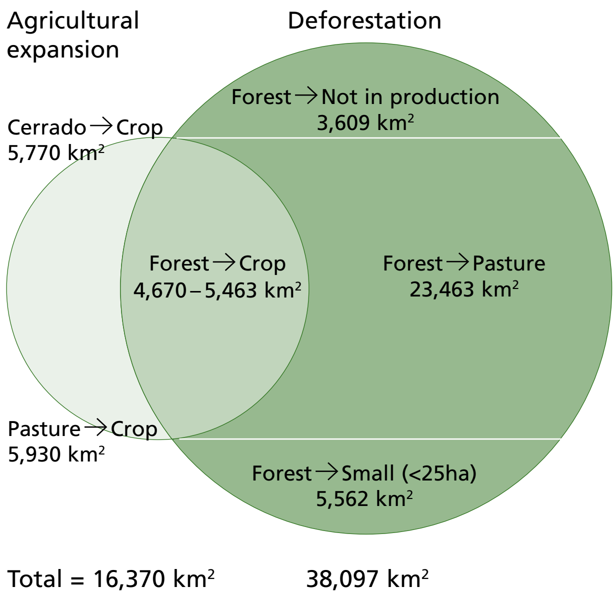

Figure 3.4. Cropland expansion and deforestation in Mato Grosso, Brazil 2001–2004 (Morton et al., 2006).

761 municipios in the Legal Amazon, the authors find that deforestation in the forest frontier is strongly related to soybean expansion in settled agricultural areas. Different statistical models (ordinary least squares – OLS; fixed effects without time lag for soybean – FE1; fixed effects with time lag for soybean – FE2) generate different elasticity estimates for the impact of a 1% reduction in soybean expansion in the settled agricultural area on the percentage reduction in deforestation at the frontier. The OLS estimate is 0.6%, FE1 is 1.2% and FE2 is 4%, all suggesting a substantial and important indirect impact of soybean cultivation on deforestation in the Amazon.

Barros et al. (2007), describe the factors behind the opening up and settling of the Cerrado in the 1980s and 1990s, which in turn facilitated soybean area expansion in Brazil over this period. Behind the more proximate factors was a national policy giving significant support to development in the Cerrado. Thus, subsidized credit, investments in transportation and storage facilities, energy, extension, rural electrification and mechanization were all supported by government policy.

Technology was also a critical factor in the expansion. Research by EMBRAPA, the widely respected Brazilian Agricultural Research Corporation, initially focused on agronomic methods for soil correction in the Cerrado using lime, fertilizer and micronutrients. Subsequently, EMBRAPA's research on soybean since 1975 has focused on the adaptation of cultivars to day-length at lower latitudes, and to resistance to diseases and pests in this warmer and more humid environment (Barros et al., 2007). Between 1990 and 2010, soybean yields in the Cerrado states increased from 2.0 t/ha to 3.0 t/ha in Mato Grosso and from 1.6 t/ha to 3.1 t/ha in Mato Grosso do Sul (data from IBGE website, August 2010). Since 2000, genetically modified herbicide-tolerant soybean varieties ('Roundup Ready' – resistant to Glyphosate-based herbicides) have expanded rapidly to reach 70% of total production in Brazil and have greatly facilitated the adoption of cost-reducing zero tillage (Tollefson, 2010).

Availability of more adaptable, higher-yielding and cost-saving soybean technologies in Brazil has undoubtedly exerted direct pressure on agricultural land expansion. Land prices have risen sharply, consistent with expansion based on land rental value associated with adoption of new technologies (Nepstad et al., 2006; Sauer, 2011). Ferraz (2001) found that expansion of crop area was determined by changes in land prices (likely related to improved technologies), government agriculture credit and roads. His findings are consistent with others in concluding that a combination of government policies and technologies encouraged expansion.

Brazil is a major player in global soybean markets, accounting for 31% of exports,[53] so increased production as a result of the adoption of new technologies would have depressed the world price for soybeans and at least partly offset the effect of technology on land rents. The only empirical study to look specifically at the effect of technological change in agriculture on deforestation in Brazil in a general equilibrium market context is Cattaneo (2001), who finds that technological change in soybean increases deforestation. Importantly, however, the model does not allow higher Brazilian production to feed back into international soybean prices (i.e. it makes the 'small country' assumption hardly appropriate for Brazilian soybeans) which precludes the possibility of a long-run land-saving effect, which we model in section 3.4.2.

It would be unfair to attribute too much of the negative impact (in terms of biodiversity and GHG emissions) resulting from this spatial shift north towards the Amazon to the profitability and feasibility of soybean cultivation (rather than, for example, inadequate forest governance and the speculative land clearance behavior of ranchers). However, it is valid to ask whether this shift would have taken place in the absence of technological change in soybean production. The fact that the new soybean varieties had a spatial bias towards extending the range of soybean farming northwards suggests that technology has certainly been a factor pushing out the land frontier, some of it at the expense of tropical forests. But this must be balanced against the even

53 FAOSTAT data, 3-year rolling average 2006–2008, soybean exports by value.

higher prices for soybean that would have prevailed in the absence of technological change, which may have also stimulated soybean expansion. Evidence supports a relatively elastic acreage response of soybean area to future prices of approximately 1.0 in the Cerrado states, and 0.6 nationally. The elasticity for total crop area response to soybean futures price is approximately 0.3 but higher at the forest margin (Almirall, 2009). Ultimately, the unrelenting growth in demand for soybean is the main underlying driver, swamping the technological effects in its importance.

Oil palm in Indonesia and Malaysia
Oil palm is especially suited for growing in the humid tropics with a high overlap with tropical humid forests that are valued for their unique biodiversity and for mitigation of climate change. For this reason, the CGIAR has had research interests in oil palm, largely from the perspective of forest policy (e.g. Danielsen et al., 2008; Schoneveld, 2010).

Malaysia pioneered the commercial oil palm industry (Martin, 2003; Rasiah, 2006). With rising land and labor costs, the industry moved to neighboring Indonesia, which at 16.9 million tons in 2008 is now the world's largest producer, slightly ahead of Malaysia (15.8 million tonnes). Together Malaysia and Indonesia now account for over 85% of global palm oil production. Planted area in Indonesia increased five times between 1991 and 2008, from about 1.3 million hectares to 6.3 million hectares, although some estimates are now more in the region of 8–10 million hectares.[54] Investment by large companies in mills and an associated production feedstock area has spurred this expansion.

The oil palm sector has been criticized for being a major contributor to deforestation and GHG emissions. Land-use change and deforestation are the largest single contributors to Indonesia's GHG emissions. Some 4.2 million hectares (approximately 70% of Indonesia's oil palm plantations) is converted forest estate land (defined as land under the control of the Forestry Department, however not all of this is actually forest) (World Bank, 2010). Accounting for crop substitution, Koh and Wilcove (2008) estimated that forest land accounted for 55–59% of expansion in Malaysia and 56% of the much larger expansion in Indonesia.[55] However, even ignoring crop substitution, oil palm could not have accounted for more than 10% of forest loss in Indonesia, where arable land and area under oil palm each expanded by about 3.8 million hectares from 1991–2007, but 30 million hectares of forest area was lost overall.

Poor forest governance is a major factor in forest loss in Indonesia. To help expand production, the Indonesian government provided land, in many cases still forested, for nominal fees. Timber sales were often used to finance planting and oil palm establishment. Nonetheless, a considerable area of forest land – up to 12 million hectares by some estimates – has been allocated to oil palm and deforested but not planted (Fargione et al., 2008; Sheil et al., 2009; Friends of the Earth, 2009). Many companies allegedly use fictitious palm oil schemes to obtain logging licenses without ever establishing oil palm estates. Casson (1999) found that only 1.4 million hectares of 9 million hectares of concessions had been developed by 1998. The main motivation for this forest loss has been timber extraction, as it has been easier to obtain concessions for forest lands for oil palm than for logging.

There is little sign that oil palm expansion is slowing. Vegetable oil markets remain buoyant and demand for feedstocks for biodiesel is picking up. The implications for GHG emissions and biodiversity loss of establishing oil palm in forested areas suggest that ways to improve productivity on already-cultivated land are of particular relevance in relieving pressure on forests. However, higher yields will also improve oil palm profitability and provide further in-

54 http://news.mongabay.com/2009/1202-indonesia_palm_oil.html – Accessed 3rd August 2011.

55 In a closer look based on satellite imagery, WWF-Indonesia (2008) reported that in the province of Riau, Sumatra, there was a 4.2 million hectare decrease in forest area (nearly all above 40 % canopy closure) from 1981 to 2007, equivalent to a 65% decline in forest area. Oil palm accounted for one third of this conversion with large shares converted by timber extraction and conversion to waste land.

centives to expand at the margin without proper safeguards for forest resources (Swarna Natha and Tisdell, 2009).

Yields of palm oil have been stagnant in Indonesia.[56] Even with good prospects to increase oil palm yields through improved management,[57] it is unlikely that higher yields will be achieved while incentives that provide cheap land relative to its true opportunity costs encourage area expansion rather than intensification. A number of policy reforms could help internalize the costs of land expansion and encourage intensification.

- *Market certification.* Responding to the controversies around oil palm and its threat to tropical forests, the industry initiated the Round Table on Sustainable Palm Oil in 2004 to develop and implement palm oil certification. Certification bans plantings that "replace primary forest or any area containing one or more High Conservation Values."
- *Payments for environmental services.* The valuation of carbon sequestration in tropical forests and the potential of REDD+ to compete with oil palm has been the subject of several studies. Although REDD+ does reduce the incentives to clear forests, it is by no means clear that it will be enough to compete with oil palm, except in peat lands. Much depends on the price of carbon, which varies widely by market segment. One recent study estimates that a carbon price of US$22 per tonne, well above current market values, would be needed to make forest conservation competitive with oil palm (World Bank, 2010).
- *Regulation through land-use zoning*. Environmental costs can be reduced by developing oil palm on degraded forests and 'imperata' ('alang alang') grasslands usually portrayed as unproductive wasteland. Costs of establishing oil palm on these lands are much lower than on forest lands, and yields are indistinguishable from those on forest land (Fairhurst and McLaughlin, 2009). However, as local people and communities may already use degraded lands, bringing these into production will require recognizing such rights and negotiating and sharing benefits with locals. NGOs are implementing demonstration activities that can provide important lessons. Guiding this expansion process towards land of lower ecological value represents a significant and important governance challenge in a country with a poor governance track record.

Research by CIFOR on expansion of plantation forestry in Sumatra has demonstrated that better implementation of existing policies and regulations can significantly slow deforestation (Raitzer, 2010). Given that oil palm expansion is governed by similar policies, it is likely that these findings also hold for oil palm as well, and there have been important contributions from CIFOR, ICRAF and other centers to this literature and in policy circles (Palm et al., 2005; Andam et al., 2008; Shiel et al., 2009).

3.4.2 Intensification of oilcrops: a further application of the GTAP model

Soybean in Brazil and oil palm in Indonesia provide a good opportunity to test whether positive impacts on land rents from technological change at the country level might outweigh the land-saving effect of increasing yields. Although both Indonesia and Brazil are major exporters, they face highly elastic demand in world markets as a result of strong potential for substitution among vegetable oils and among exporters. To

56 This situation, of stagnant yields in the major producing countries in Southeast Asia, is still consistent with a global increase in yields between 1990 and 2007 (as shown in Table 3.5). This is due to a shift over time from area under oil palm being dominated in 1990 by sub-Saharan Africa (53% of total global area), to a situation where the higher-yield Southeast Asian countries dominated oil palm area by 2007 (64% of total global area). Over the same period, Southeast Asia yields have remained higher than those for sub-Saharan Africa by a factor of at least four (all calculations based on FAOSTAT data).

57 A variety of reasons explain yield gaps – nutrient management, harvesting time, age of plantation, canopy management, and weed and pest control. On smallholdings there is an additional yield gap due to use of poor genetic stock and inadequate fertilizer. Recent initiatives are testing Best Management Practices on a commercial scale. Initial results have achieved an average of 7 t/ha in 2007 although this is only 15% above previous yields on those plantations (Donough, undated). For the medium term, there is potential to exploit the yield gap between current yields and what could be economically attained. Jalani et al. (2002) puts attainable yields of 6.3–9.5 t/ha. Shiel et al. (2009) note commercial potential of 6–7 t/ha, indicating a yield gap of 40-50%. The best-managed plantations are already obtaining yields of 6.5–7.5 t/ha (Wahid, 2004) and up to 10 t/ha have been achieved on commercially managed plots.

estimate the magnitude of the two effects, we again applied the GTAP-AEZ model. In the case of soybeans, yields increased by 57% in Brazil from 1990 to 2004[58] and most evidence indicates that gains in TFP are at least of this level.

Applying the counterfactual of no TFP increase in Brazil (although it would have changed at observed rates in other parts of the world), area under soybean in Brazil declines by 18 percent and production by 67%[59] due to lower profitability and reduced land rents. However, the decline in rental price of land for soybean encourages expansion of other activities (such as production of rice and other coarse grains). As a result, the cropland overall declines modestly by about 300,000 hectares with forest area increasing by 0.1% and pastures by 0.13%.

Within Brazil, the positive effect on land rents therefore dominates the land-saving effect. However, despite a highly elastic export demand for soybeans for world markets in general equilibrium (-2.5), cropland in the rest of the world expands (1.5 million hectares) as a result of higher soybean prices (2%) with an overall crop area expansion globally of about 1.2 million hectares. Thus globally the price effect of lower yields in Brazil considerably outweighs the depressed area in Brazil due to the land rent effect. This global result is consistent with the results for the cereal crops discussed in section 3.3.

Similar results were obtained for oil palm. However, in this case, yields have been stagnant in Indonesia and Malaysia for the period of review,[60] so we simulated the effects of a 57% TFP increase[61] (the mirror image of the soybean case). That is, we ask what would have happened to land use, had oil palm productivity increased at the rate of soybean in Brazil? This is an important question given recent calls to increase oil palm yields to save forests (e.g. Koh, 2007). With higher land rents due to technological change, crop area in Indonesia and Malaysia expands by 100,000 hectares. All of this expansion is from forest land, thus reinforcing the importance of the impact on land rents. However, crop area globally falls by 500,000 hectares, and is partly replaced by forests. Of course, these estimates do not take into account the relative value of highly biodiverse tropical forests in Indonesia and Malaysia relative to forests elsewhere that may be of lower ecological value.

The bottom line is that even with highly elastic demand, the effects of technological change on land rents locally are more than offset by the land-saving impacts worldwide in a globally integrated market economy that allows for trade. Given that we examined technological changes on the frontier in specific countries, the results suggest that land-saving effects of technological change will usually dominate land rent effects when viewed from a global perspective. However, where the expansion and where the contraction takes place is of critical importance from the perspectives of biodiversity conservation and GHG emissions. A hypothetical situation in which a net reduction in global agricultural area actually comprises an expansion into rainforests in the tropics and a contraction in agricultural areas, giving way to grasslands, in the United States could not be claimed as a victory for technological change. Large, biologically significant regions (Sumatra, Borneo and the Amazon) can be irreversibly devastated, and incur social costs that may far outweigh the benefits from net land savings globally.

3.5. Implications for the CGIAR

3.5.1 Higher agriculture yields: necessary but not sufficient for saving forests

Clearly, raising through successful research the aggregate supply in the breadbasket regions of virtually all of the CGIAR-mandated crops such as rice, wheat, maize, sorghum

58 This translates to a 55% shock variable after adjustment of the TFP with the market share of soybean in Brazil's oilseed market.

59 It should be remembered that soybean belongs to the oilseed aggregate commodity category in GTAP, and in reality the discussion of land saving or expansion from Viloria's results refer to oilseeds rather than 'soybean' *per se*.

60 Table 3.5 shows oil palm as one of the crops with the highest global yield increases (43%) in the period 1990–2008, but this is largely an artifact of the shift in production from low-yield African countries to relatively higher yielding Southeast Asia. However, over the same period (1990–2008), oil palm yields in Indonesia increased by only 3.5% in total (FAOSTAT data).

61 After accounting for the share of oil palm in the 'oilseeds sector' this represents about 46% TFP shock in this region.

and millet is likely to contribute to reducing agricultural expansion and forest loss. However, the magnitude of the effect is likely to be much less than commonly cited from the simplistic application of the method used by Borlaug.[62] Our best estimate is that the net global land savings from research on crop germplasm improvement for food crops in developing countries over the period 1965–2004 is of the order of 30 million hectares, after accounting for the price effect on land rents, consumption, trade, and substitution effects of other commodities. Although it must be noted that this estimate is based on the assumption that government policies do not differ in response to the higher food prices (i.e. there is no organized state-led initiative of additional land-clearing). Whereas the 30 million ha estimate is significant, the impacts are likely small relative to the huge impacts of the same research on lowering food prices and ultimately reducing poverty and hunger.

We need also to recognize that research that improves the profitability of agriculture specific to places with large areas of remaining forests may promote greater deforestation by raising the returns to land in agricultural uses relative to returns to forest uses. In the absence of strict control over land-use change, increases in productivity of crops such as pastures, tropical forest plantations, rubber, cacao, coffee and other tropical tree crops are likely to add pressure on forests.

It is important that claims of the land-saving effects of new technologies be carefully scrutinized, especially as many scientists continue to argue that they are saving forests through intensification (Gockowski and Sonwa, 2010, is a very recent example for cocoa research in West Africa), and improved agricultural technologies are one of the most common mechanisms proposed for making REDD+ work (along with protected areas and community or local forest management). In particular, it is critical to distinguish between adoption of new technologies over large areas of intensive agriculture (e.g. Green Revolution technology), and adoption of technologies in frontier areas, which contribute a relatively small share of total global production but may have significant effects on local forests (though small globally). Technologies that improve the productivity of traditional agricultural regions and that are relatively labor intensive hold the most promise for saving land and reducing deforestation.

In particular, three critical factors influence whether new agricultural technologies reduce or increase pressure on forests: the location of production; the characteristics of the technological change (in particular, whether it is labor saving); and the demand elasticity for the agricultural product in question.

Technologies that are predominantly adopted at or close to the forest margin and that are for a good with elastic demand on export markets will likely add to the pressure on the forest. Under these criteria, technological change in oil palm looks likely to induce further expansion, as oil palm production is located in forest areas and there is potentially unlimited demand. Technologies in crops with inelastic demand, and which are predominantly adopted away from the forest margin, will likely save land. Many of the CGIAR's mandate crops fit this description.

If the agricultural technology in question is labor saving, people may lose their livelihoods and have an incentive to move to the forest margin to try and clear some land for themselves. Labor-intensive technologies adopted away from the forest margin may actually draw people away from the forest margin uplands to work in the lowlands, thus reducing forest clearing in the process (as modeled for the Philippines by Shively, 2001).

62 It should be pointed out that the estimates by Borlaug refer to a very different counterfactual scenario than what is reflected in the GTAP-AEZ exercise. First, in the GTAP model, the counterfactual only excludes productivity growth from crop germplasm improvement research (but assumes other sources of yield growth would have continued as observed), whereas in Borlaug's calculations, the counterfactual excludes all the sources of yield growth (i.e. assumes zero yield growth). Thus, the shocks used in the GTAP model are of a much smaller magnitude than those used by Borlaug. Second, the counterfactual in the simplistic analysis estimates incremental area needed for the focused crops in the late 1990s to produce a late-1990s level of production at 1990s level of prices, whereas the GTAP model analysis estimates incremental area needed in the 1990s under all agricultural crops at different levels of food prices, food production, food consumption and international trade.

We should also recognize that the impact of technological change on land saving is likely to be a weak effect when compared with the range of other exogenous factors driving land-use change and deforestation. Even for rapidly expanding commodities on the forest margin, such as pastures, soybeans and oil palm, the effects of technological change through returns to land are likely to be much smaller than effects through better governance of land and forest resources. That is, expansion at the intensive margin through new technologies is unlikely to succeed if it is cheaper to expand at the extensive margin where forest land is readily available and poorly governed. Socially, of course, expansion at the extensive margin usually does not consider the real value of forest resources foregone. Recent experience with better governance and monitoring of the Brazilian Amazon has shown a dramatic drop in rates of deforestation, even as commodity prices have risen sharply in the past five years (Nepstad et al., 2009).

Land-cover change remains a dynamic process with a lot of potential for further deforestation to take place to meet the projected demands of a growing population, rising incomes and structural changes in diets, as well as new demands from biofuels (see Box 3.1). Conversion of natural grasslands and woodlands is likely to have lower costs in terms of ecosystem services foregone, than conversion of tropical forests with high conservation values, carbon storage and other services. Agroecological modeling of land suitability by IIASA (International Institute for Advanced Systems Analysis) has identified 1,210 million hectares of land that is still potentially suitable for conversion to rainfed agriculture (Table 3.7), even if the unculti-

Table 3.7. Existing land use and uncultivated areas of low population density suitable for cultivation – regional totals and individual countries with over 10 million hectares of non-forest or forest land suited to cultivation.

	Existing land cover		Uncultivated and suitable for cultivation	
	Forest area	Cultivated area	Forest area < 25 people/km²	Nonforest area < 25 people/km²
Sub-Saharan Africa	*509*	*210*	*163*	*201*
Sudan	9.9	16.3	3.9	46.0
DR Congo	148	14.8	75.8	22.5
Mozambique	24.4	5.7	8.3	16.3
Madagascar	12.7	3.5	2.4	16.2
Chad	2.3	7.7	0.7	14.8
Zambia	30.7	4.6	13.3	13.0
Congo Rep	23.1	0.5	12.4	3.5
Angola	57.9	2.9	11.5	9.7
Latin America	*934*	*162*	*291*	*123*
Brazil	485	62.3	130.8	45.5
Argentina	33.6	28.2	16.2	29.5
Peru	68.3	3.8	40.0	0.5
Colombia	64.5	7.3	31.3	5.0
Bolivia	54.3	2.9	21.0	8.3
Paraguay	19.1	5.4	10.3	7.3
Eastern Europe and Central Asia	*885*	*252*	*140*	*52.4*
Russia	808	120	129	38.4
East, South and Southeast Asia	494	445	46.3	14.3
Indonesia	95.7	32.9	24.8	10.5
Rest of World	*863*	*359*	*135*	*50.9*
Australia	88.1	45.7	17.0	26.2
United States	299	175	74.4	8.8
Canada	308	50.3	30.1	8.7
World Total	**3,706**	**1,503**	**775**	**446**

Source: Deininger and Byerlee (2011)

Box 3.1. Future land-use projections

Projections by the Food and Agriculture Organization of the United Nations (FAO) suggest that, until 2030, an additional 47 million hectares of land will be brought into production globally, comprising a decrease of 27 million hectares in developed and transition economies and an increase of 74 million hectares in developing countries. As cropping intensity is projected to increase as well, harvested area will expand even faster, by 92 million hectares, nearly all in developing countries where an annual expansion of 3 million hectares is predicted. Disaggregating across regions also illustrates that, while the rate of expansion will be slower than in 1990–2005, it will continue to be important factor in sub-Saharan Africa and Latin America where area under crops is expected to increase by 39 and 31 million hectares, respectively. These projections assume yield growth of 0.9% per year in line with recent experience. Importantly they do not consider land use for biofuels and forest plantations.

While these factors only extrapolate linearly, computable general equilibrium (CGE) models allow for adjustments to price and trade which induce supply responses in regions where land is relatively abundant. Doing so increases the magnitude of estimates, highlighting the conservative nature of the FAO estimates even for food and feed only. Compared to the 1.8 million hectares per year expansion predicted by FAO, other studies obtain much larger estimates of future land conversion for use by food commodities with annual values that range from 4.5 million hectares (Fischer et al., 2009) to 10 million hectares (Al Riffai et al., 2010) or even 12 million hectares (Eickhout et al., 2009). These estimates include the impacts for biofuels that are not considered in the FAO projections.

The impact of biofuels on land conversion depends not only on availability of second-generation technology but also on how strictly mandates will be enforced in light of increased evidence of high economic and environmental cost of strategies for biofuel expansion. Depending on these, the expected amount of land converted to biofuels until 2030 ranges between 18 and 44 million hectares (Fischer et al., 2009), a figure similar to that predicted by CGE models.

Although it has been one of the land-use categories with the fastest expansion over the past decades, none of the existing studies include plantation forestry. Doing so would be desirable as plantation forests are planted on marginal land, some of which is not suited to crop production, and may compete for pastureland. Estimated growth of this land-use category, between 42 and 84 million hectares in total (the latter based on continuation of past trends), can add significantly to total land demand (Carle and Holmgren, 2008). Unlike the other commodities, most of the area increase occurs in Asia and in developed and transition counties where agricultural area is projected to decline.

After accounting for projected yield growth, FAO projections for food crops are slightly below historical trends. By contrast, CGE-based models predict land-use changes that can be an order of magnitude above this figure. Adding biofuels adds roughly 1–2 million hectares per year. Plantation forestry could add some 1.5 million hectares per year, though part of the required land does not compete with crop uses. A conservative projection is that 6 million hectares of additional land will be brought into production annually up to 2030. This would imply a total expansion of land area of between 120 and 240 million hectares to 2030.

As land use in developed and transition countries is in long-term decline and as more agricultural activity shifts to developing countries, projected land-use changes in the latter are higher. Moreover, some two thirds of the land expansion in developing countries will be in sub-Saharan Africa and Latin America, the two regions in the developing world where land is still relatively abundant.

vated land is likely more marginal than currently farmed land – perhaps with a replacement value of around 0.7. Well over half of this is forested, with two thirds in tropical areas. However, around 450 million hectares is savannah or woodlands suited to crop agriculture, with two thirds of this located in sub-Saharan Africa and Latin America. The technological and governance challenge for humanity is how best to guide the seemingly inevitable continued agricultural expansion to the areas where the environmental costs will be lowest.

As a framework for achieving this governance challenge, Rudel (2009) argues persuasively for more place-based agricultural policies in preventing deforestation. In general, the principle is that policies should strengthen agriculture near major centers of population to encourage intensification rather than extensification of agriculture at a distance in response to rising demand from income and population growth.

This vision will, however, bump up against two major economic realities that will limit its political attractiveness to policy-makers. First, policies that concentrate on 'rewarding' landholders in favorable areas may be accused of being regressive and further marginalizing rural poor people. Second, with growing cities, the economic opportunity costs of farming in peri-urban areas becomes ever higher and agricultural land is subject to competition from non-agricultural uses, making the implementation of these policies more expensive.

3.5.2 Priorities for the CGIAR

What can the CGIAR do to ensure that it maximizes its potential positive impact on the issue of global land-use change? There are implications for both the generation of new agricultural technologies and for policy research. At the aggregate level, since food demand is generally inelastic, research on food staples should contribute to meeting growing food demand and forest conservation. Food-price effects and labor absorption resulting from technological change[63] are likely to reduce agricultural land use in forest margin areas. CGIAR technologies that are widely adopted in developing countries will typically have a large impact on market prices and this will reduce land expansion relative to what would otherwise occur.

Failure to move towards a sustainable agricultural intensification path would inevitably lead farmers to expand into fragile margins. But if the return on investment of new land clearing is perceived to be more productive than deepening the investment in the existing land (intensification), and there are relatively few impediments to opening up new land, then expansion occurs. This supports findings from a number of studies by the ASB partnership that indicate that intensification of agriculture is a necessary but not sufficient condition for forest protection (Palm et al., 2005; Minang, 2010). Land-saving technological change on existing agricultural lands needs to work alongside governance interventions such as forest protected areas (Nelson and Chomitz, 2009). Incentive systems such as REDD+ may ensure a win-win model of maximizing agricultural productivity and maximizing biodiversity conservation. However, tradeoffs between these two objectives are a more likely scenario (Lee and Barrett, 2001) and a priority for the CGIAR should be to help analyze such tradeoffs and promote policy dialogue around them.

CGIAR research on land-use and forest policies could potentially have even larger impacts on saving land and forests. CGIAR scientists are already very influential in the literature on land-use change. Arild Angelsen carried out much of his research on this topic while he was at CIFOR, and there remains strong research interest at that center on the relationship between agriculture expansion and deforestation (e.g. Shiels et al., 2009; Schoenveld, 2010; Pacheco et al., 2011). IFPRI scientists also contribute their expertise to research studies on, for example, the question of whether forest protected areas are actually effective (Andam et al., 2008). The ASB partnership for the tropical forest margins also has research and outreach interests in this area (Minang, 2010). This is an area in which the CGIAR has a comparative advantage, reaching across agricultural and

63 Labor shortages and/or higher wages constrain any expansion, but labor-saving technologies will foster greater migration to the frontier.

forestry expertise, and the forthcoming Consortium Research Program portfolio should ensure that these interests flourish.

Improving our understanding of the links between agricultural expansion and forests will not only depend on better micro-level studies, but also on better macro models, especially since international trade features strongly in many of the 'blame commodities.' The models presented in this review help in thinking about the range of possible pathways between agricultural research and agricultural expansion, but the uncertainty about many of the parameter estimates, and the occasionally ad-hoc assumptions required to connect global land-use data and global economy models, suggest that we are still some way away from developing robust models and parameters that capture the complexities of these pathways. Fortunately, this is a very dynamic literature driven largely from outside the CGIAR community. If CGIAR scientists can develop appropriate partnerships, they will have much better models and a broader empirical base on which to draw for future assessments of their research impacts.

In general, the CGIAR needs more analysis of the impact of its research in contexts fraught with market, policy and institutional failures. The payoffs, in terms of rates of return from only a fraction of the policy research carried out by the CGIAR, could justify the investment in a broad but inherently risky portfolio.

More generally, the CGIAR needs to better position itself with respect to the global debate on land resources including the extent of land scarcity, the synergies and tradeoffs between agricultural and forest land uses, and the recent rising global interest in private investment in farmland in land-abundant countries of Africa and Latin America. Much of this work is being led from outside the CGIAR, but has major implications for how the CGIAR sets its priorities. Through partnership with leading think-tanks in this area, the CGIAR should be able to tap into a rapidly expanding knowledge base.

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Annex to chapter 3

Full tables from Villoria (2011)

1.1. Crop germplasm improvement contributions to yield growth (1965–2004).

Annual percentage contribution of crop germplasm improvement to total factor productivity (TFP) growth (Source: Evenson, 2003, table 22.9, p.466-467).

Crop	Regions	Crop germplasm improvement
All crops	**All regions**	**0.72**
	Asia	0.88
	Latin America	0.66
	MENA	0.69
	SS Africa	0.28
Barley	MENA	0.49
Beans	All regions	0.21
	Latin America	0.22
	SS Africa	0.18
Cassava	All regions	0.22
	Asia	0.17
	Latin America	0.10
	SS Africa	0.25
Lentils	MENA	0.28
Maize	All regions	0.66
	Asia	0.96
	Latin America	0.62
	SS Africa	0.22
Millets	All regions	0.56
	Asia	1.04
	SS Africa	0.74
Potatoes	All regions	0.81
	Asia	0.82
	Latin America	0.75
	SS Africa	0.74
Rice	All regions	0.79
	Asia	0.87
	Latin America	0.82
	SS Africa	0.54
Sorghum	All regions	0.50
	Asia	0.85
	SS Africa	0.30
Wheat	All regions	0.96
	Asia	1.01
	Latin America	1.06
	MENA	0.83
	SS Africa	0.53

1.2. Evenson and Rosegrant (2003) shocks aggregated to GTAP categories (percentages).

Category	Region	Lower-bound shock	Upper-bound shock
Vegetables and Fruits	Asia	−2.30	−2.99
	Latin America	−2.74	−3.56
	Middle East and North Africa	−0.13	−0.17
	Sub-Saharan Africa	−3.01	−3.91
Coarse Grains	Asia	−43.34	−56.34
	Latin America	−23.44	−30.48
	Middle East and North Africa	−11.41	−14.83
	Sub-Saharan Africa	−9.23	−12.01

These shocks come from adjusting the shocks for cassava, potatoes and lentils from 1.1 by their value shares in the aggregated GTAP category vegetables and fruits (top four rows) and likewise, sorghum, barley and maize are adjusted by their value shares on the GTAP category 'Coarse grains'.

1.3. Decomposition of production changes in yield and area changes – developing and developed countries.

Region	Variable	Wheat	Rice	Coarse Grains	Vegetables and Fruits	Evenson and Rose-grant – All	Oilseeds	Other agric.	All Crops
Developing	Production	−43, −60	−14, −22	−6, −6	−4, −7	−10, −15	−7 −11	−3, −5	−8, −12
	Harvested area	−5, −11	19, 25	15, 25	−11, −15	5, 7	−12, −16	−11, −14	1, 1
	Yield	−38, −49	−33, −48	−21, −31	7, 8	−15, −22	4, 5	8, 10	−9, −13
	Exports	−85, −93	19, 240	−34, −38	−2, −1	−11, −7	3, 6	1, 6	−5, 0
	Imports	111, 191	228, 560	23, 50	6, 11	54, 99	−1, −2	6, 13	30, 56
Developed	Production	46, 76	53, 90	9, 15	1, 1	16, 27	−1, −2	1, 1	12, 20
	Harvested area	21, 30	24, 40	−6, −8	−12, −18	5, 8	−12, −18	−13, −20	1, 2
	Yield	25, 46	29, 50	14, 23	13, 20	11, 19	11, 16	14, 21	11, 19
	Exports	104, 177	297, 572	27, 49	4, 6	38, 65	−4, −7	4, 5	25, 43
	Imports	−2, 0	−6, −3	−1, −1	0, 0	0, 0	0, 0	0, 1	0, 1

Results reported here are percentage changes relative to the baseline year (2004) in production, harvested area, yields, exports and imports. The values are weighted averages using the following as weights: physical output in tonnes for production; hectares for area; and export and import values. For each scenario, lower and upper bounds are separated by a comma.

1.4. Comparison of results obtained by Villoria (2011) and Evenson and Rosegrant (2003).

	Variable	Wheat	Rice	Coarse Grains	Vegetables and Fruits	Evenson and Rose-grant – All	Oilseeds	Other agric.	All Crops
Villoria (2011)	Price	28.9, 59.3	68.3, 135.1	20.2, 41.7	5.7, 9.8	13.4, 26.3	4.9, 8.5	5.22, 9.3	10.0, 19.3
	Production	6, 15	−10.6, −17.3	2.8, 6.6	−3.0, −5	−1.4, −1.1	−4.7, −7.2	−2.5, −3.8	−1.9, −2.3
	Harvested area	9.4, 12.2	20.1, 26.8	8.0, 13.6	−10.6, −15.2	5.7, 8.3	−11.2, −16.1	−10.9, −15.0	1.5, 2.2
Evenson and Rosegrant (2003)	Price	29, 61	80, 124	23, 45		35, 66			
	Production	−9, −14	−11, −14	−9, −12		−8, −12			
	Harvested area	3.5, 5.6	7.5, 9.4	1.1, 1.9		1.8, 4.6			

Results are percentage changes relative to the baseline year (2004) in prices, production and harvested area aggregated using as weights: output values for prices; physical output for production and area. For each scenario, the values for the lower and upper bounds are separated by a comma. The lower part of the table shows some of the results obtained by Evenson and Rosegrant (2003, table 23.3, p. 484). Omitted are changes for other grains, potatoes and root crops. Their results for maize are under the column coarse grains.

1.5. Changes in land cover – developing and developed countries.

Region	Cropland	Forests	Pasture
Developing	0.92, 1.52	−0.53, −0.86	−0.39, −0.66
Developed	0.50, 0.87	−0.29, −0.51	−0.12, 0.36

These are productivity (rental share) weighted changes in land covers. The figures are weighted averages of all regions within developing and developed countries using land rents as weights. For each scenario, values for lower and upper bounds are separated by a comma.

1.6. New area required by crop and region (million hectares) under the counterfactual of no crop germplasm improvement since 1965.

Region	Wheat	Rice	Coarse Grains	Vegetables and Fruits	Evenson and Rose-grant – All	Oilseeds	Other agric.	All crops
Latin America	–4.65, –6.93	1.62, 2.3	4.83, 7.02	–0.52, –0.71	1.29, 1.68	0.48. 0.91	–0.59, –0.79	1.18, 1.8
S.E. Asia	–0.05, –0.06	6.98, 8.51	1.63, 2.52	–2.26, –2.84	6.3, 8.14	–3.32, –4.33	–1.86, –2.41	1.11, 1.4
Rest of Asia	5.26, 5.05	19.02, 24.91	19.64, 33.85	–17.01, –24.84	26.9, 38.96	–12.85, –18.6	–6.33, –8.97	7.72, 11.35
SS Africa	–0.42, –0.44	1.31, 1.77	2.5, 3.09	–0.96, –1.21	2.43, 3.2	–0.67, –0.87	0.08, 0.39	1.85, 2.72
MENA	–4.59, –7.97	0.42, 1.31	2.38, 3.84	1.0, 1.65	–0.79, –1.16	0.55, 0.98	0.35, 0.59	0.11, 0.41
Developed countries	24.91, 36.71	0.96, 1.56	–6.14, –8.1	–4.03, –6.31	15.7, 23.87	–7.56, –11.57	–2.16, –3.22	5.98, 9.07
All Regions	**20.46, 26.36**	**30.31, 40.37**	**24.84, 42.22**	**–23.78, –34.25**	**51.83, 74.69**	**–23.37, –33.53**	**–10.51, –14.41**	**17.95, 26.75**

1.8. Effects of declining productivity in Brazil's soybeans sector on production and exports (all model regions and crops).

Region	Variable	Oilseeds	Wheat	Rice	Coarse Grains	Vegetables and Fruits	Other agric.
Brazil	Production	–67	13	–1	1	1	3
	Exports	–95	18	50	8	10	20
Canada	Production	18	–2	–4	0	–0	–1
	Exports	25	–2	–10	–0	–1	–3
China	Production	8	0	–0	–0	–0	0
	Exports	29	2	1	0	0	–0
EU27	Production	21	0	0	–0	0	–0
	Exports	50	1	1	–0	0	–1
USA	Production	14	–1	–1	–0	–0	–1
	Exports	31	–2	–2	–1	–1	–4
Rest of the world	Production	6	0	–0	–0	0	–0
	Exports	34	–0	–2	–0	0	–1

All figures are percentage changes. Production and export value are weighted averages.

1.9. Changes in land cover in each agro-ecological zone (AEZ) in Brazil (percentages).

AEZ	Cropland	Forests	Pasture
AEZ 1	–0.00	0.00	0.00
AEZ 2	–0.00	0.00	0.00
AEZ 3	–0.00	0.00	0.00
AEZ 4	–0.01	0.01	0.01
AEZ 5	–0.13	0.03	0.09
AEZ 6	–0.05	0.04	0.01
AEZ 10	0.00	0.00	–0.00
AEZ 11	–0.00	0.00	0.00
AEZ 12	–0.05	0.02	0.03

The table shows rental share weighted percentage changes in land cover by AEZ.

1.10. Land rents per cover types and for oilseeds in each AEZ in Brazil (million US$).

AEZ	Oilseeds	Cropland	Forests	Pastures
AEZ 1	0	0	0	0
AEZ 2	3	21	0	6
AEZ 3	4	78	0	21
AEZ 4	71	301	26	62
AEZ 5	489	1484	123	555
AEZ 6	146	1321	272	204
AEZ 10	0	8	0	1
AEZ 11	0	0	0	0
AEZ 12	445	1727	67	153

1.11. New hectares by country after decline in productivity of Brazilian soybeans (million ha).

Region	Oilseeds	All other crops	All crops
Brazil	–3.9	3.6	–0.3
Canada	0.9	–0.5	0.4
China	1.4	–1.3	0.1
EU27	2.0	–1.8	0.1
Indonesia	0.3	–0.3	0.0
USA	2.6	–2.3	0.3
Rest of the world	5.3	–4.7	0.6
All regions	8.6	–7.4	1.2

1.12. Effects of increasing productivity in the oil palm sectors of Indonesia and Malaysia on production, yields and area (percentages).

Region	Variable	Oilseeds	Vegetable oils and fats
Indonesia–Malaysia	Price	–26	–17
	Harvested area	10	
	Production	68	74
	Yield	58	
	Exports	197	98
Rest of the world	Price	–1	–1
	Harvested area	–2	
	Production	–3	–9
	Yield	–1	
	Exports	–3	–18

Values for the rest of the world are weighted using the following as weights: output values for prices, physical output (tons) for area and yields and export values for exports.

1.13. New area required by country after increase in productivity of Indonesia-Malaysia oilseeds (million ha).

Region	Oilseeds	All other crops	All crops
Brazil	–0.7	0.6	–0.1
Canada	–0.2	0.1	–0.1
China	–0.3	0.3	0.0
EU27	–0.2	0.2	–0.0
Indonesia	1.1	–1.0	0.1
USA	–0.6	0.5	–0.1
Rest of the world	–2.9	2.6	–0.4
All regions	–3.8	3.3	–0.5

1.14. Change in land cover in Indonesia-Malaysia and the rest of the world (percentage changes).

Region	Cropland	Forests	Pasture
Indonesia–Malaysia	0.24	–0.24	0.00
Rest of the world	–0.02	0.01	0.01

Productivity (rental share) weighted changes in land covers. The figures for the rest of the world are land rent weighted averages of all regions except Indonesia-Malaysia.

Editing and layout: Green Ink Ltd, UK (www.greenink.co.uk)

Independent Science and
Partnership Council Secretariat
c/o FAO
Viale delle Terme di Caracalla snc
00153 Rome, Italy

www.sciencecouncil.cgiar.org
t 39 06 57056782
f 39 06 57053298
ISPC-Secretariat@fao.org

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Measuring the Environmental Impacts of Agricultural Research: Theory and Applications to CGIAR Research

Independent Science and Partnership Council October 2011