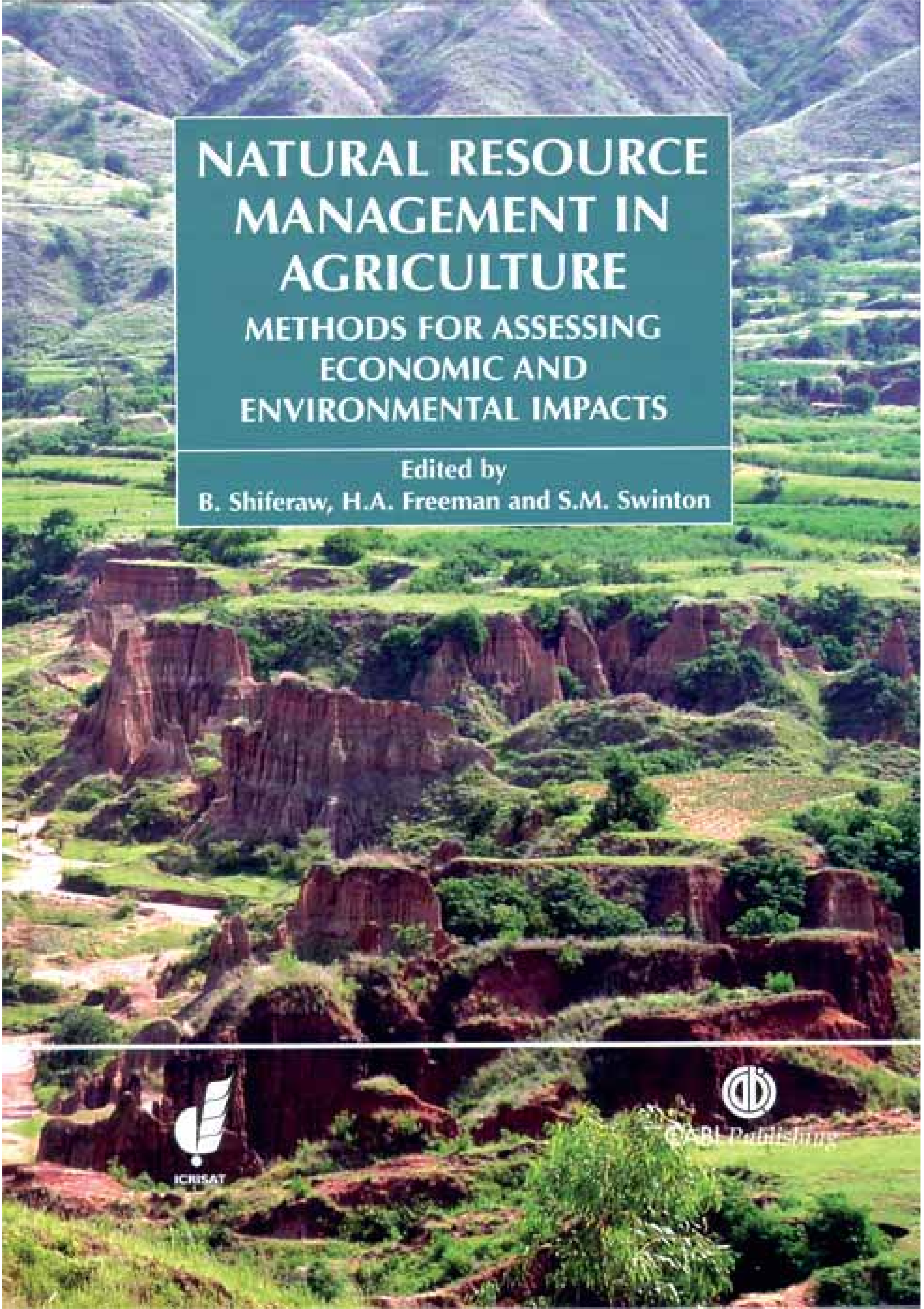

NATURAL RESOURCE MANAGEMENT IN AGRICULTURE

METHODS FOR ASSESSING ECONOMIC AND ENVIRONMENTAL IMPACTS

Edited by
B. Shiferaw, H.A. Freeman and S.M. Swinton

ICRISAT

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Part I.

Introduction

1 Assessing the Impacts of Natural Resource Management Interventions in Agriculture: Concepts, Issues and Challenges

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Introduction

One of the greatest development challenges facing the world in the 21st century is meeting the rising demand for food while maintaining the sustainability of the natural resource base. Increases in per capita income, population growth and urbanisation are expected to double global food demand in the next 40–50 years. The demand for cereals is estimated to increase from 1.9 billion tonnes (t) in 1997 to 2.5 billion t by 2020 and for meat from 209 million t to 327 million t (Rosegrant *et al.*, 2001). These trends in food demand have important implications for natural resources that provide essential support to life and economic processes.

Natural resource management (NRM) aims for the efficient and sustainable utilisation of renewable and non-renewable natural resources. In the context of this book, NRM in agriculture refers to human administration and sustainable utilisation of biophysical resources for the production of food, feed, fibre and fuel. Production in this sense entails direct husbandry, including such activities as aquaculture and planted forests, but does not include hunting, fishing and gathering of uncultivated species. Natural resources of interest include all those affected by the production process (e.g. soil, water, biodiversity, fish and forests). Accordingly, depending on the resource and environmental service flows affected, impact assessment of NRM in agriculture includes the associated changes in the environmental impacts of agricultural production.

Well-managed natural resources generate flows of benefits that provide the basis for maintaining and improving livelihoods, improve the quality

 Natural Resource Management in Agriculture: Methods for Assessing Economic and Environmental Impacts (eds B. Shiferaw, H.A. Freeman and S.M. Swinton)

of life, and contribute to sustainable growth. Agricultural production worldwide mostly depends on soil, providing the most important source of livelihoods for the majority of rural people in the developing world. Water is essential for sustaining human populations and, indeed, all species. It is also a key input in agricultural and industrial production and processing as well as an important sink for discharging waste. Fish are an important biological resource that account for 20% of the animal-derived protein consumption in low-income countries and about 13% in the developed countries (Delgado *et al.,* 2003). With increasing intensification of food production, aquaculture is becoming an important source of income and livelihoods in many parts of the world. Forests and forest resources, including agroforestry and tree crops, provide a source of livelihoods for over 1.6 billion people worldwide. Forests also contain at least 80% of the remaining global biodiversity, they help to protect water resources, and they are a significant carbon sink mitigating climate change (World Bank, 2001). Biodiversity enables animal and crop improvement programmes that maintain and increase productivity. Properly managed natural resources provide an essential foundation for reducing poverty and promoting sustainable growth.

However, the combined effects of population growth, higher levels of economic activity per capita, and mismanagement are putting increasing pressure on the natural resource base. There is abundant evidence of natural resources degradation worldwide. Over the past 45 years an estimated 1.2 billion ha has been degraded as a result of human activity. This affects more than 900 million people in 100 countries. Erosion, salinisation, compaction, and other forms of degradation afflict 30% of the world's irrigated lands, 40% of rainfed agricultural lands, and 70% of rangelands. Every year an additional 12–15 million ha of forests are lost to deforestation. The world is facing a systemic water crisis resulting from the unsustainable use and management of water resources. New threats and challenges to water supplies arise from urbanisation, over-extraction of surface and ground water, pollution, and loss of aquatic biodiversity (World Bank, 2001).

Degradation of natural resources has real economic, social, and human costs with substantial impacts on national economies. It also directly threatens the long-term growth of agricultural productivity, food security, and the quality of life, particularly in developing countries. Investments in agricultural research have resulted in dramatic increases in food production generated from higher-yielding crop varieties with improved resistance to pests and diseases, mostly in areas of high agricultural potential in developing countries. The dramatic increase in production of rice, maize and wheat, referred to as the Green Revolution – was credited with averting widespread per capita food shortages and starvation in the later half of the 20th century, particularly in Asia and Latin America. The short-term crop productivity gains of the Green Revolution are however associated with long-term degradation of soils, water, biodiversity, and marginal lands. Pingali and Rosegrant (1998) provided empirical evidence linking the intensification of rice–wheat systems in the Indo-Gangetic plains of South Asia to the build up of salinity and waterlogging, depletion of groundwater resources, formation of hard pans, soil nutrient deficiencies, and increased incidence of soil toxicity. Thus, while improving agricultural productivity is an essential component in many poverty-reduction and growth strategies, degradation of natural resources can threaten the achievement of this objective.

Natural resource degradation is particularly costly for the poor. Poor people often depend directly on natural resources for their livelihoods, making them especially vulnerable when natural resources lose their productive potential. There is growing awareness that sustainable use of natural resources can contribute to poverty alleviation and improvements in human welfare. Project, programme, or policy interventions that improve the management of natural resources can lead to significant economic gains that directly benefit poor people, resulting in substantial improvements in their welfare.

The linkages between sustainable management of natural resources and improvements in the well being of the poor have contributed to a resurgence in development lending and research investments on environment and NRM over the past two decades. The World Bank, for example, is increasing lending for environment and NRM issues after a period of decline over the last few years. In 2003 US$1.1 billion was allocated for environmental and NRM issues, representing 6% of overall lending, an increase from 4.7% in 2002 (World Bank, 2003). Similarly, international organisations focusing on sustainable increase in agricultural productivity and improvement in rural livelihoods such as the Consultative Group on International Agricultural Research (CGIAR), have increased the share of NRM research in their overall research portfolio (Kelley and Gregersen, Chapter 15, this volume). Between 1994 and 2001, CGIAR research investments in protecting the environment rose from 15 to 19% of total resource allocation, while investment on biodiversity almost doubled from 6 to 11% (Barrett, 2003). These trends in resource allocation generally reflect the growing consensus that the objectives of poverty alleviation, food security, and sustainable management of natural resources are highly interdependent.

This chapter identifies key issues involved in assessing the impacts of NRM interventions. Such interventions include adoption of changed NRM practices arising from investments in research and outreach that are implemented through NRM projects, programmes, and policies. The focus is on impact analysis of NRM interventions, not on conducting NRM projects *per se*. The next sections discuss the purposes of impact assessment, followed by the underlying concepts and techniques for conducting impact assessment. This is followed by a discussion of the special challenges that complicate impact assessment of NRM interventions. The chapter ends by providing an overview of the conceptual and empirical approaches for NRM impact assessment.

Why Assess NRM Impacts?

Impact assessment should enhance the understanding of the extent to which project, programme, and policy interventions affect the target population and the magnitude of these intervention effects on the welfare of the intended beneficiaries. Resources are limited and managers in research and development institutions are under pressure to allocate available resources efficiently and effectively.

Impact assessment, whether it is backward-looking, evaluating the impact of past research and development (R&D) investments (*ex post* impact assessment) or forward-looking, evaluating the impact of current and future R&D investments (*ex ante* impact assessment) should help in setting priorities over competing interventions and inform policy decisions on efficient allocation of scarce resources.

Impact assessment can be used to measure the outcomes and impact of development interventions, aiming to discern intervention effects from the influence of other external factors. As noted above, this is particularly challenging with NRM interventions.

Donors, policy makers, and development managers need information to monitor progress in achieving outputs and outcomes, providing a basis to demonstrate results, and strengthening accountability for results that may justify continued funding. Often, broad indicators of impact such as aggregate rates of returns to investments and benefit–cost ratios are used as indicators to provide evidence of the effectiveness of past and future interventions. These indicators are used to make decisions on whether to expand, adjust, or drop project, programme, or policy interventions. *Ex post* evaluation also provides lessons that could be used to improve the design and management of service delivery and other future interventions. Comprehensive impact assessment that includes both productivity and environmental and sustainability impacts provides an objective basis for comparing the effectiveness of alternative interventions in achieving the stated welfare and sustainability objectives. Such information is useful for planning, setting priorities, and allocating resources to alternative interventions. However, evaluating the actual livelihood and poverty impacts of agricultural and NRM interventions would require analysis of distributional and equity impacts in addition to computation of such simple efficiency indicators as net present values, benefit–cost ratios, and internal rates of return. New methods and approaches are needed to extend traditional impact assessments to address such policy-relevant concerns.

R&D organisations are increasingly interested in assessing a broad range of impacts from NRM interventions. This, however, requires examining a range of multi-dimensional impacts that may include impacts on the quality of the resource base as well as the flow of ecosystem services that provide basic life support functions in agro-ecosystems. These non-market benefit objectives imply that conventional economic impact analyses are fundamentally incompatible with measuring the benefits that NRM projects seek to obtain. Methodological development in the approaches and techniques for valuation of ecosystem and environmental goods and services is enabling assessment of environmental impacts associated with NRM interventions that have been largely neglected in past impact assessment studies.

Nevertheless, methods for assessing the multi-faceted impacts from NRM interventions are far less developed than methods for assessing impact for crop improvement research (Izac, 1998; Shiferaw and Freeman, 2003). This explains, in part, the dearth of credible quantitative evidence, *ex ante* or *ex post*, that assesses the impact of NRM research compared to the evidence on the effects of crop improvement research. For example, of the 1886 rates of return on research investment reviewed by Alston *et al.* (2000) over 50% were for crops research, while NRM research accounted for less than 5%. The limited number of studies on NRM impact assessment, despite the increased interest on sustainability issues, suggests that tracing the practical linkages between NRM interventions with changes in the resource base, the environment, and human welfare is fraught with complexities (Nelson and Maredia, 1999). The specific challenges and empirical difficulties that impact evaluators face in undertaking valid and plausible assessment of NRM impacts are discussed below.

Impact Assessment: Concepts and Processes

In the literature, the term 'impact assessment' is used interchangeably with 'impact evaluation'. Impact assessment determines the welfare changes from a given intervention on individuals, households and institutions and whether those changes are attributable to the project, programme, or policy intervention (Baker, 2000; World Bank, 2002).

Impact assessments are often undertaken *ex ante*, evaluating the impact of current and future interventions, or *ex post*, evaluating the impact of past intervention. Impact assessment can also be made concurrently within the project cycle. *Ex ante* assessment intends to inform policy decisions as to whether a proposed project or programme intervention should be carried out at all. Such evaluations gather information on the likely economic and environmental impacts and how the flow of costs and benefits is distributed across the affected populations. The distributional impacts and identification of winners and losers are critical elements in evaluating the social impacts of proposed interventions. The *ex ante* assessment compares the expected benefits and costs over time along with the anticipated social impacts. Such information is often used to prioritise interventions and inform policy choice as to whether the expected social benefits would outweigh the costs – to justify implementation of proposed interventions. *Ex post* impact assessments gener-ally intend to measure realised benefits and costs of programme interventions to see whether stated objectives have been met and whether the realised benefits indeed outweigh the direct and indirect costs incurred. *Ex post* assessment also attempts to understand the pathway through which observed impacts have occurred and why interventions fail or succeed in attaining stated objectives. Hence, *ex post* assessments can inform policy choices as to whether related planned programme interventions should be discontinued, modified, improved or sustained in the future.

An important aspect of impact assessment is to understand how interventions affect the beneficiaries or affected populations and whether any outcomes and improvements are a direct result of the intervention. An intervention will not enhance economic efficiency unless the realised or anticipated benefits exceed the overall costs. In cases where the desired impact is not being achieved, the evaluation can also provide useful information on how the programme design could be improved.

Measuring project outcomes alone is not sufficient to assess impacts. In many cases, there may be other factors or events that affect outcomes other than the project itself. For example, if an agroforestry outreach project is initiated and shortly thereafter the national government ceases to subsidise imported fertiliser, farmers may begin to rely upon agroforestry methods to meet crop nutritional needs. In order to measure the real impact of the agroforestry outreach intervention, it is important to control for other confounding factors such as the subsidy termination, and to net out those outcomes that can be attributed only to the intervention itself. This means that impact assessment must estimate the counterfactual, i.e. what would have happened had the intervention never taken place.

Determining the counterfactual is at the core of evaluation design (Baker, 2000). Three broad quantitative methods can be used to identify an appropriate counterfactual (Heckman and Robb, 1985; Heckman and Smith, 1995), including estimation methods used with randomised experimental design, non-randomised quasi-experimental methods, and non-experimental designs.

In the experimental design approach, groups are selected randomly from the same population as the programme participants, while the control group is randomly assigned among those who do not receive the programme. The control group should resemble the treatment group in every sense, with the only difference between the two being the presence of the programme intervention in the treatment group. The main benefit of this technique is the simplicity in interpreting the results – intervention impact can be estimated by the mean difference between the treatment and control groups. While the experimental design is considered the ideal and most robust approach to estimating intervention impacts, it has several disadvantages. Firstly, randomisation, which involves denial of benefits for a certain group of people, may not be ethically acceptable for many interventions. Secondly, randomisation may not be politically acceptable. Thirdly, the proposed project, programme or policy may have economy-wide effects that make randomisation unfeasible. Fourthly, experimental designs may be technically impossible (e.g. due to mobile populations) or expensive and tedious to implement.[1] These difficulties often limit the practical usefulness of the experimental design approach for establishing a valid counterfactual.

Quasi-experimental designs such as matching, reflexive comparison, and double difference methods, and non-experimental designs, such as instrumental variables methods, can be used when it is not possible to construct treatment and comparison groups through experimental design. Matching involves identifying non-programme participants comparable in essential characteristics to programme participants to be matched on the basis of common characteristics that are believed to influence programme outcomes. The propensity score matching approach that is based on the predicted probability of participation given observed characteristics is the most commonly used approach for matching. The reflexive comparison method compares programme participants before and after the programme. The double difference method compares both programme participants and non-participants before and after the programme. Instrumental variables consist of using 'instruments' that matter to participation but not to outcomes given participation, allowing identification of exogenous variation in outcomes attributable to the programme, while recognising that its placement may not be random but purposive. Instrumental variables are first used to predict programme participation; then the programme impact is estimated using predicted values from the first equation (Baker, 2000).

Selection bias is a major challenge to measuring programme impacts in non-experimental settings. Selection bias occurs when pre-existing conditions skew outcomes in a way that is not truly attributable to the programme intervention. For example, if farmers with the best land adopt a practice of soil conservation faster than farmers with poor land, the yield gain they achieve may exceed what other farmers could expect, due to their higher land quality. When bias exists, the assessment may provide inaccurate results that could lead to erroneous inferences and conclusion about the impacts of the intervention (Friedlander and Robins, 1995). Randomised experiments avoid selection bias through random selection. The quasi-experimental and non-experimental designs must rely upon statistical methods to minimise bias due to non-random data. Certain statistical methods allow comparison of programme participants and non-participants while controlling for the process of selection (Pender, Chapter 6, this volume; Greene, 1997; Baker, 2000). However, these methods tend to be less robust statistically than ones that use experimental data. Moreover, the statistical methods for correcting selection bias can be quite complex (e.g. Kerr, 2001), and it is often difficult to fully correct for it in practice (Baker, 2000).

Qualitative methods are also used for impact assessment. Such methods seek to determine impacts by relying on methods other than the counterfactual (Mohr, 1995). Qualitative approaches involve understanding the processes, behaviours and conditions surrounding NRM interventions. Often qualitative methods are participatory, relying upon the perceptions of the individuals or groups being studied (Valadez and Bamberger, 1994). Qualitative approaches tend to use open-ended designs for data collection, including focus group discussions, key informant surveys, and participatory appraisals. Examples can be found in Chapters 11 (Bantilan *et al.*) and 14 (Douthwaite *et al.*) in this volume. Commonly used analytical tools include stakeholder analysis and beneficiary assessment. Qualitative approaches provide insights into the way in which households and communities perceive a project and how they feel affected by it. Qualitative methods can be simple, quick, flexible, and tailored to specific socio-economic conditions. However the subjectivity involved in data collection, the lack of a counterfactual and limited statistical rigour make the results less conclusive and more difficult to generalise than quantitative assessments.

Qualitative approaches are increasingly used in conjunction with quantitative approaches (Baker, 2000), and such combinations can enhance the validity and reliability of impact evaluations (Bamberger, 2000). While quantitative approaches allow statistical tests for causality and isolation of programme effects from other confounding influences, qualitative methods allow in-depth study of selected issues and help the evaluator find explanations for the results obtained in the quantitative analysis. In short, quantitative methods excel at answering impact assessment questions about 'what' and 'how much', whereas qualitative methods are preferred for exploring questions of 'how' and 'why'. A mix of quantitative and qualitative approaches is ideal because it provides the quantifiable impacts of the intervention as well as an explanation of the processes and relationships that yielded such outcomes.

The evaluation design chosen for NRM impact assessment needs to capture the special features, complexities and multiple outcomes associated with such interventions. For example, assessing the impacts of NRM technology and policy interventions requires accounting for both the tangible and the less-tangible and diffuse productivity and environmental impacts. The process of tracking these relationships and impact pathways may involve several steps. Nelson and Maredia (1999) discussed five steps in assessing environmental costs and benefits in NRM projects. These steps involve:

- Understanding the causes and impact of changes in the use of natural resources such as declining soil fertility, land degradation, water pollution, deforestation, loss of biodiversity, etc.
- Identifying the main types of economic costs and benefits. Economic costs could include depletion of the stock of natural resources and species losses. An important consideration is to identify the distribution of the burden of these costs over time and space and across affected communities
- Determining whether or not there is a means to measure costs and benefits in monetary terms
- Assessing the extent of changes in the use of natural resources and the environmental consequences resulting from these changes. This includes collecting data to estimate the impact of environmental effects on such indicators as productivity, income, and human health
- Using economic techniques to place values on environmental changes.

Key biophysical processes and related indicators of NRM status are explored in this volume with foci on the soil (Pathak *et al.*, Chapter 3), water resources (Sahrawat *et al.*, Chapter 4), and ecosystem services (Wani *et al.*, Chapter 5). Shiferaw *et al.* (Chapter 2), discuss several methods for placing economic values on non-market ecosystems services, while Drechsel *et al.* (Chapter 9) provide examples of applying some of the commonly used valuation methods to valuing changes in soil fertility.

Challenges in NRM Impact Assessment

Apart from the general challenges of attribution and selection bias in impact analysis, there are special conceptual and methodological challenges that arise from several unique features of natural resource management. NRM impact assessment needs to address important challenges of attribution, measurement, spatial and temporal scales, multidimensional outcomes, and valuation. The cross-commodity and integrated nature of NRM interventions makes it very challenging to attribute impact to any particular one among them. In crop genetic improvement where the research outputs are embodied in an improved seed, it is less difficult to attribute yield improvements to the investment in research. Changes in NRM frequently involve observable research products adopted by farmers as well as qualitative information about recommended management practices. Knowledge about such improved management practices may be transmitted through formal and informal outreach activities and by the self-experimentation and indigenous knowledge of the farmers themselves. In many cases, for such knowledge and information-based changes in NRM practices, it is difficult to identify the impacts attributable to the intervention. Also, it is not uncommon for different agencies to be involved in the development and promotion of new NRM technologies, making it hard to separate the impacts attributable to specific programmes. For example, in the evaluation of watershed programmes in India, it was difficult to attribute improvements in resource conditions and farm incomes to specific interventions, since increased participation and collaboration among a range of R&D partners was identified as a significant determinant of success (Kerr, 2001). The fact that most agricultural NRM interventions are information-based but not embodied in an easily measured package vastly complicates the attribution of observed impacts.

Identifying an appropriate counterfactual in NRM interventions is particularly challenging because quantifying the biophysical impacts of interventions on natural resources can be costly, imprecise, and slow. For NRM interventions that aim to halt resource degradation, the counterfactual may be a significant productivity decline. Hence, a properly measured counterfactual may reveal that achieving non-declining productivity represents a major gain over what would otherwise have occurred.

Identifying appropriate spatial boundaries for assessing NRM impact is often fraught with difficulty (Campbell *et al.*, 2001; Sayer and Campbell, 2001). Agricultural NRM typically involves different spatial scales, from farmers' fields to entire watershed catchments, implying that many levels of interaction may need to be considered in assessing the impacts of research interventions. Multiple scales of interaction create upstream and downstream effects that complicate impact assessment. For example, assessing the impact of land use interventions in a watershed may need to take into account multiple interactions on different scales because erosion and runoffs in the upper watershed may not have the same impact on water quality downstream. It is also likely that interventions could have different effects, which in some cases can generate opposite impacts on different spatial scales. For example, soil and water conservation interventions can have a positive impact on crop yields upstream but negative impacts by reducing water availability downstream when water is a limiting factor for production, or positive impacts by reducing sedimentation, runoff and flooding when water is not a limiting factor.

In the temporal dimension, methodological challenges for NRM impact assessment arise from slow-changing variables and substantial lags in the distribution of costs and the benefits. For example, soil loss, exhaustion of soil fertility, and depletion of groundwater resources take place gradually and over a long period of time. In some cases it may be difficult to perceive the costs or the benefits of interventions to reverse these problems. In other cases, assessing the full range of the impacts of investments related to these slow-changing variables in a holistic manner may involve intensive monitoring of multiple biophysical indicators on different spatial scales over long periods of time. These factors make impact monitoring and assessment of NRM interventions a relatively slow and expensive process. Differences in time scale for the flow of costs and benefits are translated into lags in the distribution of costs and benefits that complicate impact assessment. Typically, costs are incurred up-front while delayed benefits accrue in incremental quantities over a long period of time (Pagiola, 1996; Shiferaw and Holden, 2001). For example, the benefits from the biodiversity that is used in genetic improvement of crop and animal varieties accrue in the long term but costs of *in situ* and *ex situ* conservation are incurred in the short term. The timing of an intervention can also affect its impact. This is, for example, the case for improved crop management practices that require optimising sowing date, fertiliser application, weeding and harvesting.

When outcomes are delayed and tend to vary according to local biophysical conditions, simulation models can facilitate the *ex ante* evaluation of NRM technology options that fit micro-climatic and agro-ecological niches. Biophysical process models are mainly used to explore the biophysical and productivity impacts of changes in agricultural and NRM practices (Wani *et al.*, Chapter 5, this volume). Bioeconomic models, on the other hand, interlink economic and biophysical information to simulate optimal resource use and investment behaviour (Holden, Chapter 8, this volume; Shiferaw and Holden, Chapter 12, this volume). Both kinds of models require biophysical and experimental agronomic data to calibrate and validate them to local conditions.

NRM interventions may generate multidimensional biophysical outcomes across resource, environmental and ecosystem services. These might include changes in the quality and movement of soil, quantity and quality of water, sustainability of natural resources, and conservation of biodiversity. Appropriate indicators are needed to monitor the impacts of NRM interventions on the biophysical conditions of the soil (Pathak *et al.*, Chapter 3, this volume), water resources (Sahrawat *et al.*, Chapter 4, this volume), and the flow of ecosystem services that support agro-ecosystems (Wani *et al.*, Chapter 5, this volume). The multidimensionality of outcomes from NRM interventions means that impact assessment often faces difficult measurement challenges, including very different measurement units and potentially the integration of very different natural resource outputs into some kind of uniform aggregate yardstick (Byerlee and Murgai, 2001).

The multidimensionality of NRM outcomes extends to those directly or indirectly affecting human beings. NRM interventions can generate environmental and health benefits whose values might not be reflected in current markets, but on which society places a value for multiple reasons. For example, water and water-based ecosystems provide not only direct values in consumptive uses (e.g. fishing, irrigation) and non-consumptive uses (e.g. aesthetic value), but also indirect use values such as ecosystem functions and services, option values for possible future uses and applications and non-use values for intrinsic significance (existence and heritage value). Empirical valuation of non-market benefits is explored by Shiferaw *et al.* (Chapter 2, this volume). But depending on how NRM ideas are conveyed, the human outcomes may extend even further. Integrated NRM projects engage in participatory activities that may empower individuals and communities in ways that extend far beyond the realm of agricultural NRM, as discussed by Douthwaite *et al.* (Chapter 14, this volume).

Approaches for Assessing NRM Impacts

Impact assessment for NRM interventions ultimately needs to show the social costs and benefits associated with the research, promotion, and adaptation of these interventions. Given the complexities and challenges associated with measuring, monitoring and valuing such changes, more innovative assessment methods are required. An important factor that needs to be considered in the selection of appropriate methods is the capacity to account for non-monetary impacts that NRM interventions generate in terms of changes in the flow of resource and environmental services that affect sustainability and ecosystem health. As discussed earlier, a mix of quantitative and qualitative methods may be the optimal approach for capturing on-site and off-site monetary and non-monetary impacts. The economic surplus approach is the commonly used method for evaluating the impacts of agricultural research investments, particularly for crop improvement technologies. This approach estimates benefits as changes in `economic surplus' (the aggregate value that consumers are willing to pay above and beyond what it costs producers to supply the good or service in question). The cumulative benefits are then compared to cumulative R&D costs over time. Specifics and the challenges of incorporating non-marketed on-site effects and off-site externalities are discussed by Swinton (Chapter 7, this volume), with Bantilan *et al.* (Chapter 11, this volume) providing an empirical application.

Promising analytical methods that can be used to quantify economic changes due to NRM interventions include econometrics (Alston *et al.*, 1995) and bioeconomic optimisation modelling. For example, econometric methods can be used in empirically estimating the demand for marketed or certain non-marketed goods and services, providing elasticities for calculations of economic surplus. Econometric methods can also be used to link a time-series of measures of output, costs and profits directly to past R&D investments (Alston *et al.*, 1995). Likewise, they can be used to establish statistical relationships between changes in NRM practices and measured performance indicators, such as land productivity, total factor productivity, production costs, net farm income, or income volatility. Pender (Chapter 6, this volume) discusses the conceptual and empirical issues while Kerr and Chung (Chapter 10, this volume) provide an empirical application of this method.

Bioeconomic modelling nests essential biophysical processes within economic behavioural models. Their constrained optimisation perspective allows evaluating how technological and/or policy changes would affect economic welfare, sustainability, and environmental conditions over time. The integrated framework captures biophysical process evolution along with rational human management responses. Holden offers a conceptual treatment of bioeconomic modelling (Chapter 8, this volume), while Shiferaw and Holden provide an empirical application for a farm household (Chapter 12, this volume) and Holden and Lofgren demonstrate the use of an economy-wide computable general equilibrium model for evaluating NRM technology and policy impacts (Chapter 13, this volume).

As a response to the complexities that impact assessment practitioners face in evaluating the multi-faceted impacts of NRM, there is an increasing interest in developing more holistic and 'softer' assessment methods. Integrated natural resource management (INRM) calls for participatory NRM interventions at multiple scales with frequent adaptive feedback and multiple stakeholders (who often hold contrasting objectives) (Campbell *et al.*, 2001; Sayer and Campbell, 2001). Douthwaite *et al.* (Chapter 14, this volume) explore the conceptual underpinning of the INRM framework and its implications for evaluating NRM impacts.

Organisation of the Book

The chapters in this book address the conceptual framework, methodological challenges and selected empirical experiences of NRM impact assessment. In so doing, they explore many of the complexities identified in this introductory overview. The book's 16 chapters are organised into five parts. Following this initial part that introduces the challenges and approaches to NRM impact assessment, Part II includes four chapters that deal with the valuation of ecosystem services and the measurement of biophysical indicators of NRM impacts. Part III introduces advances in methods used to evaluate the economic and environmental impacts of NRM technology and policy interventions. Part IV deals with NRM impact assessment in practice. Five case studies illustrate the methodological advances discussed in Part III. The final part of the book (Part V) highlights some of the existing controversies and outlines best practices, research issues, and recommendations for NRM impact assessment into the future.

Endnote

[1] One way to enhance ethical and political acceptability of randomisation is to phase the intervention such that some groups gain access to programme benefits at a later stage. In this way the random selection determines when a given group gains access to the benefit, not if they receive it.

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Part II.

Valuation of Ecosystem Services and Biophysical Indicators of NRM Impacts

2 Valuation Methods and Approaches for Assessing Natural Resource Management Impacts

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Introduction

Impact assessment in natural resource management (NRM) is nascent but developing rapidly in response to felt needs. Determining how to value changes in NRM is a major challenge. The value of some NRM investments can be estimated via the value of increases in yield of marketable products or the value of savings in production costs, but many NRM investments generate benefits that are hard to measure because they are not bought and sold in markets. Environmental and resource service flows that offer indirect use and non-use benefits to society certainly have value, but their measurement is a challenge. This chapter focuses on valuation methods and associated issues for measuring the social benefits that result from NRM investments.

Several thorny issues are associated with valuation of the productivity and environmental impacts of NRM investments. These include incomplete understanding of ecosystem functions and difficulties in predicting the effect of interventions on major ecosystem functions and services; lack of measurable performance indicators when effects are relatively well understood; and problems in relating changes in the flow of ecosystem services to human welfare. The non-tangibility of the benefits, time lags, and spatial (scale) effects further complicate the measurement of social, economic, and environmental impacts from NRM interventions. Farmer investments in NRM often provide non-excludable and non-consumptive public goods to the local community and beyond. For example, vegetative barriers and trees planted on the upper reaches of a watershed by a private land-user provide watershed protection, biodiversity conservation and carbon sequestration services to the local community, some of the benefits of which may even extend beyond the micro-watershed to regional and global levels. Although precise estimation of the full economic value of such investments is costly and difficult, the application of appropriate valuation methods may provide useful estimates for the direct and indirect, marketed and non-marketed ecosystem services generated by NRM investments.

Despite the extensive work on environmental valuation and benefit–cost analysis, there is a dearth of literature on methods for valuation of ecosystem services from NRM technology adoption and a serious lack of empirical examples in the developing countries that estimate the social impacts of NRM research and development efforts. This chapter provides an overview of the valuation methods and methodological approaches used to evaluate the economic and environmental impacts of NRM interventions. How NRM investments affect the flow of ecosystem services, the issues involved in translating changes in service flows to welfare gains, and some promising approaches for valuation of welfare changes are discussed. The suitability of the methods described depends on such specific circumstances as: resource types and interventions, anticipated economic and environmental effects, and interaction of biophysical changes with socio-economic conditions. The second part of the chapter summarises the multiple ecosystem services associated with NRM. In the third part, the core issues involved in the valuation of ecosystem services are discussed. This is followed by a presentation of the theoretical foundations and overview of valuation methods along with some examples on applications in the area of natural resources. The next part summarises how economic and environmental impacts can be integrated to provide an assessment of the social net benefits from NRM interventions, and the conclusion highlights the major issues and most promising valuation methods.

Agro-ecosystem Services and Functions

Agro-ecosystems are communities of plants and animals interacting with their physical and chemical environments that have been modified by people to produce food, fibre, fuel and other products for human consumption and processing (Altieri, 2002). Watersheds and agro-ecosystems offer a number of ecosystem services of value to society (for simplicity ecosystem goods and services are referred to as ecosystem services). Ecosystem services consist of flows of materials, energy, and information from the natural capital of ecosystems that provide direct and indirect human welfare benefits. In many cases, such services are public goods that cannot be privatised at low cost (high costs of exclusion) and whose consumption by one consumer does not reduce the amount available for others (non-rival). Hence, self-interested private individuals may lack the economic incentive to provide such services in socially optimal quantities.

Ecosystem services that embody public goods include: biodiversity conservation, flood and erosion control, carbon sequestration, nutrient recycling, and water retention and storage (Bingham *et al.*, 1995). In other cases, the ecosystem service may involve high costs to privatise and may also be congestible (consumption by one reduces consumption by others). Costs of exclusion are often high for ecosystem services that arise from such common property resources as groundwater, community pastures and woodlots.

Technological interventions for NRM may have diverse effects on an ecosystem at various levels. The first step towards evaluating the economic and environmental impacts of NRM interventions requires an understanding of how such investments change ecosystem functions. Each ecosystem function can be conceived as a subset of complex ecological processes that provide specific goods and services that directly or indirectly satisfy human needs. The diverse ecosystem services generated through ecosystem functions therefore provide various economic, environmental, and socio-cultural benefits and values to people. De Groot *et al.* (2002) developed a typology for the classification of ecosystem functions and services. Although their general typology is meant for natural ecosystems, it can be adapted for use in agro-ecosystems to understand the likely impacts of NRM interventions. Changes in the scale and intensity of managing natural resources in agriculture will change the flow of agro-ecosystem services, which will in turn influence the quantity and/or quality of goods and services produced. Depending on the type of NRM technology used, the typology developed by de Groot *et al.* (2002) suggests that valuable ecosystem services may be generated through any of the following ecosystem functions:

- Production
- Regulatory services
- Habitat
- Socio-cultural (information) services.

Table 2.1 summarises the major ecosystem functions and services together with selected indicators of change due to NRM interventions in the context of agro-ecosystems.

Production of food and raw materials is a major ecological function of agro-ecosystems that includes food, feed, fuel, raw materials and medicines. This function is transmitted through the conversion of solar energy into edible plants by autotrophs for human and animal consumption. Farm animals convert fodder and herbaceous material into economic goods and services for use by humans. Natural resource investments may also influence the ability of the agro-ecosystems to produce products for ornamental and medicinal use, and the conservation of biological diversity. As shown later in this chapter, when data are available, simulation models and statistical methods can be used to establish the relationships between NRM investments and changes in the flow of goods and services (see also Chapter 5, this volume). These effects are typically realised on-site and create economic incentives for resource users to adopt new technologies. When the productivity effects are limited, farmers' direct economic benefits and the incentives for adoption and adaptation of NRM technologies will be low.

The regulation function relates to the role of agro-ecosystems in the maintenance of essential ecological processes and life-support systems. Such ecosystem services may be transmitted through changes in land cover that

Table 2.1. Ecosystem functions and potential indicators of change in agro-ecosystem services associated with natural resource management (NRM) interventions.

Ecosystem services	Ecosystem functions (processes and components)	Indicators for changes in agro-ecosystem services
A. *Production functions* – Provision of natural resources as factor inputs in production activities		
Food	Conversion of solar energy into edible plants and animals for humans	Changes in land productivity (crop and livestock)
Raw materials	Conversion of solar energy into biomass for feed, construction and other uses	Changes in fodder, fuelwood, timber, etc., output
Genetic resources	Conservation of genetic materials	Changes in agro-biodiversity
Medicinal resources	Bio-chemical substances, medicinal uses	Changes in availability of medicinal plants or changes in use benefits from medicinal plants
Ornamental resources	Ornamental use	Changes in economic benefits from ornamental plants and animals
B. *Regulation functions* – Maintenance of essential ecological processes and life support systems		
Climate regulation	Influence of land cover and carbon sequestration on climate	Changes in land cover and carbon sequestration
Water regulation	Role of land cover in regulating runoff and river discharge	Changes in runoff and sediment loss
Water supply	Filtering, retention and storage of fresh water	Changes in water availability and quality
Soil retention	Role of vegetation root matrix and soil biota in soil retention	Changes in rates of soil erosion and sediment loss
Soil formation	Weathering of rock, accumulation of organic matter	Changes in soil depth
Nutrient regulation	Role of biota in storage and recycling of nutrients	Changes in nutrient balances, soil fertility and organic matter
Pollination	Role of biota in movement of floral gametes	Changes in pollinating insects
C. *Habitat functions* – Providing habitat for wild plant and animal species		
Refugium function	Suitable living space for certain desirable species	Changes in the stock of wildlife, soil flora and fauna
Nursery function	Suitable reproduction habitat for certain desirable species	Changes in rates of reproduction
D. *Sociocultural functions* – Providing opportunities for cognitive development		
Aesthetic information	Attractive landscape features	Changes in landscape and scenery
Recreation	Variety in landscapes with (potential) recreational uses	Changes in recreational benefits (agro-ecotourism, outdoor sports, etc.)

Table 2.1 Continued.

Ecosystem services	Ecosystem functions (processes and components)	Indicators for changes in agro-ecosystem services
Cultural and artistic	Features with cultural and artistic value	Changes in cultural and artistic use (e.g. motivation for books, films, advertising, etc.)
Spiritual and historic	Agro-ecosystem types with spiritual and historic value	Changes in use for religious and historical use (e.g. heritage, spiritual symbol)
Science and education	Agro-ecosystem types with scientific and educational value	Recognition for scientific or educational purposes

Source: Updated based on Costanza *et al.* (1998) and de Groot *et al.* (2002)

influence and regulate: climate change (e.g. through carbon sequestration), water flows (runoff and river discharges), and protect soils from erosion, water supply through filtering, retention and storage of fresh water (e.g. wetlands, check dams, etc.); soil formation through decomposition of organic matter and weathering of rocks; nutrient regulation through storage and recycling of nutrients; biological control of pests; pollination through the role of fauna in the movement of floral gametes. A number of useful ecological (biophysical) indicators can be developed to monitor the NRM technology impacts on these kinds of ecosystem services (Chapters 3–5, this volume).

The habitat function indicates the useful services provided by agro-ecosystems in the provision of habitat (suitable living space) and nursery (reproductive space) services for uncultivated and cultivated plant and animal species. People derive non-material well-being from the flow of these services. It is difficult to develop simple indicators to monitor NRM impacts on these ecosystem services. The number of species in a given habitat (species richness) and the species diversity can be measured using different biological indices (Chapter 5, this volume).

Natural resource investments also provide such socio-cultural services as aesthetic information (e.g. attractive landscape), recreational services (e.g. ecotourism), and scientific and spiritual services. These are mainly public goods that provide useful services to society or the community as a whole.

When markets exist, changes in some of these agro-ecosystem services resulting from NRM investments can be quantified and valued in monetary terms. For public goods (e.g. changes in biodiversity, water and air quality) markets are either missing or often imperfect. The quantification of benefits and valuation therefore presents special difficulties. Before valuation methods are considered, the major issues and challenges surrounding valuation of ecosystem services are briefly described.

Issues in Valuation of Agro-ecosystem Services

There are two fundamental steps in the valuation of impacts from NRM investments: firstly, understanding and predicting the changes in the flow of ecosystem services attributable to the technological or policy intervention, and secondly, devising acceptable methods for valuing these changes. The first helps identify and quantify *what is to be valued* while the second one helps develop *suitable methods* for valuing the changes. In this section the issues involved in uncovering what is to be valued and how it is to be valued are discussed. As described above, ecosystems are very dynamic and complex, and human knowledge about them is very incomplete. This limits the ability to understand and quantify the changes in the ecosystems service flows associated with human interventions. The effects of NRM interventions can be physical, chemical or biological, and may take different forms over temporal and spatial scales. However, understanding and predicting the impacts of interventions on ecosystem functions is the prerequisite to economic valuation. Good valuation depends on sound agroecological information on the effects of policy and management interventions. Functional inter-linkages and feedback effects make it difficult to determine the causal relationship between human interventions and changes in ecosystem functions and processes (Bingham *et al.,* 1995). Any sensible effort to assess the impacts of NRM interventions requires a reasonable understanding of how and to what extent the different ecosystem service flows will change as a result of human interventions. This implies an interdisciplinary effort involving agroecologists, agronomists, biophysical scientists and economists. Bingham *et al.* (1995) argue that if there is no agreement on the effect of changes on the flow of ecosystem services, there can be no agreement on valuation of the impacts.

If changes can be predicted or quantified reasonably, the next question will be – which of these changes can be valued in monetary terms? The choice of which changes to value is an important challenge for the economist. Before values can be placed on the impacts, it is necessary to know what is to be quantified and how it can be measured. Indicators of changes in the service flows (immediate impacts) are critical for valuation. Indicators can be developed through experimentation and appropriate monitoring of changes over a sufficient period of time, or through the application of exploratory and predictive simulation models. The latter approach is most useful when changes are slow to evolve or when complexity of anticipated interactions makes actual experimentation very difficult. NRM combines both features and involves multiple interventions that make it problematic to isolate partial effects. In the absence of good counterfactuals, experimental data might not provide useful insights about the anticipated impacts. Oriade and Dillon (1997) provide a good review of applied simulation models used in agricultural systems.

There are various efforts to develop measurable indicators for changes in the flow of agro-ecosystem services (Dumanski and Pieri, 2000; Arshad and Martin, 2002). The next three chapters in this volume provide a detailed account of the measurable indicators for soil, water, and other agro-ecosystem services. The challenge is to develop indicators that could be easily monitored on-farm as part of the project cycle. Good indicators are those that capture major elements in a complex interactive system while simultaneously showing how the value obtained relates to some ideal or desired level. Smyth and Dumanski (1993) reported the use of participatory rural appraisal techniques for developing land-quality indicators for sustainable land management for sloping lands in Indonesia, Thailand, and Vietnam. The framework for evaluating sustainable land management was used to develop threshold levels for the sustainability of land-management indicators. Campbell *et al.* (2001) proposed linking indicators to changes in five livelihood assets (natural, physical, financial, social, and human capital). They suggest a collective measure for each of the capital assets that could be used to develop an aggregate index. Although the aggregate index gives little guidance as to what needs to be included under each asset category, it might serve as an organising framework to develop a few indicators under each asset category for those projects expected to have wide-ranging impacts.

Once the relevant changes are identified and quantified through appropriate indicators, the next question becomes – how to value these changes? There are many vexing issues on how ecosystem service flows are valued. Even if effects can be predicted and monetary valuation is possible, many still argue if money values could adequately inform decision-making, especially when irreversible changes, trade-offs, and distributional effects are involved. The term 'value' may also have different concepts and meanings for different disciplines (Bingham *et al.*, 1995; Bockstael *et al.*, 2000; Farber *et al.*, 2002). In common usage it means 'importance' or 'desirability'. An economic value measures the change in well-being associated with the change in the quantity or quality of the service flow. Changes in resource and environmental service flows can affect human welfare in complex ways and through marketed or non-marketed activities. The most common approach to translating these changes into monetary units is to express the welfare change as the amount a person would pay or be paid (in compensation) to be as well-off with the change as without it. The amount that individuals are willing to pay, or might accept as compensation, is not an absolute value; it will vary across individuals depending on property rights, perceived welfare gains/losses, the context, and the availability of substitutes.

There are two key questions that need to be answered in the process of economic valuation of ecosystem services. The first is how to construct a measure of how much better or worse-off an individual is because of the change in the quantity or quality of the service flow. The second is how to add up the individual welfare changes (gains and losses) to assess the value of this change for society as a whole. Recent advances in economic theory provide answers to these two fundamental questions and offer useful methods for the valuation of many ecosystem service flows regardless of the functioning of markets.

Valuation Techniques

Unlike agricultural products harvested in fixed time periods, environmental and ecosystem services associated with NRM interventions flow in real time on a continuous basis. Understanding the changes in ecosystem service flows, measuring and monitoring outcomes across time and space is very important for quantifying environmental impacts. The basic principles that guide valuation exercises and the different valuation methods relevant for NRM, including their strengths and weaknesses are discussed.

Theoretical foundation

The economic approach to the valuation of resources is based on the contribution of the resource to human welfare. Whether the good or service is marketed or non-marketed, its unit economic value is determined by the welfare contributions that it makes to humans. Changes in welfare are measured in terms of each individual's personal assessment of changes in well-being (Bockstael *et al.*, 2000). For traded commodities, the demand curve depicts the marginal willingness to pay (WTP) (or marginal benefit) for the good or service. The height of the demand curve at each point of the quantity demanded shows the maximum WTP for the commodity. The household will consume all units of the commodity where the marginal WTP exceeds the market price. The consumer enjoys a consumer surplus for all points where the marginal WTP is higher than the market price. The welfare change associated with a change in the price of a marketed commodity is often measured using the change in consumer surplus, derived from the Marshallian demand curve with a constant level of income. For a non-marketed ecosystem service, the maximum WTP for an improvement in quantity or quality is the area between the initial and new levels of the resource under the demand (marginal benefit) curve. Value estimation then involves determining directly or indirectly the shapes of these marginal WTP curves for the ecosystem services (Freeman, 1993).

Environmental and resource service flows typically exhibit public-good characteristics of high costs of exclusion and non-rivalry. This makes it very difficult for markets to value these ecosystem goods and services accurately, and leads to a market failure and non-tradability. In order to illustrate how the values for such non-marketed resources could be estimated, let us assume that a given household maximises its welfare (U) from consumption of a vector of marketed goods (c), ecosystem goods and services (q) and has a fixed budget y, such that:

$$Max\ U = U(c, q) + \lambda(y - p'c) \tag{1}$$

The standard utility-maximising solution to this problem will give the Marshallian demand function for the tradable commodity:

$$c^* = c(p, q, y) \tag{2}$$

which is a function of a vector of market prices (p), the disposable income (y) and the ecosystem services (q) considered to be a public good. If this is substituted into the utility function, the indirect utility function could be derived:

$$v(p, q, y) = U(c(p, q, y), q) \tag{3}$$

The marginal effect of the change in the level of the public good q_i on household welfare can be derived as:

$$\frac{\partial v(p, q, y)}{\partial q} = \frac{\partial U\left(c(p, q, y), q\right)}{\partial q} \tag{4}$$

This is equal to the marginal valuation of the environmental good in question. It is a measure of the marginal welfare benefit (demand curve) for the public good q (Johansson, 1987). For a given change in q from q^0 to q^1, the welfare effect on household h can be estimated as:

$$\Delta U^h = v^h(p, q^1, y) - v^h(p, q^0, y) = \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, y)}{\partial q} \right] \tag{5}$$

The total welfare effect (WTP) summed over all the affected households (h) can be calculated as:

$$\sum_h \Delta U^h = \sum_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, y)}{\partial q} \right] \tag{6}$$

In general utility functions are unobserved and it would be useful to convert Equation 6 into a monetary measure of welfare change. This is done by assuming constant marginal utility of income (λ_h) for each household, and dividing the marginal valuations in Equation 6 by (λ_h). This is the same as vertical summation of the demand curves and will provide the aggregate uncompensated WTP for all the affected households given the change in q from q^0 to q^1. The compensating surplus (*CS*) and equivalent surplus (*ES*) measures (analogous to the *CV* and *EV* measures for price changes) can also be directly derived from the indirect utility function. For an improvement in q from q^0 to q^1 the *CS* and *ES* measures can be computed as:

$$v(p, q^1, y - CS) = v(p, q^0, y) \tag{7.1}$$

$$v(p, q^1, y) = v(p, q^0, y + ES) \tag{7.2}$$

In terms of empirical applications, the expenditure function that can be derived from the indirect utility function using the envelope theorem by solving for the expenditure level (y) that will provide a given level of utility, can be very useful in directly estimating the monetary measure of the welfare change associated with provision of the public good (q). The expenditure function for household h is given by $e^h(p, q, \overline{u})$. The aggregate welfare change measure for a change in q from q^0 to q^1 for *CS* can be given as:

$$CS = \Sigma_h \left(e^h(p, q^0, u^0) - e^h(p, q^1, u^0)\right) = \Sigma_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, u^0)}{dq} \right] dq \tag{8}$$

The aggregate *ES* measure for a change in q from q^0 to q^1 can be given as:

$$ES = \sum_h \left(e^h(p, q^0, u^1) - e^h(p, q^1, u^1)\right) = \sum_h \int_{q^0}^{q^1} \left[\frac{\partial v^h(p, q, u^1)}{dq}\right] dq \tag{9}$$

The *CS* is the maximum amount of money that the individual is willing to pay to secure an increased provision of the public good q. The *ES* measures the minimum sum of money that must be given to individuals before the change to make them as well-off as they would have been following an increase in q. This forms the basis for valuation of non-marketed ecosystem services. When the environmental and resource flows serve as inputs in production of market goods by producers, the equivalent welfare measure of the change in productivity is the change in producer and consumer surplus (Ellis and Fisher, 1987). Freeman (1993) demonstrates the other indirect benefit estimation approaches where q enters the production function as a factor input or as an input in the household production of utility-yielding commodities. Before the various methods and approaches used for eliciting values for ecosystem services are discussed, the components of the total economic value and the effect of markets and externalities on the choice of valuation methods are briefly demonstrated.

Valuation of impacts

The valuation of changes in ecosystem services needs to take into account intended and unintended outcomes. Individuals may attach values for such changes because of the use benefits derived, or any anticipated or conceived non-use welfare benefits. Agricultural activities often impose external costs on society mainly because individual resource-use decisions occur at points that equate marginal private benefits and costs. Soil erosion and sedimentation, and use of fertiliser, pesticides and other chemicals are some examples that impose costs on other agents and ecosystems. Unintended economic effects that spill over to other agents are often called externalities. More formally, externalities are unintended effects on the production or consumption activities of an economic agent resulting from the activities of another economic agent that are not mediated through markets. Adoption of 'best practice' NRM technologies like integrated pest management (IPM) or upland watershed management reduces external costs imposed on ecosystems and on other farmers in the lower reaches of the watershed. Hence, NRM investments may provide multiple ecosystem services to different economic agents across spatial scales as illustrated in Table 2.2 (Pagiola *et al.*, 2002). The use value (*UV*) of a given NRM investment includes the sum of direct and indirect use benefits (marketed and non-marketed) that accrue to all beneficiaries on-site and off-site. The challenge is how these dispersed benefits could possibly be valued. This requires good knowledge about the nature of the effect, how long the effect will last, its spatial dispersion, and the affected parties. The use value of the resource to different groups of economic agents cannot exceed the perceived benefits accruing to the group. Therefore, local forest managers,

Table 2.2. Perceived (on-site and off-site) benefits from integrated watershed management investments (soil, water and vegetation).

Ecosystem goods and services	Local communities	Downstream water and land users	Distant stakeholders and global community
Supplemental irrigation			
Improved agricultural productivity			
Fuelwood, pasture and construction materials			
Reduction in flooding and siltation			
Water purification			
Carbon sequestration			
Biodiversity preservation			

for example, will not consider the water quality captured by downstream water users, and biodiversity benefits to the global community. In this case the total use value of the resource will be given as:

$$UV_i = \sum_{i=1}^{n}\sum_{j=1}^{m} \gamma_{ij} Y_i \qquad (10)$$

where γ_{ij} is the distribution parameter reflecting the ecosystem service i captured at location j, and Y_i is the total use value of the ecosystem service i. Thus, the UV is the sum of all ecosystem services captured by all the beneficiaries across locations. Obviously, this poses practical difficulties in mapping out the benefit dispersion and in elicitation of values from different agents.

The total economic value (TEV) of a given resource may however include non-use values. The non-use values (NUV) include what are called option value (OV), bequest value (BV) and existence value (ExV). Figure 2.1 illustrates the components of the total economic value. OV is a measure of how much individuals are willing to pay for the option of preserving the asset for future personal use. BV is the value that individuals are willing to pay to ensure that the resource will be preserved for future generations. ExV is the value that individuals attach to the mere existence of a given natural resource or environmental asset unrelated either to current or optional use. Thus:

$$\begin{aligned} TEV &= UV + NUV \\ &= (\textit{Direct use value} + \textit{Indirect use value}) + (OV + ExV + BV) \qquad (11) \end{aligned}$$

The nature of the externality and the structure of markets will have substantial implications on the choice of effects to be valued and the valuation methods to be used. This can be seen by relating the anticipated benefits across spatial scales and the existence of markets to value these benefits (Dixon *et al.*, 1994). As can be seen from Table 2.3, the benefits from goods and services in Group I are both tradable within the local economy and are captured on-

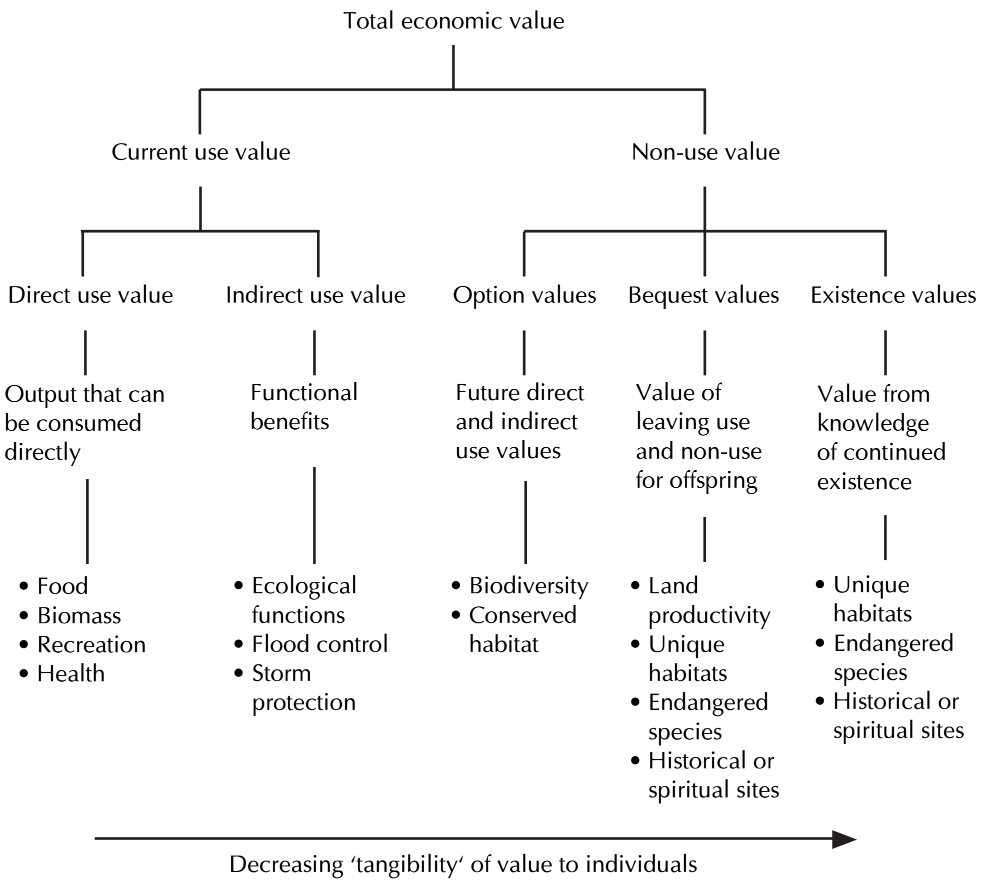

Fig. 2.1. Decomposition of total economic value of ecosystem goods and services (Munasinghe and Lutz, 1993).

site. These goods and services could be valued using market prices, with adjustments for any distortions that may exist (e.g. monopoly, externalities, and existing taxes and subsidies not reflecting external costs and benefits). For goods and services in Group II, market prices may exist, but local producers do not capture benefits, i.e. the lion's share of such benefits is 'externalised'. To the extent that these external benefits can be quantified, they could be incorporated into the social economic analyses of NRM impacts.

For those in Group III, benefits accrue within the local economy (of the household or village) but many of the goods and services are non-tradable. Missing markets mean that such benefits, however large, are seldom included in empirical impact assessments. Even the most difficult for valuation are goods and services generated from NRM investments, which are neither captured by the producers nor traded through markets (Group IV). Examples in this category include benefits of climate regulation (carbon sequestration) and biodiversity conservation resulting from upland

Table 2.3. Valuation of ecosystem goods and services from tree planting: The role of markets and externalities.

	Location of goods and services	
Tradability	On-site	Off-site
Marketed	**Group I** Benefits accrue on-site (e.g. fuelwood, fodder, timber, etc.) and are tradable – Usually included in impact assessment (IA)	**Group II** Off-site tradable benefits (e.g. higher crop yields or more hydropower resulting from reduced siltation in dams) – Sometimes included in IA
Non-marketed	**Group III** Benefits accrue on-site but are highly non-tradable (e.g. soil and water conservation, recreation, regulation of micro-climate, etc.) – Seldom included in IA	**Group IV** Off-site non-tradable benefits (e.g. Carbon sequestration, reduced flooding, biodiversity conservation) – Usually ignored in IA

Source: Modified based on Dixon *et al.* (1994) and others

tree planting. In the absence of markets, such non-tradable and external benefits need to be estimated by other methods using surrogate markets or constructed (hypothetical) markets in areas where the benefits are captured. Because of non-excludability, the level of investment by local producers to generate such services may be socially sub-optimal, necessitating many governments to intervene in their production and management.

Valuation methods

Advances in resource and environmental economics in the last few decades have provided many useful methods that can be employed to value use and non-use values of ecosystem goods and services, both marketed and non-marketed (Mitchell and Carson, 1989; Braden and Kolstad, 1991; Freeman, 1993; Bateman and Willis, 1999). The intention here is not to review the extensive literature on environmental valuation but to highlight briefly the methods that can be applied for valuation of NRM impacts in the context of developing countries. Table 2.4 presents an overview of some of these methods that can be potentially applied to value ecosystem services resulting from NRM investments. The methods can be distinguished by the type of market used, as well as the implied behaviour of the economic agent in the valuation of goods and services. Changes in productivity, replacement cost, avoided cost and opportunity cost methods use actual markets, but

Table 2.4. Valuation methods for ecosystem goods and services.

Implied behaviour	Conventional market	Surrogate market	Constructed market
Based on potential or stated behaviour	Productivity change approach (PCA) Replacement cost approach (RCA) Avoided costs Opportunity costs		Contingent valuation (CV) method Choice modelling (CM)
Actual or revealed behaviour	Provision costs Defensive expenditures Relocation costs	Hedonic pricing (HP) (land value) Wage differentials Travel cost	Experimental markets

behaviour is *assumed*, i.e. no actual outlays or market transactions have occurred. Defensive expenditures, provision costs and relocation costs also use actual markets but behaviour is *expressed* or *observed* through conventional markets. Methods like hedonic pricing and wage differentials use surrogate markets to value non-tradable goods and services indirectly through marketed goods and services that embody their values. In the extreme case, it may be possible to construct experimental markets (where behaviour is *revealed* with actual WTP or accept compensation for a change) or hypothetical markets (where behaviour is *stated* without actual transactions as in the case of the contingent valuation method). Methods that use information from conventional markets are presented below followed by those that use surrogate markets and those requiring constructed markets. The treatment gives more emphasis to methods that offer high potential for application in NRM impact valuation.

Productivity change approach (PCA)

Some agricultural resource-improving investments lead to changes in productivity and/or production costs. This means that physical changes in production or overall farm profits derived from adoption of such technologies can be established and valued using market prices.[1] This approach is quite attractive and suitable for evaluating NRM impacts, because physical changes in productivity can be observed and measured. Production functions, erosion damage functions, fertiliser response functions, crop–water responses to supplementary irrigation, and simulation models that relate changed resource conditions to productivity are good examples of PCA. A number of studies on the economic costs of land degradation, soil erosion, etc. have used this method to value the benefits from resource management investments. Magrath and Arens (1989) used detailed erosion–yield relationships to measure the on-site costs of soil erosion in Java, Indonesia. The capitalised cost of a 1% productivity decline is estimated to amount to 4% of the total value of dryland crops in Java. They also estimated off-site costs of sedimentation in reservoirs, irrigation systems and harbours, and found that these costs are about a quarter of the total erosion damage costs. Bojo (1991) used this approach to value the economic cost of soil degradation in Lesotho. However, Barbier (1998) noted these case studies probably overestimated the scarcity value of soil, because the economic value of conservation was not assessed as a net gain in profitability over the erosive (conventional) system. When there are no economically viable options for mitigation, soil degradation might not have on-site opportunity costs (i.e. on-site costs of soil degradation exist only when conservation is profitable on-farm). Shiferaw and Holden (2001) estimated various erosion–yield functions to evaluate conservation benefits and the net gain to farmers from adoption of conservation methods. Gebremedhin *et al.* (1999) also estimated returns to investments in terracing in the Ethiopian highlands, using experimental data on crop yields under different conservation methods. This is a good approach for valuing the economic cost of soil degradation or for evaluating conservation benefits.

Although production functions with resource conditions as factor inputs (along with other usual input factors) can be used to estimate the economic value of the resource, lack of such data often limits the application of this approach. One major difficulty is that it takes a long time for NRM investments to have an observable effect on the flow of ecosystem services. The first step in applying this method is to quantify the effect of changes in NRM on the quantity or quality of the resource base that affects resource productivity. For example, changes in NRM may affect rooting depth, water-holding capacity or organic matter levels in the soil. In this way, the condition of the resource stock (S) will be a function of the conservation effort and other exogenous characteristics such that:

$$S = g(K, Z) \tag{12}$$

where K is the level of NRM investment per ha, and Z is a set of exogenous factors (e.g. soil type, agroecological zone, rainfall, etc.). When the experimental data needed to estimate this statistically are lacking, simulation models may be used to estimate the effect of the change in K on the condition of the resource or the flow of ecosystem services (assuming that suitable parameters are available for the simulation models).

Moreover, the changes in productivity associated with changes in K may take a long time to be visible to farmers. Use of such other inputs as fertilisers and high-yielding varieties also often mask NRM investment benefits to farmers. When data that relate crop productivity with input use and biophysical conditions (e.g. soil depth, soil moisture, soil types) are available, econometric methods can be used to establish useful relationships such that:

$$Q = f(X, S, Z) \tag{13}$$

where Q is the productivity of land, X is a vector of inputs used, S is a vector of resource quality indicators, and Z is a vector of other exogenous factors that influence crop productivity. Controlling for variable inputs and fixed exogenous factors, the marginal effect of the anticipated change in the quality of the natural resource will be given as:

$$\partial Q(\cdot)/\partial S = f_S(X,S,Z)\Big|_{\{X=\overline{X},\, Z=Z_i,\, \forall i\}} \tag{14}$$

In some cases, the level of input use may not remain unchanged, e.g. improvement in soil fertility may prompt credit-constrained farmers to reduce the demand for fertilisers to produce the same level of output. In such cases, the productivity change associated with changes in NRM can be measured using the savings in input costs.

As shown in Equation 14, the marginal effect of NRM investments will depend on the quality of the resource stock and other exogenous factors. The total effect on productivity can be estimated by integrating over the level of change in *S* resulting from the change in *K*.

$$\Delta Q = \int_{S_0}^{S_1} \left(\frac{\partial Q(\cdot)}{\partial S} \right) dS \tag{15}$$

where S_0 is the old and S_1 the new levels of the resource condition associated with changes in *K* (NRM investments). The economic value of the change in NRM needs to be calculated as a producer surplus by including the opportunity cost of the variable inputs used in production (including the cost of *K*). If the change in output does not induce price changes, the producer surplus will be the value of the change in output minus the cost of production. The productivity changes may flow at different rates as the resource condition changes from S_0 to S_1 over a period of time. If the productivity changes are long-lasting, the present value of net productivity benefits (producer surplus) can be computed using the social rate of discount. While the approach is attractive and widely used, it has some disadvantages. These include high data requirements (when the econometric approach is used), lagged effects of NRM that hinder reliable assessment of productivity changes on-site, and difficulties in accounting for any off-site (externality) effects of the change in NRM. This last limitation is perhaps the greatest, because the PCA approach does not measure the value of non-marketed environmental goods and services.

Replacement costs approach (RCA)

Under this approach, potential expenses that may be needed to replace or restore the damaged natural resource asset are estimated using the prices of marketable products. The resulting estimate is not a measure of the benefits of avoiding the damage in the first place, since the damage cost may be higher or lower than the replacement cost. The implied expenditure to restore a given resource to a pre-damaged state or baseline condition may however be different from the costs of replacing its functions. Because of this, the RCA is mainly used in the latter context where the estimated resource values reflect the potential expenses needed to replace the services of the damaged resource through some substitutes (e.g. use of fertilisers or other fertility management practices to replace lost soil nutrients). Replacement costs can be a valid measure of economic value when the following conditions are met (Dixon *et al.*, 1994; Bockstael *et al.*, 2000):

- The magnitude of the damage is measurable and there are no secondary benefits associated with the replacement expenditure
- The substitute provides functions that are similar to the lost ecosystem service
- The substitute is the least-cost option for replacing the lost service
- Affected individuals in aggregate would, in fact, be willing to incur these costs if the natural functions were no longer available
- When the replacement costs are greater than the aggregate WTP or the social value of the productive resource destroyed, it will be economically inefficient to replace the damaged ecosystem service.

As Barbier (1998) noted, when these conditions are not fulfilled and least-cost replacement options are not known, simplistic application of the approach could lead to overestimated and misleading values. For the case of soil erosion, he noted that some of the eroded soil may be deposited on-farm and cannot be considered lost completely. Moreover, all the eroded soil might not have economic value if its marginal productivity effect is negligible. In these situations, the RCA can lead to overestimated resource values. By definition, the RCA includes only the costs of replacing damaged ecosystem services on-site, but the concept is equally applicable for valuation of any associated external effects. While the full restoration costs may include non-use values, the replacement costs reflect the use value of the resource or ecosystem service.

A number of studies have used this method. One example is the case study by Kim and Dixon (1986), which assessed the viability of alternative soil conservation techniques in upland agriculture in Korea. The difference in the estimated cost of physically replacing lost soil and nutrients (estimated based on differences in soil erosion) was taken as a measure of the potential benefits of preventing soil erosion. With the assumption that the value of retaining productive soil is higher than the replacement cost, the study found that preventive measures were more economical than physically replacing lost soil and nutrients.

Provision costs

Economic values for non-market ecosystem services that contribute to human welfare can sometimes be derived from people's decisions to use related resources or to substitute other resources where the quality of the service flow is impaired. The provision costs approach (PCA) refers to the actual expenditures that farmers or communities may incur to provide vital environmental goods and services. Unlike the mitigating expenditures, these expenses are directly targeted in the provision and production of the required good or service. While it can be considered as a variant of the RCA, the PCA does not refer to restoration of the ecosystem service, but to costs of providing the damaged service through alternative means. Some examples include farmers' expenses on drilling wells for irrigation and drinking water when water regulation services of watersheds are damaged, and the costs of alternative sources of household energy after deforestation. The strength of the method is in trying to value the resource in question using the actual cost outlays in producing the required good or service. However, the costs may also serve other purposes, and external benefits are excluded when private provision costs are considered. The method also relies on existence of markets for major inputs used in the production of the environmental good or service.

Defensive expenditures

Farmers, communities, and governments often incur actual expenditures to mitigate or prevent productivity loss or reduce degradation problems. When the extent and potential effect of resource degradation or improvement is difficult to assess, actual preventive or defensive expenditures may be used to assess a rough value of the change in the resource quality. Kim and Dixon (1986) use lowland farmers' defensive expenditures to prevent deposition of silt on rice fields to evaluate alternative soil management techniques designed to stabilise upland soils. There are several problems in the use of this method. Firstly, defensive expenditures, like all WTP, are limited by income and the value so obtained may not reflect the social scarcity value of the resource. It may at best be a lower-bound estimate. Secondly, the value tends to be quite arbitrary as actual expenditures may be targeted to meet several objectives (e.g. conservation of multiple resources).

For use in NRM impact assessment, it is important to determine the anticipated change in resource conditions attributable to the intervention, and how much farmers often spend to prevent an equivalent deterioration in the resource. If defensive expenditures on-site and off-site can be estimated, they may provide a rough indication of the value of the improvement in the resource. In some cases, relocation costs associated with environmental change can be considered part of defensive expenditures. Hence, the relocation costs approach is not discussed separately.

Hedonic pricing (HP)

The theory of hedonic prices is based on the premise that market prices reflect a bundle of observable characteristics and attributes of differentiated products (Rosen, 1974). Different attributes of the same product reflecting differences in its inherent worth will have an associated price, and consumers can easily identify what they are paying for in selecting various options. When goods and services contain non-priced environmental attributes embedded in them, consumers may also place implicit values on each of the attributes so that market prices are composed of environmental and non-environmental attributes. Therefore, when the good or service provided by NRM investments cannot be directly valued using conventional markets, behaviour revealed through surrogate markets can be used for valuation.[2] For example, the value of access to clean water and air can be estimated indirectly through the differences in market prices for houses in polluted and clean localities, after controlling for their structural and other attributes (Harrison and Rubinfeld, 1978). Wage differentials for occupations with different levels of health or environmental risk have also been used to estimate certain environmental values. The HP method is designed to control for certain non-environmental attributes so that the remaining property value differentials can be 'surrogate' values of the non-priced environmental goods and services.

To the extent that surrogate markets are competitive, the HP approach can therefore be very useful for valuing NRM impacts. For example, land values in competitive markets can be used to value differences in land quality. If prices for agricultural land reflect quality changes, the hedonic function for a given parcel with a vector of biophysical (environmental) characteristics $L = (l_1, l_2, \ldots, l_n)$ and socio-economic characteristics of the location and the buyer $Y = (y_1, y_2, \ldots, y_n)$ can be estimated econometrically as:

$$P = g(L, Y) \tag{16}$$

where P is the market price of a unit of land. The socio-economic characteristics include such variables as buyer characteristics, population density, distance to urban areas, distance to markets, and type of crops grown. The coefficients of this model can be used to determine the implicit price associated with land characteristic, holding all other factors constant. For example, for soil characteristic l_i (e.g. soil depth) the implicit price is the partial derivative with respect to soil depth such that:

$$\frac{\partial P}{\partial l_i} = g_l(L, Y) \tag{17}$$

If the impact of NRM investments on the biophysical conditions of the resource is known, market prices can be used to value indirectly the changes in resource attributes. One disadvantage of this method is that it requires extensive information on selling or rental prices and associated socio-economic and biophysical characteristics of the property. Even when such data are available, market prices may not be competitive or may not fully reflect such non-observable quality differentials as changes in nutrient balances or biophysical attributes of the soil. The method works quite well if markets reflect quality differentials. Even when they do, market prices may reflect only the capitalised value of future on-site productivity gains from using the land. Changes in non-productivity benefits (e.g. biodiversity, carbon sequestration) and reductions in off-site effects from NRM investments might not be reflected in market prices. In a recent study Shiferaw *et al.* (2003) found that farmers' perceived value of land parcels in semi-arid Indian villages were able to clearly reflect soil and farm characteristics that affect land productivity. Factors such as irrigation, soil depth, soil fertility levels, and soil type had significant effects on perceived land values. For example, irrigated plots, *ceteris paribus*, were perceived to have values 45% higher than non-irrigated plots, whereas a one-level rise in ordinal soil depth increased land values by 5% and in soil fertility by 18%. Such other factors as conservation investments and erosion risk were found to have no significant effects on land values. This shows that the land value method can be used as an alternative to PCA for valuing the effect of NRM investments on land quality aspects that influence productivity. Due to market failures and imperfections, including incomplete land tenure rights, changes in other attributes like public goods aspects and non-use values cannot be easily valued using the land value approach. The contingent valuation (CV) method is useful for valuing such changes.

Contingent valuation (CV) method

In cases where people's preferences are not revealed directly or indirectly through conventional markets, the CV method is used to assess their WTP for marginal changes in quantity or quality of goods and services by posing hypothetical questions. The CV method is a direct stated preference method that involves asking a sample of a relevant population questions about their WTP or willingness to accept (WTA). The monetary value of the change in NRM is acquired by asking respondents about their WTP for a benefit, or what they are WTA by way of compensation to tolerate a cost or forgo a benefit. The name *contingent valuation* originates from the fact that the valuation is contingent on the hypothetical scenario put forward to the respondent. CV is mainly used for valuation of non-marketed ecosystem services and the non-use values associated with non-excludable and non-divisible resource and environmental flows. Unlike the indirect methods that use observed or revealed behaviour, the CV method relies on stated or potential behaviour as expressed in hypothetical markets. An important advantage of the CV method is that responses to WTP and WTA questions provide theoretically correct measures of welfare change as defined in Equations 5–9.

As discussed earlier, the appropriate welfare measures for changes in environmental quantity or quality are compensating surplus (CS) and equivalent surplus (ES) measures. Theoretically, an individual can be asked about WTP or WTA for either an improvement or a deterioration (Table 2.5). Which question is appropriate depends on the implied property right for the specific situation. The CS measure relates to the initial welfare level and implies entitlements to the *status quo*. Thus, asking about WTP to secure an improvement, or WTA compensation to tolerate a loss, implies that the individual is entitled to the existing level. The ES measure relates to the welfare level after the change and suggests the implied property rights in the change. Asking about WTA compensation to forgo an improvement implies an entitlement to the higher level, while WTP to avoid deterioration implies an entitlement to the lower level. WTP is also constrained by income whereas WTA is not. As a result, estimates of WTA tend to be higher than WTP. Some authors suggest using WTP for situations where individuals are expected to gain from an improvement and WTA in situations where people are forced to give up or suffer some damage to their welfare (Carson, 1991). Mitchell and Carson (1989) discuss ways to frame the payment questions to elicit WTP. Arrow *et al.* (1993) provide a guide for best-practice CV studies.

Table 2.5. Welfare measures for environmental quality and quantity changes.

	Compensating surplus (CS)	Equivalent surplus (ES)
Improvement	WTP[a] for the change to occur (to secure a benefit)	WTA[b] compensation for the change not occurring (to forgo a benefit)
Deterioration	WTA compensation for the change occurring (tolerate a loss)	WTP for the change not to occur (to prevent a loss)

[a]WTP = willingness to pay.
[b]WTA = willingness to accept.

In a nutshell, application of the CV method requires the following steps:

- Create a survey instrument to elicit WTP/WTA and the means of payment or compensation
- Administer the survey instrument with a sample population
- Analyse the responses and estimate the average and marginal WTP/WTA
- Estimate the total WTP/WTA for the population of interest.

In developed countries, various survey methods including mail and telephone surveys have been used. In developing countries, in-person interviews remain the most feasible and reliable option. Such surveys often start with discussions with key informants and focus groups followed by pilot testing of the survey format. The actual data collection should introduce the changes in the resource or environmental conditions being valued and the expected benefits or trade-offs to society resulting from this change. Pictures and maps can be used to illustrate these points. The survey should also include standard data on the socio-economic condition of the respondent (e.g. age, education, assets, income, etc.). Various approaches to eliciting WTP or WTA are suggested. Open-ended questions like 'What is the most you are willing to pay for…' or 'What is the minimum that you are willing to accept as compensation for …' have been commonly used in the past. This approach has been criticised for inviting strategic bias, by which respondents may use their replies to influence a more favourable research outcome (e.g. to reduce a payment they might expect to have to pay). In actual markets, buyers are offered a price and may bargain from there to arrive at the selling price. Many respondents find the open-ended approach difficult and fail to provide any bids. The iterative bidding approach that starts with an initial amount to be revised up or down until a no-change point is reached, has been used as an alternative to open-ended questions. This approach is now being abandoned because of a starting-point bias, i.e. the WTP/WTA amount tends to be systematically related to the initial bid value. An alternative approach that is gaining popularity is the binary choice or referendum format, where respondents are asked whether they would vote in support of a proposed change in policy or environmental condition that would cost a US$$x$ increase in tax payments. The offered amount can be varied and randomly assigned to the sample. Follow-up questions to the binary choice payment questions have also been used to identify the upper and lower bounds for the bids. It seems that depending on the design, a discrete-choice format with follow-up questions can mimic a bargaining process, commonly used in transactions in developing countries (FAO, 2000).

Once the data from a representative sample are collected, statistical analyses will be needed to estimate the average WTP/WTA and the aggregate value of the ecosystem service. The type of analysis of CV responses depends on the elicitation format used. If the payment question is open-ended, the stated WTP/WTA bids can simply be averaged.[3] The sample average is an unbiased estimator of the population mean. In cases where outliers influence the average bid, the median is a best estimate of a representative central value. As defined earlier, the WTP can be given as:

$$WTP_h = e(p, q^0, u^0, x) - e(p, q^1, u^0, x) \quad (18)$$

where $e(.)$ is the expenditure required to attain a given level of utility, WTP_h is the *WTP* for household *h* and $\boldsymbol{x}$ represents the socio-economic characteristics of the respondent and other exogenous variables that affect the WTP. In order to check the internal validity of the CV method, a regression model can be fitted as:

$$WTP_h = X_h\,\beta + \eta_h \quad \eta_h \sim (0, \sigma^2) \quad (19)$$

where X_h is a vector of explanatory variables and η_h is the error term distributed normally with means *0* and standard deviation σ. This function is often called the valuation function. As shown below, the valuation function is especially relevant for use in benefit transfer studies. It allows the new user to plug in mean values of explanatory values to predict the benefit value for a new setting. If the binary choice payment format is used, alternative methods for estimating the mean bid are discussed in the literature (Hanemann, 1984). Additional analyses will be needed to estimate the average bid and aggregate values for the change. The binary response is an indicator for the WTP/WTA that is observed only when the respondent's WTP/WTA is less than the offered bid value. A maximum likelihood probit model can be estimated using these binary responses to identify the factors that determine the probability of a positive response to a given bid. The mean WTP/WTA can then be obtained by calculating the predicted value of the valuation function at the mean values of the covariates.

Once the average WTP or WTA values for a representative group of people have been determined, they are aggregated to a total value directly dependent on the number of individuals affected. For ecosystem services that provide international public goods, the number of people with a positive WTP is likely to be large, and a modest estimate of the population size needs to be made. In principle, scaling up the average WTP/WTA across the affected population is analogous to the vertical summation of individual compensated demand curves for public goods.

As the examples in Box 2.1 demonstrate, carefully designed and administered CV surveys can provide useful estimates of the value of the changes in non-marketed ecosystem services resulting from NRM investments. The reliability of estimates and validity of results depend on the design and implementation. Of course, they also share the weaknesses of all stated-preference methods.

Box 2.1. NRM-related CV studies in the developing countries.

Today, there are several examples and good reviews of CV applications in developing countries (Munasinghe and Lutz, 1993; FAO, 2000; Pearce *et al.*, 2002). Whittington (1998) examines issues and lessons learned in administering CV surveys in developing countries. Two studies relate to NRM impact assessment in agriculture. Holden and Shiferaw (2002) applied the CV approach to estimate farmers' WTP to mitigate soil degradation in Ethiopia. In the light of increasing land degradation in the highlands, the intention was to elicit the farmers' WTP for NRM technologies that might not provide immediate benefits to farmers. The survey questions were framed to reflect the attributes of available and proposed NRM technologies with three alternative scenarios. Farmers were asked about their WTP for new NRM technologies that: a. sustain land productivity at current levels, b. enhance productivity by a fixed amount from the second year onwards, and c. enhance productivity by a fixed amount from the sixth year onwards. Teff (*Eragrostis tef*), the locally grown cash and staple cereal, was used as numeraire. The WTP surveys were administered as part of a larger survey where broader socio-economic data were collected that allowed estimating regression equations to identify the WTP covariates and check for internal validity. Farmers' expressed WTP for land management options was significantly lower than those implied by experimental and econometric estimates of soil erosion and productivity decline. Shyamsundar and Kramer (1996) applied the CV method to value forest ecosystem services in Madagascar using a binary choice payment format to elicit the local people's WTA compensation for welfare losses associated with land-use restrictions and loss of access to forests. Due to the extreme poverty of farmer respondents, Shyamsundar and Kramer used WTA questions specified in terms of bags of rice, the local staple food. They estimated a probit function and a valuation function to infer the WTA for specific households and the mean for the sample. This was used to estimate the aggregate use value of the forest service flows to the local people.

Although the approach has been widely applied for benefit–cost analysis of projects with environmental impacts, its use in assessment of technology and policy impacts in agriculture and natural resources has been scanty. CV surveys can be very useful for generating information that will inform policy choices in developing countries where market failures are more pervasive. The method is a relatively simple and cost-effective means (especially when literacy is widespread) to estimate the value of public goods and non-market ecosystem services associated with NRM investments.

Choice modelling (CM)

Choice modelling (also called choice experimentation) is an indirect stated preference method that arose from conjoint analysis and has been employed in marketing, transportation and psychology. Bennett and Blamey (2001) provided a collection of papers on the theory and application of CM in environmental valuation. Alpizar *et al.* (2003) provided a good review of using CM for non-market valuation. It differs from typical conjoint methods in that individuals are asked to choose from alternative bundles of attributes (alternatives) instead of ranking or rating them. Under the CM approach, respondents are asked to choose their most-favoured choice out of a set of three or more alternatives, presenting variations in the attributes of the item being valued (Adamowicz *et al.*, 1998). The *status quo* is given as one of the alternatives in the choice set. Each respondent gets the same number of choice sets, but the composition of the choice sets varies across respondents. This allows the researcher to value changes in attributes and the trade-offs compared to the *status quo* and different alternatives. Furthermore, in the case of damage to a particular attribute, compensating amounts of other goods (rather than compensation based on money) can be calculated. While several statistical methods can be used, multinomial and conditional logit models are commonly used to analyse the choices that people make. This approach can provide substantially more information about a range of possible alternative policies and can reduce the sample size needed compared to the CV method. However, survey design issues with the CM approach are often much more complex due to the number of goods that must be described and the statistical methods that must be employed. This may limit its application for valuation of NRM impact in the context of developing countries.

Comparison of alternative valuation methods

This section has reviewed the promising methods that can be used in valuation of NRM impacts. The choice of valuation methods depends on the existence of markets, the spatial and temporal diffusion of the impact, and whether the values relate to use or non-use values. Direct-use values such as productivity changes can be measured indirectly using data from observed or stated market behaviour of producers and consumers in conventional or surrogate markets. For non-use values, like benefits captured by future generations (sustainability) and indirect-use values such as ecosystem regulation functions, there is no observable market behaviour that contains relevant information, hence hypothetical behaviour in constructed markets must be used. The PCA, RCA, HP and CV methods are the most commonly applied in relation to environmental resources, and they offer promising opportunities for valuation of NRM impacts. Each of these methods measures different aspects of the total economic value (see Fig. 2.1) and has its strengths and weaknesses. The PCA and RCA use observed market information to measure use values indirectly. HP is also an indirect method that uses surrogate markets to measure use values. The CV is the direct stated preference method mainly used in respect to non-use values, but it could also be applied for use values.

Perhaps because of their relative ease and cost-effectiveness, the PCA and RCA are most commonly used in NRM valuation exercises. These two approaches measure different aspects of resource degradation focusing on productivity change and the costs of replacing damaged ecosystem services; hence, they often provide divergent estimates. The relative size of the two estimates may also be useful for NRM technology choice and farmer-investment decisions. Farmers are unlikely to adopt resource management practices unless the productivity benefits are higher than the investment costs. Drechsel *et al.* (Chapter 9, this volume) discuss this and the related pros and cons of these methods in more detail. The PCA needs to be computed as a net gain over the less-conserving alternative. When data on changes in resource conditions and productivity are available, the PCA is a recommended method for measuring the values of marketed productivity impacts. A major weakness is its inability to value external effects and non-market benefits.

The RCA imposes strict assumptions – that the substitute be the least-cost alternative and that the cost be less than the aggregate WTP. The RCA cannot be used to value ecosystem services that do not have marketed substitutes, and it cannot measure non-use values. When production data are limiting, RCA can be a useful alternative to value changes in certain resources like soil quality.

If markets reflect changes in environmental quality and resource conditions, the HP method is another promising technique for estimating benefits. The disadvantages of HP applied to land markets include lack of transaction data and failure of land markets to fully reflect non-productivity related changes in ecosystem services. When sales transactions are limited, land rental markets may provide an alternative source of relevant information.

When existing markets cannot be used to acquire the necessary information, the CV and CM methods can be the most useful approaches for NRM valuation. The strength of these approaches is their flexibility to generate information from constructed markets to measure both use and non-use values relevant to a given situation. When properly applied, the WTP/WTA estimates provide theoretically correct measures of welfare change. Although the survey design is more complex, the CM requires less data and provides more policy-relevant information than the CV method. However, these methods are criticised for their reliance on hypothetical markets where true behaviour is unobservable and also for survey implementation problems that may bias results. Several approaches can be used to reduce bias. If non-market ecosystem services and non-use values are a significant part of NRM impacts (as is often the case), these are the only conceptually justified approaches, and should be carefully applied depending on the availability of technical and financial resources.

When new valuation studies cannot be made due to time or financial constraints, the *benefit transfer approach* can be used to apply valuation estimates from other studies of similar changes in environmental quality at a new site. Although termed '*benefit transfer*', damage estimates can also be transferred. Four benefit transfer approaches exist: unit value transfer (e.g. direct transfer of mean WTP per household), adjusted unit value transfer (e.g. corrected for differences in per capita income levels), value function transfer, and meta-analysis. Value function transfer uses regression equations estimated for one location to predict resource values in another location, while meta-analysis uses independent case studies to synthesise and provide a summary estimate of resource value for specific conditions. Value transfer generally increases the uncertainty in the estimated environmental value. The early examples of benefit transfer were conducted in an uncritical manner, often lacking sound theoretical, statistical and empirical basis, and did not question the validity and reliability of the transferred values.

Recently, there has been growing interest in the development of benefit transfer methods and statistical techniques (Navrud, 2004; Navrud and Ready, 2004). Results from validity tests have shown that the uncertainty in spatial and temporal benefit transfer can be quite large, especially when economic and ecological conditions are quite different. Thus, care should be taken in using benefit transfer in policy uses where the demand for accuracy is high.

At present, there is a dearth of both benefit transfer applications in developing countries and sufficient valuation studies for meta-analyses. There is also a lack of validity tests of benefit transfer between developing and developed countries. One such study underlines the considerable uncertainty in using benefit transfer estimates (Barton and Mourato, 2003). Correcting for differences in gross domestic product (GDP) per capita seems to improve benefit transfer, but the actual difference in income levels in the two samples does not typically correspond to the differences in GDP, so correcting for income levels in unit-value transfers often makes things worse (Barton and Mourato, 2003; Navrud and Ready, 2004). Since the explanatory power of WTP functions is often poor, value function transfers may not do a better job in transferring benefits than simple value transfers.

Impact Evaluation

Since NRM interventions are expected to provide multiple economic and environmental benefits to various stakeholders including smallholder farmers, NRM impact evaluation should include non-marketed ecosystem goods and services along with marketed economic benefits. The market and non-market values of changes in goods and environmental services estimated using the valuation methods discussed in this chapter are vital in estimating costs and benefits that are used to evaluate the overall impact of the intervention. This requires a more holistic approach that would expand conventional impact assessments (Baker, 2000) to include non-tradable goods and environmental services. The welfare gains from NRM investments associated with direct economic benefits (e.g. yield gains or cost savings) can be assessed using a conventional approach. Unfortunately, as shown earlier, NRM investments generate other sustainability benefits and ecosystem services that have use and non-use values to people. Indirect welfare gains from such environmental improvements are legitimate parts of the welfare changes associated with NRM interventions and need to be considered in impact evaluation. The total welfare gain to people can then be decomposed into direct economic benefits derived from productivity changes and indirect environmental economic components. When NRM technologies generate productivity (including cost-saving) benefits in addition to changes in resource quality and sustainability, both sources of welfare gain are likely to be significant. In cases where the impact is expressed mainly in terms of non-tangible ecosystem service flows, the environmental and sustainability benefits could become a major part of the total welfare gain. Although precise estimation of non-market ecosystem service flows is always difficult, the valuation methods discussed above can be used to estimate the multiple welfare benefits associated with NRM interventions.

The conventional economic surplus approach (Alston *et al.*, 1995; Swinton, Chapter 7, this volume) includes changes in consumer surplus and producer surplus associated with supply shifts and price changes from changes in agricultural technology. As shown above, welfare gains associated with changes in environmental conditions are measured using the ES and CS measures of welfare gains to consumers and the producer surplus benefits to producers. When these extended economic and environmental welfare benefits are known, the social impact of research and development (R&D) investments in NRM can be evaluated using the benefit–cost analysis approach. The economic welfare gains from NRM can be given as:

$$\pi_t^P = \pi_t^{PN} - \pi_t^{PT} \tag{20}$$

where π_t^P is the period t productivity-related economic gain from change in NRM that can be calculated as the difference in net benefits between the new (π_t^{PN}) and the traditional (π_t^{PT}) NRM practices. π_t^P is essentially the flow of consumer and producer surpluses associated with productivity changes generated by NRM interventions. The environmental welfare gains from NRM can similarly be given as:

$$\pi_t^E = \pi_t^{EN} - \pi_t^{ET} \tag{21}$$

where π_t^E is the period t environmental welfare gains that can be calculated as the difference between environmental benefits from the new (π_t^{EN}) and the traditional (π_t^{ET}) NRM practices. This is the total WTP/WTA measure of welfare change resulting from changes in the flow of non-productivity related ecosystem services valued by people. π_t^E is essentially the social WTP for better NRM to enhance agricultural sustainability and the flow of ecosystem services (environmental quality). These values can also include the changes in external or off-site effects of NRM interventions. In order to assess the social impact of NRM interventions, additional information on the research, development and extension costs will be needed. If it is assumed that the flow of these costs is given by RE_t, such costs incurred up front could be quite significant, especially when the benefit flow is delayed because of the long time required for technology development and adaptation and when a positive discount rate is used in the calculation of net present values from the investment. The net welfare gain from NRM interventions will then be estimated as:

$$NPV = \sum_{t=1}^{n} \left(\pi_t^P + \pi_t^E - RE_t\right)\left(1 + r\right)^{-t} \tag{22}$$

where NPV is the social net present value of the NRM intervention, r is the real social rate of discount. Some of the changes in ecosystem services (e.g. soil fertility) may be reflected in productivity changes. The additive framework given in Equation 22 is valid when the productivity benefits and non-productivity related environmental or sustainability outcomes are clearly separable. When such separation is not possible, the approach can lead to double counting and overestimation of NRM impacts. An important area for further research is on the mechanisms used to separate productivity and sustainability effects, and under which conditions the estimated productivity and environmental values can be additive. If the social benefits of interventions are higher than the costs of the interventions, i.e. the total benefit is higher than the costs and gainers from the intervention can overcompensate losers, then NRM is considered to be socially beneficial. This may not be the case when externalities are ignored in the analysis. When environmental net benefits (π_t^E) cannot be estimated, the impact could be assessed in terms of the required tradeoffs and implications for sustainability of productivity gains. This could also include situations where the impact of NRM interventions is reflected in terms of reductions in production risk, improved stability of production, and reduced vulnerabilities of rural households to droughts, floods and other environmental shocks.

Parameters estimated for linking NRM changes with ecosystem goods and services (e.g. to estimate the effect of soil and water conservation on productivity) can also be integrated into bioeconomic models. The integration of important biophysical information and ecological processes with economic decision behaviour through bioeconomic modelling allows simultaneous assessment of welfare effects and environmental and distributional outcomes. Holden (Chapter 8, this volume) and Shiferaw and Holden (Chapter 12, this volume) further discuss these issues. One innovative approach for future research to evaluate the social impacts of NRM interventions is to compare the stream of aggregated net benefits (estimated based on optimised values derived from the model) with R&D investment costs.

Conclusions

The changes in environmental and resource service flows associated with NRM investments accrue over different temporal and spatial scales. Many of these ecosystem services generate valuable direct and indirect welfare benefits to people. When NRM investments generate private and public goods benefits, valuation of such changes is a crucial first step in the evaluation of overall social impacts. A prerequisite to effective valuation of NRM impacts is the ability to predict the changes in ecosystem service flows that can be attributed to the intervention itself. This requires a strong partnership between agroecologists and economists. The scientific understanding of ecosystem functions and services and how they are affected by human interventions is still incomplete. More work is needed to understand and quantify the effect of NRM interventions on ecosystem functions and services. Appropriate indicators are needed to measure selected changes in ecosystem services. Without reliable data, valuation efforts will not provide any useful economic values. With advances in agroecology and biophysical simulation modelling, the ability to predict the likely effects of certain interventions has improved. The economic approach to valuation of ecosystem services is based on the trade-offs that people are prepared to make in exchange for these services. The changes in the flow of ecosystem services can affect human welfare in complex ways and through marketed or non-marketed activities.

This chapter has offered an overview of ecosystem services from NRM investments, the need for indicators of ecosystem condition, key challenges to valuation of environmental services, and recent advances in the methods available for valuation of economic and environmental benefits. There is a dearth of examples in valuation of NRM impacts, especially in the context of developing countries. However, the recent progress in developing valuation methods has created new opportunities. For NRM impact assessment, the estimated values of changes in ecosystem services need to be social scarcity prices that account for non-marketed outcomes and external effects. Impact assessment of agricultural technologies has often ignored external effects and environmental impacts. However, resource management interventions typically generate non-marketed sustainability and environmental benefits. The greatest challenge in valuation of NRM impacts is in quantification and measurement of these non-productivity related outcomes and non-market benefits. Such standard techniques as the productivity change approach or revealed preference methods like defensive expenditures, provision costs or hedonic pricing can be used to measure productivity-related outcomes. However, markets and observed behaviour cannot be used for valuation of impacts on non-use and indirect use values related to regulation and habitat provision functions. When the impacts can be quantified using measurable indicators, stated preference methods could be used for the valuation of such effects.

The CV method is the most appropriate option when the indirect and surrogate market options cannot be used to value the change in ecosystem services. It is most appropriate for valuing non-use values and non-tradable use values of ecosystem services. However, the CV method has only rarely been applied to NRM impact assessment in the developing countries. Since poverty limits the ability to pay, the WTA compensation is a preferred approach for valuation of ecosystem services in poor communities. Choice modelling is an alternative and promising sated preference method. It is important to test and enhance these methods for valuation of non-market ecosystem services associated with NRM. Case studies are required to gain experience and develop improved protocols in the application of CV and/or CM methods for NRM impact assessment. In some situations benefit transfer approaches can be used to inform urgent policy decisions. However, more research is needed to enhance the transferability of benefits between countries or eco-regions.

Once the values of changes in ecosystem services are estimated, impact evaluation needs to compute the overall social gains from NRM interventions. Many NRM interventions imply supply shifts for both market and non-market goods and environmental services. This implies the need to estimate the size of the supply shift and the resulting effect on estimated unit resource values. However, more work is needed to understand how such values can be effectively integrated into impact assessment studies. In some cases, high uncertainties about the nature and magnitude of changes and the temporal and spatial impacts of NRM interventions may limit the policy relevance of monetary values. More research is needed to improve the validity and reliability of these estimates for use in policy analysis and impact assessments.

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Endnotes

[1]In *ex ante* assessments, prevailing market prices may not be an appropriate way to value productivity changes unless a 'small project' assumption is imposed so that prices remain largely unaffected. If productivity changes are expected to affect market prices, appropriate adjustments can be made using the general equilibrium framework.

[2]The travel cost method is another surrogate market approach for valuation of recreational use values of ecosystem services. It has been applied widely for the valuation of wildlife in protected areas. Since typical agricultural NRM investments do not provide marketable recreational benefits, the method is not discussed here.

[3]Apart from average WTP, marginal WTP is also of interest. This can be determined by estimating an inverse demand curve with price as a function of quantity. An inverse demand curve is also essential for estimating economic surplus rather than assuming constant average WTP.

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3 Measurable Biophysical Indicators for Impact Assessment: Changes in Soil Quality

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Introduction

Soil plays a key role as the interface between terrestrial and aquatic ecosystems on the one hand and the atmosphere on the other. The importance of soil in meeting food, feed and fibre needs and maintaining environmental sustainability cannot be overemphasised. A healthy or good quality soil acts as an environmental filter in cleaning air and water. Soil is a major sink for global gases and its appropriate management favourably affects the carbon dioxide (CO_2) balance that is important in combating global warming. If mismanaged, soil can work against us; it can pollute the air and water and lead to a fall in agriculture production.

Decline in soil quality has occurred worldwide, particularly in the semi-arid tropical (SAT) regions and is manifested as adverse changes in physical, chemical and biological soil properties and its contamination by inorganic and organic chemicals (Arshad and Martin, 2002; Lal, 2004). In many parts of world production of major cereals is declining mainly due to soil degradation coupled with inadequate soil and water management (Steer, 1998).

Natural resource management (NRM) interventions in terms of fertility, soil and water management practices in various farming systems have become necessary to address the problem of soil degradation and hence increasing investments in NRM research and development are being made worldwide. To diagnose and quantify the impacts of various NRM interventions, reliable soil quality indicators are necessary. Impact assessment is essential for the development of suitable management strategies for soil quality and to maximise productivity and sustainability for the benefits of society.

Appropriate and measurable soil quality indicators are needed to assess the impact of various NRM interventions on soil quality in agricultural lands. Measurable and simple soil quality indicators are important because many of the conventional soil attributes used to characterise soils become useful only after soil degradation has already taken place. To have soil quality indicators together with the soil quality thresholds needed to monitor and assess the impact of NRM technologies seems rather a tall order. Modern agricultural practices used to intensify agriculture have complicated the selection of such indicators, but several measurable indicators are available and can be used to assess the biophysical impact of NRM practices. Unfortunately, there is no universal set of indicators that is equally applicable in all cases, so the selection of those relevant to specific conditions is extremely important.

The objective of this chapter is to identify and discuss with examples from recent literature the use of biophysical indicators in monitoring the impact of NRM interventions on soil quality attributes. The use of simulation modelling to assess the long-term effects of NRM interventions on soil quality and future research needs are also covered.

Soil Quality Indicators

Soil quality indicators are measurable soil attributes that influence the inherent capacity of the soil to perform its production and environment-related functions. Attributes that are management-responsive are most desirable as indicators. During the past 10 years many definitions of soil quality with similar elements have been proposed (Arshad and Coen, 1992; Doran and Parkin, 1994; Karlen *et al.*, 1997). A recent definition was proposed by Karlen *et al.* (2003) and a committee of the Soil Science Society of America: "the fitness of a specific kind of soil, to function within its capacity and within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation"; this seems to be inclusive and appropriate for the objectives of this chapter.

It should, however, be mentioned that the soil quality paradigm has received several criticisms because of its general lack of sufficient quantification and scientific rigour (Letey *et al.*, 2003; Sanchez *et al.*, 2003; Sojka *et al.*, 2003). These authors believe that in assessing soil quality emphasis should be directed towards using available technical information to motivate and educate farmers on 'quality soil management' involving management practices that optimise the combined goals of high crop production, low environmental degradation, and sustained resource use (Sojka *et al.*, 2003). However, several scientists believe that with further refinement, soil quality indicators could provide a more useful tool for assessing soil quality. It may be useful to note that indicators for monitoring soil quality could also help towards developing quality soil management.

Scientists aim to develop a set of basic soil characteristics to serve as key soil quality indicators (Stott *et al.*, 1999) that are sensitive to climatic and management interventions. Ideally, the best soil quality indicators are those attributes or characteristics that show observable and significant changes between 1 to 3 years, with 5 years being an upper limit to usefulness.

Given the complex nature of soil and the exceptionally large number of soil properties that may have to be determined, it is important to be able to select properties that are appropriate and practical. Stephen (2002) grouped attributes that can be used as indicators of soil quality into four broad groups: physical, chemical, biological and visible indicators. Karlen and Stott (1994) proposed a framework for evaluating physical and chemical indicators of soil quality. Turco *et al.* (1994) discussed the various microbial indicators of soil quality. Arshad and Martin (2002) proposed selected physical, chemical and biological soil quality indicators (Table 3.1). In the light of diverse soil functions for which indicators are used, the quality indicators listed may not be sufficient to evaluate the changes in soil quality resulting from various agricultural and NRM interventions. Depending upon the local conditions, some may have to be added or excluded. These are discussed in turn below.

Table 3.1. Selected physical, chemical and biological soil quality indicators used to assess soil quality.

Soil quality indicator	Rationale for selection
Physical	
Top soil-depth	Estimates moisture availability, rooting volume for crop production and erosion
Aggregation	Indicates status of soil structure, erosion resistance, crop emergence can be an early indicator of soil management effect
Texture	Indicates retention and transport of water and chemicals
Bulk density	Shows plant root penetration and air-filled porosity
Infiltration	Indicates runoff, leaching and erosion potential
Biochemical	
pH	Indicates nutrient availability, sorption and desorption of molecules
Organic matter	Affects fertility, structure, water retention and sorption and desorption of molecules
Electrical conductivity	Defines salt content, crop growth, soil structure and water infiltration
Suspended pollutants	Affects food quality, water quality and human and animal health
Soil respiration	Indicates biological activity, biomass activity and organic matter quantity and quality
Form of soil N	Defines availability to crops, leaching potential, mineralisation/ immobilisation rates
Extractable N, P and K	Indicates capacity to support plant growth and serve as an environmental quality indicator

Source: Adapted from Arshad and Martin, 2002

Physical quality indicators

Physical indicators are principally concerned with the physical arrangement of solid particles and pores. They include soil texture, moisture-holding capacity, bulk density, porosity, aggregate strength and stability, crusting, surface sealing, compaction and depth.

Chemical quality indicators

The list of potential soil chemical indicators attributes is very large and the final selection will depend upon the soil function and process under consideration. These attributes include: pH, salinity (electrical conductivity), organic matter content, cation-exchange capacity (CEC), plant nutrient status, concentrations of potentially toxic elements, and – possibly the most important attribute – the capacity of the soil to buffer against change.

Biological quality indicators

Biological parameters are relatively dynamic and sensitive to changes in both soil management and climate. This gives biological indicators a comparative advantage over physical or chemical parameters, so they can be used as indicators of soil quality at an early stage. Some of the parameters that could serve as such indicators are: populations of micro-, meso-, and macro-organisms, soil respiration rate, enzyme activities, rate of nutrient mineralisation, microbial biomass, and more detailed characterisation of soil organic matter fractions.

Visible quality indicators

It is often the observation of visible attributes that brings to attention the changes in soil quality and causes public awareness and, at times, alarm. But in many cases, when there is visible evidence of decline in soil quality, the process of decline may have proceeded too far, and the chance to restore quality may have already been lost. The visible attributes include evidence of erosion in the form of rills and gullies, exposure of subsoil, surface ponding of water, surface runoff, and poor plant growth (Stephen, 2002).

NRM and Soil Quality Indicators

Changes in physical quality indicators

The recent developments in soil quality research emphasise the importance of identifying key soil indicators and their threshold values in relation to specific soil functions. The potential of a soil to support crop growth is largely determined by the environment that the soil provides for root growth. Roots need air, water, nutrients and adequate space to develop and make water and nutrients accessible to plants.

Such physical attributes as bulk density, porosity, air-filled porosity, crusting, sealing, water-holding capacity and depth all determine how well roots develop. Changes in these physical soil attributes directly affect the health and productivity of crops. The influence of agricultural practices, specifically NRM interventions, on changes in some soil physical attributes/indicators are discussed in the following examples.

Bulk density

Compact soil layers with high bulk density in the soil profile impede root growth by reducing the effective soil rooting volume. Measurements of soil bulk density along with penetration resistance (interpreted with respect to water contents) are used to identify root-impeding layers in the soil profile. When the bulk density of a soil increases to a critical level, it impedes root penetration and restricts root growth and the soil volume explored by roots. For example, in many Alfisols in SAT regions, soil compaction is one of the major constraints to crop establishment and productivity (El-Swaify *et al.*, 1985). Pierce *et al.* (1983) reported critical values of bulk density for soils varying in texture. Compaction by wheeled traffic has direct and at times irreversible effects on soil structure.

A long-term experiment conducted at the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT), Patancheru, India examined the impact of improved management options on soil physical attributes (Table 3.2). It was found that management practices in Vertisol

Table 3.2. The effect of management practices on physical properties of Vertisols at ICRISAT, Patancheru, India (1975–99).

	Improved land management technology		Traditional technology
Soil properties	Broadbed	Furrow	Flat
1. Texture (0–10 cm soil layer)			
Clay (%)	50.8		46.3
Silt (%)	21.5		21.4
Fine sand (%)	15.5		15.4
Coarse sand (%)	12.2		16.9
Gravel (%)	4.8		14.5
2. Bulk density (g/cm^3)	1.2	1.5	1.5
3. Total porosity (%)	52.1	39.5	41.5
4. Air-filled porosity (%)	41.0	33.0	32.0
5. Penetration resistance (MPa)	1.1	9.8	8.5
6. Sorptivity ($mm/h^{1/2}$)	121.2	100.6	88.5
7. Cumulative infiltration in 1 h (mm)	347.2	205.7	264.7

Source: ICRISAT, experimental results

watersheds caused significant differences in soil bulk densities. Throughout the soil profile, the bulk density was found to be significantly lower in the watershed with improved technology than in the watershed where a traditional system was used (Fig. 3.1).

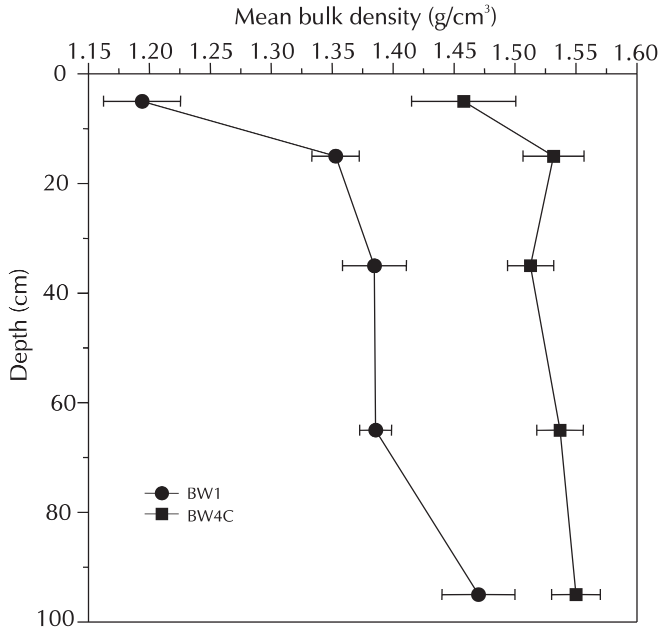

Fig. 3.1. Long-term effects of improved (BW1) and traditional (BW4C) management practices on bulk density of Vertisols, ICRISAT, Patancheru, India, 1975–98.

However, the differences in bulk densities were relatively greater in the top 15 cm layer. The maximum difference in bulk density was recorded in the 0–5 cm layer. The data clearly show the advantage of the improved technology where the soil is kept loose. This has major implications for root growth and tillage operations especially when tillage operations are done using animal power.

Penetration resistance

Penetration resistance measurement can be measured to identify root-impeding layers. When the penetration resistance of a soil increases to a critical level, it becomes more difficult for roots to penetrate and their growth is impeded. The long-term ICRISAT experiment showed that the soil in a watershed where improved land and water management was practised had a lower penetration resistance in the cropping zone than the corresponding zone in one traditionally managed (Fig. 3.2). The penetration resistance however increased with depth, but it was consistently lower with improved management. In the long term the broadbed-and-furrow (BBF) land configuration in the improved system led to progressive improvement in soil tilth in the bed zone. The BBF land configuration and reduced penetration resistance allows timely tillage operations that are crucial for Vertisols because they are difficult work, both in dry and very wet conditions. Klaij (1983) reported similar results on the positive effects of land surface treatments for Alfisols where lower penetration resistance is crucial for crop emergence and root growth.

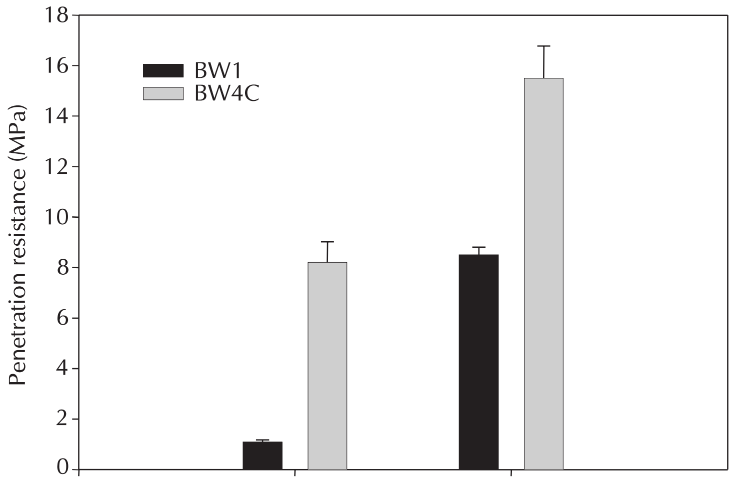

Fig. 3.2. Long-term effects of improved (BW1) and traditional (BW4C) management practices on soil penetration resistance of Vertisols, ICRISAT, Patancheru, India, 1975–98.

Porosity/air-filled porosity

The problem of temporary waterlogging and the resulting lack of adequate aeration is quite common in many soils. In medium to high rainfall areas, crops on Vertisols often suffer extensively from temporary waterlogging and poor soil aeration (El-Swaify *et al.*, 1985). In such situations, maintaining high air-filled porosity is crucial to increasing crop productivity. A long-term experiment on Vertisols at ICRISAT showed the improved system had higher air-filled porosity than the traditional system (Fig. 3.3). In the improved system, the air-filled porosity in the top 10 cm layer improved by 28% during 1975–98. This improvement contributed to better crop growth and higher yields.

Rooting depth

Rooting depth is the depth in the soil profile to which roots penetrate and access water and plant nutrients. Rooting depth is especially important in dryland agriculture where the shortage of both water and nutrients limit plant growth and productivity. Exploration of large volumes of soil by roots can increase the accessibility of water and nutrients to growing plants. Deep-rooted crops are considered better at extracting water and nutrients from

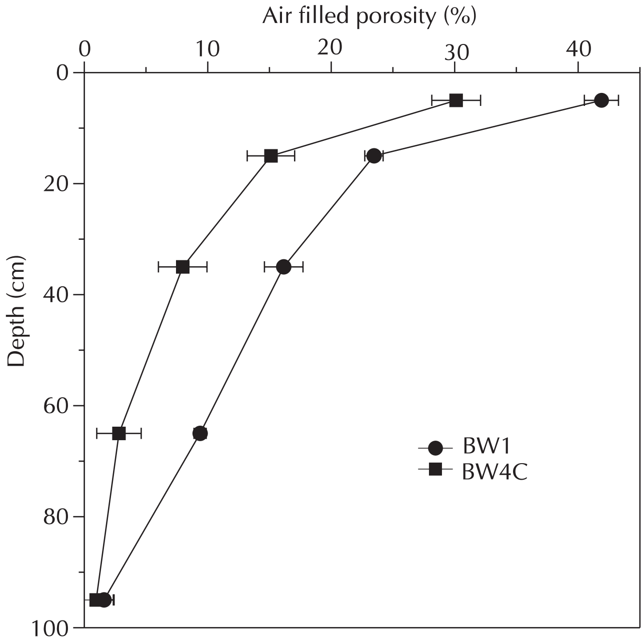

Fig. 3.3. Long-term effects of improved (BW1) and traditional (BW4C) management practices on air-filled porosity of Vertisols, ICRISAT, Patancheru, India, 1975–98.

deeper layers in the soil profile. Dryland crops such as sorghum may send their roots over 1 m deep into the soil in search of water (El-Swaify *et al.*, 1985). Irrigated crops such as rice have relatively shallow rooting depths (up to 60 cm). The threshold values for rooting depth vary with crop and the irrigated or non-irrigated conditions under which the crop is grown. The deeper rooting depths of dryland crops need to be considered while using indicators.

It is not surprising that rooting depth has been related to crop productivity. Crops grown in soils in which the rooting depth is limited by the presence of a physical or chemical constraint are generally less productive. As limiting layers move closer to the soil surface where erosion removes the topsoil, crop productivity generally declines. The effect of rooting depth on crop productivity varies with crop type (Taylor and Terrell, 1982). Soil management practices can have important effects on rooting depth. For example, erosion reduces rooting depth by removing the top soil layer while compaction reduces it by bringing to the surface layers in the soil that are impenetrable by crop roots (National Research Council, 1993). Changes in soil management practices influence root mass and length in the soil that are indicative of changes in rooting depth and can be monitored by sampling the roots at various depths in the profile.

Water-holding capacity

An important attribute of a soil is its ability to store and release water to growing plants. The water-holding capacity of soils is measured as the total amount of water stored in the different soil layers of a given profile. Plant-available water capacities of soils are required as inputs for nearly all crop simulation models. Water-holding capacity is directly related to soil structure and texture. The rate and direction of water flow through the soil is an important factor determining the effect of farming practices on soil quality (Sahrawat *et al.*, Chapter 4, this volume).

Management of the soil can have significant effects on its water-holding capacity by changing the depth and texture of surface layers (through soil erosion), the structure and compactness of surface and subsurface layers, and by affecting the rate of infiltration of rainfall. Ritchie (1981) discussed the importance of water available to plants and the techniques for measuring plant-available water in soils. Plant-available water capacities are determined at the depth of rooting considering temporal changes in plant-available water capacities during the growing season. The water-holding capacity of soil is estimated by the difference in water content at field capacity and wilting point of soil. Both these parameters can be measured in the laboratory or field using methods described by Singh and Vittal (1997).

Soil loss

Soil erosion has an overriding influence on soil characteristics that determine soil quality for productivity and environment-related functions. Eroded sediments usually contain higher amounts of plant nutrients than do bulk soils, thus soil erosion depletes the soil of nitrogen (N), phosphorus (P), potassium (K), and total organic carbon (C) reserves (Barrows and Kilmer, 1963; Young *et al.*, 1985; Lal, 2004). Erosion can also bring subsoil horizons closer to the surface of the soil profile. These horizons might have different pH, low available water-holding capacities, and high bulk densities and can thus influence soil quality. For example using the productivity index model, Pierce *et al.* (1983) and Larson *et al.* (1985) determined which of the four soil attributes in the subsoil – available water-holding capacity, bulk density, pH, or rooting depth – would cause the greatest decline in soil productivity on 75 major soils of the Corn Belt of the USA, assuming that erosion removed 50 cm (20 inches) of soil from the surface. Of the 75 soils tested, the productivity index decreased significantly in 37. This was associated with a significant degradation in the available water-holding capacity in the subsoil (13 soils), increased bulk density (4 soils), decreased rooting depth (7 soils), and increased bulk density combined with decreased rooting depth (13 soils) (Larson *et al.*, 1985).

Soil erosion removes organic carbon along with sediments. Since organic carbon content is an important indicator of soil quality, it is suggested that current rates of erosion may have significant effects on long-term soil quality (National Research Council, 1993).

Soil loss can be measured using suitable sediment samplers (Pathak *et al.*, 2002). Soil loss is commonly estimated using equations such as the Universal Soil Loss Equation (USLE) or the Water Erosion Prediction Project (WEPP) model. These models require data on soil properties, slope, erosion control practices in use, vegetative cover, rainfall and other climatic parameters.

The effects of management practices on soil loss can also be measured using field experiments. For example, El-Swaify *et al.* (1985) reported a long-term watershed experiment on Vertisols at ICRISAT, Patancheru, soil loss from erosion from watersheds under improved and traditional systems (Table 3.3).

The annual soil loss gives a good indication of the long-term effect of soil erosion on the productive capacity of soils. It is also useful in determining off-site sediment damages and the effectiveness of conservation technologies. The changes in soil physical quality indicators reported in Table 3.2 took place only in the long term. They might have been partly due to a differential loss of soil under improved and traditional management practices (Table 3.3).

Table 3.3. The effect of management practices on runoff and soil loss in watersheds at ICRISAT, Patancheru, India 1974–82.

	Improved			Traditional		
Year	Rainfall (mm)	Runoff (% of seasonal rainfall)	Soil loss (t/ha)	Rainfall (mm)	Runoff (% of seasonal rainfall)	Soil loss (t/ha)
1974	811	14.3	1.30	811	27.5	6.60
1975	1041	15.6	1.39	1055	24.0	5.21
1976	687	10.6	0.98	710	33.3	9.20
1977	585	0.2	0.07	586	9.0	1.68
1978	1125	24.3	2.93	1117	36.7	9.69
1979	690	10.6	0.70	682	29.6	9.47
1980	730	15.9	0.97	688	24.1	4.58
1981	1126	29.5	5.04	1126	38.6	11.01
1982	615	1.6	0.20	615	3.3	0.70
Mean	823	13.6	**1.51**	821	25.1	6.46

Source: Adapted from El-Swaify *et al.*, 1985

Changes in chemical quality indicators

The objective of using appropriate chemical indicators is to sustain agricultural productivity without adversely affecting soil quality. Chemical quality indicators used to monitor soil quality include organic matter, cation exchange capacity (CEC), soil acidity and exchangeable bases, soil salinity and sodicity, total and available P, total exchangeable and non-exchangeable K, total and available sulphur (S) and soil reserves of total and available micronutrients (Table 3.1).

In the following section, examples are given of the use of some chemical indicators for monitoring soil quality in crop production systems.

Organic carbon

Organic matter is an important component of soil and consists of organic C and total N. Generally, organic C constitutes 58% of soil organic matter, and is used as a measure of when to convert organic C to organic matter in soils.

Organic matter plays a critical role in maintaining physical, chemical and biological integrity of soils. Total organic C is measured using wet digestion or combustion methods in the laboratory. The dynamics of soil organic matter are controlled by management practices and agroclimatic factors, especially rainfall, temperature and soil-water regime. The maintenance of organic matter status in soils, especially in arable production systems in tropical regions, is a challenge. In contrast, it is relatively easy to maintain organic matter levels in wetland rice soils because compared to arable systems, organic matter preferentially accumulates in soils under wetland paddy culture (Jenny and Raychaudhari, 1960; Sahrawat, 2004). Although the decomposition of organic matter is fast in tropical conditions, the primary productivity of wetlands is much higher than that of arable soils; this, combined with several other factors, results in a preferential accumulation of organic matter in wetlands (see Sahrawat, 2004).

Soil organic C influences the physical, chemical and biological characteristics of soil which directly or indirectly influence crop productivity. Compared to soils in the temperate or humid tropics, soils in SAT regions have relatively low contents of soil organic matter. The traditional farming practices followed by farmers in the dryland areas do not maintain sufficient soil organic matter content (El-Swaify *et al.*, 1985). The changes in soil organic C that can be measured accurately take a long time to occur, and depend on the determination methods used and their precision. However, the changes in soil organic C can be an important soil quality indicator for evaluating the impact of management practices in both agricultural and forest lands.

The long-term effects of improved and traditional management on soil chemical and biological properties of Vertisols are shown in Table 3.4. Soil organic C, total N and available N, P, and K, microbial biomass C and N were higher in the improved than in the traditional system (Wani *et al.*, 2003).

Total nitrogen

Like organic matter, total N is an important indicator for soil chemical quality. Total N with organic C constitutes soil organic matter. Total N consists of organic and inorganic N; organic N is the source of N supply to growing plants. Total soil N is commonly measured in the laboratory using digestion but combustion methods can also be used for its determination.

A long-term (1985–97) experiment on a Vertisol at ICRISAT studied the effects of introducing different legumes into cropping systems and their rotation to improve system productivity through the supply of N by legumes. The total soil N concentration in the 0–15 cm layer increased by 125 μg N/g of soil in 12 years in pigeonpea-based systems that had no input of N (Rego and Rao, 2000). In the traditional (rainy-season fallow, postrainy-season sorghum) and non-legume based system, the total soil N declined compared to the baseline.

Cereal-N requirements are large and an increase in N use efficiency is highly desirable, not only for economic considerations, but improved N-use efficiency also reduces chances of surface and groundwater resources

Table 3.4. Biological and chemical properties of semi-arid tropical Vertisols in 1998 after 24 years of cropping under improved and traditional systems in catchments at ICRISAT, Patancheru, India.

Properties	System	Soil depth (cm) 0–60	Soil depth (cm) 60–120	Standard error
Organic carbon (t C/ha)	Improved	27.4	19.4	0.89
	Traditional	21.4	18.1	
Soil respiration (kg C/ha)	Improved	723	342	7.8
	Traditional	260	98	
Microbial biomass C (kg C/ha)	Improved	2676	2137	48.0
	Traditional	1462	1088	
Microbial biomass N (kg N/ha)	Improved	86.4	39.2	2.3
	Traditional	42.1	25.8	
Non-microbial organic N (kg N/ha)	Improved	2569	1879	156.9
	Traditional	2218	1832	
Total N (kg N/ha)	Improved	2684	1928	156.6
	Traditional	2276	1884	
Mineral N (kg N/ha)	Improved	28.2	10.3	2.88
	Traditional	15.4	26.0	
Net N mineralisation (kg N/ha)	Improved	-3.3	-6.3	4.22
	Traditional	32.6	15.4	
Olsen P (kg P/ha)	Improved	6.1	1.6	0.36
	Traditional	1.5	1.0	

Source: Wani *et al.*, 2003

pollution with N (Sahrawat *et al.*, Chapter 4, this volume). Hence, N-use efficiency should be considered an important soil and water quality indicator for monitoring the biophysical impacts of NRM.

Changes in available soil nutrient reserves

In addition to the use of organic C and N as chemical quality indicators, several other soil attributes are used for soil quality for agricultural and environment-related functions. These include changes in CEC and total and extractable nutrient status with regard to major (N, P, and K), secondary (calcium (Ca), magnesium (Mg), and sulphur (S)) and micronutrients (iron (Fe), zinc (Zn), manganese (Mn), boron (B), and molybdenum (Mo)).

Nutrient balances in production-systems can also be effectively used to ascertain the sustainability of the systems. Soils have a nutrient reserve controlled by their inherent fertility and management. A negative balance of such nutrients as N, P and K indicates nutrient mining and non-sustainability of the production systems.

Diffuse reflectance spectroscopy

Assessments of soil attributes normally rely on laboratory data resulting from the analysis of large numbers of samples required to adequately characterise spatial variability beyond the plot scale. Methods for rapid estimation of soil properties are needed for quantitative assessment of soil quality parameters. Shepherd and Walsh (2002) developed a promising approach that estimates several soil properties simultaneously, directly from diffuse reflectance spectra in rapid non-destructive ways. The method is based on scanning air-dried samples using a portable spectrometer (0.35–2.5 μm wavelength) with an artificial light source. Soil properties are calibrated to reflectance using multivariate adaptive regression splines and screening tests are developed for various soil fertility constraints using classification trees. At random, one-third of the soil samples are used for validation purposes (using standard and the proposed methods). Using this technique from about 3000 African soils belonging to nine orders, Shepherd and Walsh found that the soil attributes could be calibrated directly to soil reflectance spectra with validation R^2 values ranging from 0.70 to 0.88, indicating good agreement between the values obtained by their proposed method and standard laboratory methods. The soil attributes calibrated included: exchangeable Ca; effective cation-exchange capacity (ECEC); exchangeable Mg; organic C concentration; clay content; sand content and soil pH.

The spectral technique provides a tool for generating results of soil assessments that are conducted at a limited number of sites and thereby increase the efficiency of expensive and time-consuming soil-related studies. The rapid nature of the measurement allows soil variability to be more adequately sampled than by the conventional approach.

The spectral library approach of Shepherd and Walsh (2002) provides a coherent framework for linking soil information with remote sensing information for improved spatial prediction of soil functional capacity for agricultural, environmental, and engineering applications. Indeed, as shown below, Sanchez *et al.* (2003) found this approach useful for fertility capability classification (FCC) when assessing soil quality.

Changes in biological quality indicators

The dynamic nature of soil microorganisms makes them sensitive indicators of the soil processes leading to changes in soil quality. Biological indicators based on microbial composition, number and processes provide advanced indication of subtle changes in soil quality. However, changes in soil physical and chemical properties alter the soil environment that supports the growth of the microbial population (Lee and Pankhurst, 1992; Stott *et al.*, 1999).

Total number of microorganisms

Total microbial counts can be used as a good indicator to assess the impact of a particular management treatment on soil biological activity. The microbial population is enumerated by microscopy. Microorganisms are extracted from soil and transferred to an optically suitable background before enumeration. Several studies have recorded increase in microbial numbers in soils soon after adding an available C source (Jenkinson and Ladd, 1981).

Soil respiration

Soil respiration is the oxidation of organic materials by soil microorganisms that generates energy for microbial growth and maintenance, and produces carbon dioxide (CO_2). The soil respiration rate provides a comprehensive picture of total soil biological activity. Soil respiration is measured by determining the amount of CO_2 evolved under well-defined conditions during a given time period. Soil respiration rates were found to be higher in Vertisols under an improved than in a traditionally managed system (Table 3.4).

Microbial biomass carbon (C) and nitrogen (N)

Microbial biomass C and N in soils represent a readily available source of plant nutrients. Because the decay and turnover of microbial biomass in soils is rapid it results in the release of CO_2–C and available N. Thus, measurement of microbial biomass C and N provide a dynamic indicator of soil quality which by accurate standardisation can also be used to measure the extent of soil degradation. Microbial biomass C and N are measured as the net release of C and mineral N (ammonium plus nitrate) that results from fumigation of soil samples (Jenkinson and Ladd, 1981).

Soil and water conservation, tillage, and cropping systems influence microbial biomass C and N (Table 3.4). It has been suggested that soils with a higher proportion of soil organic C as microbial biomass gain C; those with a lower proportion lose C (Anderson and Domsch, 1986).

In a long-term (24-years) experiment on Vertisols, microbial biomass C was about 10.3% of the total soil organic C in the improved system compared to only 6.4% in the traditional system. Improved Vertisols management practices resulted in higher values (10.3 vs. 6.4%) of biomass C as a proportion of soil organic C to 120-cm soil depth, indicating that with improved management these Vertisols would reach a new C-storage equilibrium. The microbial N was 2.6% of the total biomass N in the improved system and 1.6% in the traditional system (Wani *et al.*, 2003).

Potentially mineralisable nitrogen

Along with microbial biomass N, potentially mineralisable N serves as a surrogate for the 'active N fraction' for soil quality impact assessment. The measurement of potentially mineralisable N in soils is based on the net release of mineral N (ammonium plus nitrate) from soil samples incubated for a given period under well-defined moisture and temperature conditions. Cropping systems and inputs of organic matter affect potentially mineralisable N (Wani *et al.*, 1994).

Earthworm activity

Changes in earthworm populations can significantly affect soils by influencing soil structure, nutrient cycling dynamics, and soil microbial populations.

The earthworm population decreases as soil degradation increases, and this can serve as a very sensitive indicator of soil degradation (Tian *et al.*, 2000). The earthworm population can be measured by earthworm counts per soil volume (e.g. number/m^3 of soil) in the cultivated layer.

Integrated Soil Quality Indicators

Whilst there may be doubts about the efficacy of developing integrated indices of soil quality, there is a continuing demand for them, given the complex nature of the soil and the exceptionally large number of soil properties that need to be determined. At the Rodale International Conference on the Assessment and Monitoring of Soil Quality, there was a general consensus that soil quality (Rodale Institute, 1991, cited in Arshad and Martin, 2002) encompasses three broad issues:

1. The ability of the soil to enhance crop production (productivity component)
2. The ability of the soil to function in attenuation of environmental contaminants, pathogens, and off-site damage (environment component)
3. The linkage between soil quality and plant, animal and human health (health component).

It has, therefore, been suggested that any protocol designed to determine soil quality must provide an assessment of the function of soil with regard to these three issues. To do this effectively, soil quality assessment must incorporate specific performance criteria for each of the three elements listed above, and it must be structured in such a way as to allow for quantitative evaluation and unambiguous interpretation using one aggregate soil quality index (that incorporates the above three soil functions). The objective of the proposed approach is in defining a single integrated soil quality index and not to replace past research on specific indicators but to complement it by presenting a more clearly defined framework for the development of mathematical relationships driven by basic soil attributes (Doran and Parkin, 1994).

Soil quality indices

Since soil quality encompasses plant and biological productivity, environmental quality, and human and animal health, it is imperative that the soil quality indicator provides an assessment of these functions. To achieve this objective effectively, the soil quality indicator must incorporate specific performance criteria for each function. This concept gave birth to an index.

Parr *et al.* (1992) proposed a soil quality index (*SQ*) as follows:

$$SQ = f\,(SP, Q, E, H, ER, BD, FQ, MI) \tag{1}$$

where *SP* are the soil properties, *Q* the potential productivity, *E* the environmental factors, *H* the health (human/animal), *ER* the erodibility, *BD* the biological diversity, *FQ* the food quality/safety, and *MI* the management inputs.

There has been some effort to define the exact mathematical form of the generic functional form given in Equation 1. Subsequent to the Rodale Conference, many soil scientists have proposed more detailed procedures for evaluating soil quality functions by combining and integrating specific soil quality elements into soil quality indices (Doran and Parkin, 1994; Karlen and Stott, 1994). These procedures allow for weighting of various soil quality elements, depending upon the user goals, site-specific considerations and socio-economic concerns. For example, Doran and Parkin (1994) proposed the following index of soil quality as a function of six specific soil quality elements:

$$SQ = f(SQ_{E1}, SQ_{E2}, SQ_{E3}, SQ_{E4}, SQ_{E5}, SQ_{E6}) \quad (2)$$

where the specific soil quality elements (SQ_E) are defined as:
SQ_{E1} = food and fibre production
SQ_{E2} = erosivity
SQ_{E3} = groundwater quality
SQ_{E4} = surface water quality
SQ_{E5} = air quality
SQ_{E6} = food quality

The advantage of this approach is that the different functions of soil can be assessed by specific performance criteria established for each element for a given ecosystem: for example, yield goals for crop production (SQ_{E1}); limits for erosion losses (SQ_{E2}); concentration limits for chemical leaching from the rooting zone (SQ_{E3}); nutrient, chemical, and sediment loading limits to adjacent surface water systems (SQ_{E4}); production and uptake rates for trace gases that contribute to ozone (O_3) destruction or the greenhouse effect (SQ_{E5}); and nutritional composition and chemical residue of food (SQ_{E6}).

One suggestion to operationalise this aggregate index is to use a weighted simple multiplicative function:

$$SQ = (K_1 SQ_{E1})\,(K_2 SQ_{E2})\,(K_3 SQ_{E3})\,(K_4 SQ_{E4})\,(K_5 SQ_{E5})\,(K_6 SQ_{E6}) \quad (3)$$

where K_i = weighting coefficients for the different soil quality parameters.

There could be several ways to develop an aggregate index from a set of different soil quality indicators. Campbell *et al.* (2003) propose various approaches including simple additive indices, principal components methods, canonical correlations and simple radar diagrams for evaluating the performance of NRM interventions. For example, to develop a simple additive index, it is necessary to know the maximum and minimum values of each indicator. A standardised value for each indicator is then calculated using the formula: (Indicator value at time x – minimum) / (maximum – minimum). For each indicator the potential values range from 0 (least desirable) to 1 (most desirable). A composite index is then calculated as the average of the indicator values. Weights can also be added if the relative importance of the different performance indicators is known. Details of the advantages and disadvantages of the other approaches can be found in Campbell *et al.* (2003).

Although the proposed indices would seem promising since they integrate several soil attributes in a single index, there are no published reports on their practical application and evaluation in the field. If this approach is going to be useful for NRM impact assessment, further research on the different ways of developing a comprehensive indicator would need to be carried out. When multiple variables are measured to characterise soil quality, it may not be easy to reduce the various indicators into a single and meaningful index.

Fertility capability classification (FCC) approach

Sanchez *et al.* (2003) stated that the soil quality paradigm that was originally developed in the temperate region is not very suitable for the tropics. According to them, soil quality in the tropics should focus on three main concerns: food insecurity, rural poverty and ecosystem degradation. Soil quality in the tropics must be considered a component of the integrated natural resources management (INRM) framework, therefore Sanchez *et al.* (2003) suggested that based on quantitative topsoil attributes and soil taxonomy, the fertility capability soil classification (FCC) system is probably a good starting point for measuring soil quality in the tropics. To overcome certain limitations, they proposed a new FCC version 4 (Sanchez *et al.*, 2003).

The proposed system consists of two categories. The first – type/substrata – describes topsoil and subsoil texture. The second – condition modifier – consists of 17 modifiers defined to delimit specific soil conditions affecting plant growth with quantitative limits. The type/substrata types and condition constitute soil attribute modifiers in terms of their capability for plant growth. The 17 condition modifiers include: soil drought stress (dry); nutrient capital reserves; erosion risk; aluminium toxicity; major chemical limitations; P fixation; waterlogging; leaching potential; calcareous reaction; cracking clays; gravel; shallow depth; salinity; alkalinity; presence of amorphous materials; volcanic soils; high organic content; and sulphidic soils. Like other soil indices, the FCC approach can be used to evaluate and monitor soil quality for soil productivity and sustainability purposes by measuring the FCC index at regular intervals starting with baseline measurement.

There are several important issues not addressed by this new version of FCC. These include nutrient depletion, soil compaction, surface sealing, surface crusting and others related to air and water flow. The FCC-based soil index is still at an initial stage and many more details still need to be worked out before it can be used.

Models to Assess Soil Quality

The development of relationships between soil attributes and the functions of soils is a monumental task. Simulation models can be useful tools in tracking and understanding these relationships. Algorithms in existing simulation models [e.g. the Nitrate Leaching and Economic Analysis Package (NLEAP), Erosion-Productivity Impact Calculator (EPIC), Chemical, Runoff and Erosion from Agricultural Management Systems (CREAMS), and Water Erosion Prediction Project (WEPP)] may provide a useful starting point (Doran and Parkin, 1994). These models provide a predictive tool for the process such that, given what is known, if one of the parameters that affect the process changes, the associated change in a given indicator can be predicted (Arshad and Martin, 2002). Models are normally constructed using results of detailed long-term data. Because agroclimatic conditions often vary from year to year, reliable long-term data is essential to capture the historical reality and predict future events with some degree of confidence. By using soil process models, the rate of change and the direction of change in selected soil quality indicators can be predicted. Models allow the researcher to simulate various management practices in order to predict their consequences and impacts on biophysical soil conditions and on such economic outputs as grain yield. Wani *et al.* (Chapter 5, this volume) discuss the use of simulation modelling to predict the likely impacts of NRM on various soil quality indicators.

One of the major limitations in using these models is that most of them require calibration and testing before they can be used in a given region. To the extent that the impacts of NRM interventions tend to be location-specific, lack of data from a given location can become a major limiting factor in validating the models to local conditions.

Summary and Conclusions

The intensification of agricultural activities to meet the increasing demands from fast-growing populations, particularly in the developing countries, without sufficient investments to sustain the system has led to rapid soil degradation. There is also increasing conflict among the various agricultural and environmental functions of soils. Various NRM interventions have been designed to counter the process of degradation or to enhance the sustainability of the system. In order to enhance the effectiveness of these interventions, and to attain the desired objectives, suitable indicators are required to monitor the biophysical impacts of management practices on soil conditions. This calls for the development of threshold levels for the various indicators, as these values are likely to vary by ecoregion and soil type.

In practical terms, it is not feasible to recommend the use of a large number or a common set of indicators for all agricultural interventions because of the varying size and complexity of agricultural and watershed development projects. Therefore, the selection of a few relevant indicators based on the purpose and an adequate understanding of various processes at the local level is extremely important. However, there is a general consensus that any assessment of soil quality must include a minimum set of physical, chemical and biological soil parameters. In this context the importance of a baseline characterisation of soils and sites to measure the changes attributable to a given management intervention cannot be overemphasised.

Review of the available literature and empirical examples has indicated that in general, biological indicators, followed by chemical and physical indicators could be successfully used to monitor the impact of soil management options. Changes in such physical indicators as texture, infiltration, moisture holding capacity, bulk density, porosity, aggregate stability, surface crusting and sealing, soil compaction and penetration resistance take considerable time. However, depending on the magnitude of the change, simple physical measurements such as runoff and soil loss can serve as supplementary indicators of changes in soil quality. Such soil chemical indicators as pH, salinity, organic C, organic matter content, CEC, status of plant nutrients, and concentration of potentially toxic elements can also provide good indicators. Changes in soil organic C, CEC, or soil pH or the build up of toxic elements require a long time span and cannot be monitored during a short NRM intervention. Amongst the biological indicators soil respiration, microbial biomass C and N mineralisation are commonly used to monitor changes in soil quality.

Recently, Lal (2004) reviewed the progress made in identifying soil quality indicators, especially those that are relevant to the developing countries. The key soil quality indicators listed in Table 3.5 have been proposed and evaluated for universally monitoring soil quality, although the rate of change in these parameters and threshold values varies between soils in tropical (mostly developing) and temperate (mostly industrialised) countries (Lal, 2004). Obviously, since the purpose of NRM interventions is to enhance or sustain productivity, there is a need to relate the soil quality indicators to agricultural productivity and sustainability indicators.

Table 3.5. Recent developments in identifying physical, chemical and biological indicators of soil quality.

Indicator	Associated soil characteristics/properties
Minimum data set	Aggregate stability, clay content, bulk density, soil organic C content, pH, total N, available P, S, micronutrients, mineralisable N and microbial biomass C and N
Soil N	N use efficiency and INM (integrated nutrient management)
Soil P	Environmental threshold levels of soil P
Soil K	Threshold K values, positive K balance
Cations and acidity	Critical pH and cations (K, Ca, Mg and Na), Al and Mn toxicity
Soil organic matter status	Key indicator of soil quality and environment moderation
Subsoil compaction	Soil strength
Soil structure	Critical values of soil organic C concentration
Erosion	Soil organic C, effective rooting depth, available water capacity
Soil biological quality	Microbial biomass and activity, earthworm and termite biomass

Source: Adapted from Lal (2004)

In the tropics lack of sufficient data often impedes a full assessment of the overall impact of NRM interventions on soil quality. Recently the use of an FCC, based on topsoil quantitative attributes and soil taxonomy, has been proposed. Version 4 of the FCC provides an alternative to qualitative approaches for assessing soil quality. Moreover, the development of such new tools as reflectance spectroscopy to predict soil functional attributes provides techniques for rapid measurement of soil characteristics. Simulation modelling and geographic information systems (GIS) can be useful tools for assessing the impact of NRM interventions on soil quality and the trade-offs between returns and environmental quality, especially when long-term and costly experimentation is not feasible.

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4 Measurable Biophysical Indicators for Impact Assessment: Changes in Water Availability and Quality

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Introduction

It is estimated that 94% of global water is in oceans and seas and that freshwater accounts for a mere 6% of the total volume. Freshwater is a scarce resource in many regions of the world, particularly in arid and semi-arid areas and during dry seasons in many regions that may otherwise have a surplus during wet seasons. Global freshwater availability is not a limiting factor but it is increasingly becoming a development constraint in regions with low rainfall, and in places where it is not easily accessible for human use. Thus, maintaining high quality freshwater resources is important to human, domestic livestock, and wildlife health (van der Leeden *et al.*, 1990).

Increased population and demand for food, floriculture, livestock, feed and fibre production is leading to over exploitation of freshwater in areas with limited renewable supplies. It is estimated that irrigation accounts for about 72% of global and 90% of developing-country water withdrawal (Cai and Rosegrant, 2003). In the dry areas (e.g. in West Asia and North Africa), agricultural use accounts for about 80% of the total consumption of water (Oweis and Hachum, 2003). Population growth is also leading to increased demand for freshwater for other competing uses such as domestic, agricultural, industrial and recreational activities. Agricultural activities could have adverse effects on both the quantity and quality of surface and groundwaters. Excessive and over-exploitation of groundwater is resulting in the depletion of water resources. Groundwater resources are heavily exploited for agriculture, particularly where they provide cheap water supplies that do not require large capital investments and/or do not incur high pumping costs.

The adverse effects of agricultural activities on surface and groundwater quality occur in both extensive and intensive agricultural production systems. In extensive agricultural systems, the quality of surface and groundwater is affected by the soil erosion associated with inappropriate management and over-exploitation of soil resources. Adverse effects on water quality can also occur when shifting cultivation or subsistence agriculture are practised on marginal or fragile lands, or on lands in ecologically sensitive regions. In the early phases of extensive agriculture, the use of chemical fertilisers was low and fallow periods were long, allowing soil fertility to recuperate. Such agricultural production systems also allowed soil to be conserved, and maintained its physical, chemical and biological integrity. Hence the effects on water quality were limited. Under intensive production systems, water resources become contaminated due to the increased intensity of fertiliser and pesticide use. The intensification of agricultural production systems based on high inputs of chemicals, especially in environmentally sensitive regions dominated by light-textured soils such as the porous soils of the Punjab in India, has led to nitrate contamination of surface and groundwater resources (Bajwa *et al.,* 1993).

Natural resource management (NRM) interventions can have substantial impacts on agricultural productivity and system sustainability. Similarly, agricultural and NRM practices can greatly impact water availability and quality. Assessing the impacts of agricultural and NRM interventions on water quantity and quality requires the development of appropriate indicators for measuring and monitoring such effects.

In this chapter the impact of agricultural and NRM practices on water quantity and quality are examined. The various biophysical indicators proposed to assess surface and groundwater quantity and quality impacts of agricultural and NRM interventions are discussed with examples drawn from recent literature and case studies from watersheds in the semi-arid tropics. Future research needs for developing more effective and measurable indicators of water quantity and quality for the purpose of monitoring the biophysical impacts of technological and resource management interventions are highlighted.

Agricultural Practices and Water Quantity

Water availability indicators

The water available for agricultural production includes soil moisture or water stored in the soil profile, surface water, and groundwater. Water stored in the soil profile is a function of rainfall quantity and intensity and its distribution, the storage capacity of the soil, bedrock contact, and water infiltration as influenced by ground slope and soil surface configuration and cover conditions. The available water in a watershed can be manipulated through harvesting excess rainwater and by directing the harvested water to storage in water tanks for future use.

NRM interventions can have impacts on water stored in the soil profile. For example, long-term experiments by the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT) on Vertisols and Vertic Inceptisols on a watershed scale in India showed that a broadbed-and-furrow (BBF) land configuration compared to flat land treatment on average stored 40–50 mm more water in the soil profile and reduced runoff (from 45 to 25% of rainfall), soil loss (from 6.5 to 1.5 t/ha) and nitrate-N loss (from 15 to 10 kg/ha) (Singh *et al.*, 1999; Wani *et al.*, 2002, 2003). Similar results were also reported by Srivastava and Jangawad (1988) and Gupta and Sharma (1994) who showed that the BBF landform system compared to a flat land configuration reduced water runoff, soil loss and nitrate loss in runoff water during the rainy season on Vertisols and associated soils. Recent research on a watershed (500–1000 ha) scale in India has also shown that NRM interventions (the use of improved varieties along with soil fertility management and soil and conservation practices) reduced soil loss and increased groundwater recharge and storage in surface tanks (Wani *et al.*, 2002).

Various indicators can be used to monitor the changes in water availability that result from NRM interventions. The indicators commonly used to characterise surface and groundwater availability are summarised in Table 4.1. The indicators cover soil moisture, surface water flow, surface water availability and groundwater availability; each of them is discussed in the following sections.

Table 4.1. Selected indicators commonly used to characterise water availability.

Impact outcome	Indicator used	How measured
Soil moisture	Total water in soil profile Plant available water	Gravimetric method Moisture meters (neutron probes) Pressure membrane method
Surface water flow	Runoff volume	Stage level runoff recorder with hydraulic structure
Surface water	Number of water storage structures and their capacities	Through surveys and topographic maps
	Water levels in storage structures	Staff gauge readings Remote sensing
Groundwater	Water levels in open wells Water levels in tube wells and piezometers	Water level recorders' readings at regular intervals
	Water recovery rate after the pumping	Time in h or days to recover the water level
	Duration of water pumping	Pumping time in h or days

Indicators for available surface water

Available surface water constitutes water stored in water storage structures (introduced as part of an NRM intervention) such as tanks, check dams, ponds and streams. The indicators used to measure changes in surface water quantity on a watershed scale are based on the estimation of water available from tanks, check dams and streams together with their utilisation and seasonal and long-term trends (El-Ashry, 1991; Rao *et al.*, 1996). These indicators are, however, difficult to measure. To assess surface water quantity, it may therefore be useful to consider the use of such proxy indicators as:

- Total area irrigated from surface storage structures or reservoirs
- Number of reservoirs of different capacities
- Number of reservoirs that contain water at the middle and end of the cropping season
- Number and/or length of perennial rivers
- Duration of flows for ephemeral rivers.

The data required to measure the total available surface water in a watershed include the total water storage capacity of all water storage structures in the watershed, weekly or monthly observations on the quantity of available surface water, and its use. Long-term measurements are essential to develop trends of water availability that in turn are critical for the development of accurate surface water availability indicators (Hazell *et al.*, 2001).

Indicators for surface water outflow (runoff)

Surface water outflow (runoff) as an indicator is used to measure the extent of water outflow through runoff from a given hydrological unit (e.g. a watershed). The three runoff indicators commonly used are runoff depth, runoff volume, and peak runoff rate. They indicate runoff in terms of runoff water depth, runoff water volume, and the peak runoff water rate during a given rainfall event or averaged over the entire season. These indicators are useful in determining the effectiveness of various measures and/or watershed technologies in conserving water in a watershed (Farroukhi, 1995). The surface water outflow indicator provides a useful signal of the general quality of watershed management. Equally important, the three runoff indicators can also be used to assess the long-term effects of watershed management technologies on watershed hydrology (Pathak *et al.,* 2004). The loss of soil through soil erosion that has implications for short- and long-term agricultural productivity is also directly related to this measure of surface water loss.

Water runoff can be directly measured using a suitable runoff recorder (Pathak *et al.*, 2002), or by using runoff simulation models that incorporate data on soil, slope, vegetative cover, rainfall and other climatic parameters (Littleboy *et al.*, 1989; Pathak *et al.*, 1989; Rose, 2002). For example, in India in the Adarsha watershed, Kothapally, Andhra Pradesh, and Lalatora watershed, Madhya Pradesh, where ICRISAT is conducting on-farm trials for integrated community-based watershed management, runoff was used as an indicator to assess the impact of watershed management interventions in reducing water losses. The runoffs from treated and untreated sub-watersheds were measured and compared using digital runoff recorders. The results showed a significant reduction in runoff from the treated sub-watershed compared to that from the untreated sub-watershed. Results also showed that the peak runoff rates in treated and untreated watershed were similar, suggesting that the runoff volume is the main variable that changes between treated and untreated watersheds. During the 2000 rainy season, during which higher than the average rainfall was received, the runoff in the treated sub-watershed of Adarsha was 45% lower than that in the untreated sub-watershed. The same was true for Lalatora watershed in 1999. Even during years of low rainfall, the runoff in treated sub-watersheds was about 30% lower than that observed in the untreated counterpart. Results also showed that the peak runoff rates in treated and untreated watersheds were similar, suggesting that runoff volume is the main variable that changes with treatment (Table 4.2). These empirical results demonstrate how NRM interventions affect water availability and surface water flow. The difference in selected indicators between the two management regimes can be used to measure the impact of the new technologies on surface water flow.

Table 4.2. The impact of watershed management interventions on runoff and peak runoff rate at Kothapally and Lalatora watersheds (1999–2001) (ICRISAT, unpublished).

Location/ Year	Rainfall (mm)	Runoff[a] (mm)		Peak runoff rate (m^3/second per ha)	
		Untreated	Treated	Untreated	Treated
Kothapally					
1999	584	16	NR[b]	0.013	NR
2000	1161	118	65	0.235	0.230
2001	612	31	22	0.022	0.027
Lalatora					
1999	1203	296	224	0.218	0.065
2000	932	234	NR	0.019	NR
2001	1002	290	55	0.040	0.027

[a]Untreated = control, with no development work; treated = with improved soil, water, and crop management technologies.
[b]NR = not recorded.

Runoff depth, volume and peak runoff rate indictors are useful in measuring the effectiveness of improved soil and water conservation and other NRM technologies (Samra, 1998) and to determine whether or not additional interventions in the upstream parts of watersheds are needed. Such runoff indicators can be easily measured using recorders installed in a watershed. Pathak *et al.* (2002) used data on seasonal runoff and peak runoff rates to measure runoff from treated (with water harvesting structures) and untreated (without land treatment) sub-watersheds in Madhya Pradesh. The empirical results from runoff hydrograph measurements are shown in Fig. 4.1. For a period of 10 days (5–14 September 1999), the runoff from the

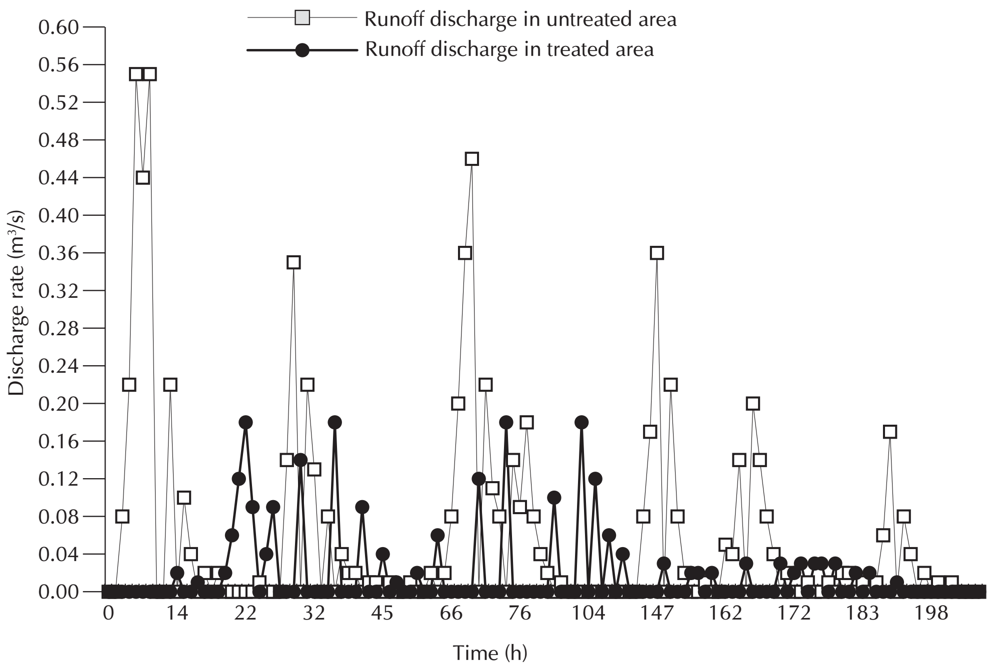

Fig. 4.1. The impact of integrated watershed management interventions on runoff as measured by a runoff hydrograph from untreated and treated sub-watersheds at Lalatora watershed, India, during 5–14 September 1999 (Pathak *et al.,* 2002).

treated sub-watershed was 130 mm compared to 150 mm in the untreated counterpart. Clearly, during the period under investigation the runoff discharge rate in the treated sub-watershed was lower than that in the untreated watershed. The majority of farmers from the treated sub-watershed reported that seasonal flooding (both frequency and the area affected by the floods) have significantly reduced. Their perception is that the construction of large check dams and other water-harvesting structures has helped to reduce flash floods. These results were influenced by the size of the sub-watersheds. This approach is designed for watersheds on a 500–1000 ha scale. However, results from this study show that treatment effects on water discharge rates are dynamic, even though they do not indicate whether the effects are sustainable.

Indicators for upstream and downstream temporary flood frequency and area affected

Flooding is caused by several factors. *In situ* flooding is caused by high rainfall on ground with low slope and soils with low infiltration (Vertisols) or with an impermeable layer (Planosols). Flooding in plains, known as induced waterlogging, is caused when a river bursts its banks or by flood irrigation. Main flooding indicators include the area affected, frequency, and duration of flooding; these indicators are important for decision-making and for assessing the impacts from NRM interventions.

Flooding indicators are used to characterise and measure the extent to which temporary or seasonal flooding upstream affects downstream parts (reaches) of streams and their tributaries. Apart from the human miseries and loss of property, seasonal flooding causes destruction of standing crops and loss of agricultural productivity, silting of lands in the course of rivers, and waste of rainwater (McCracken, 1990; Wasson, 2003). Temporary flooding or waterlogging is of major concern because it results in decreased crop productivity and/or complete destruction of crops and excess sedimentation (McCracken, 1990). For example, Vertisols in medium to high rainfall areas are very prone to severe damage as a result of temporary or seasonal flooding, particularly in downstream areas. This is mainly due to the low water infiltration rates associated with their high clay content and shrink–swell characteristics.

Data requirements for flood indicators include upstream, middle and downstream flood frequency records and estimates of damage, the extent to which land and water management practices are implemented, the number of water storage structures in a given area, and the implementation of other vegetative control measures (Sharma *et al.*, 1991). For large watersheds, aerial photographs taken during periods of temporary flooding and the use of other types of periodic remote-sensing tools are useful. These can be complemented by interviews with local farmers to assess short-term flood frequency and damage (Rao *et al.*, 1993). For small- and medium-sized watersheds (500–1000 ha), the peak runoff rate and total runoff volume can be used as indicators of temporary flooding and the area affected by such flooding (Pathak *et al.*, 2004).

Indicators for groundwater availability

The part of rainfall water that percolates deep into the ground strata, beyond shallow depth (due to a perched water-table), becomes part of groundwater. It is essential that rainfall recharges groundwater to a desirable level each season to ensure the sustained maintenance of available groundwater. Groundwater levels in many areas are declining despite the implementation of several measures to improve groundwater recharge because of excessive withdrawal of water (Moore, 1984; Khepar *et al.*, 2001). However, NRM interventions can be used to improve groundwater levels by changing the level of recharge. For example, this problem can be addressed by reducing runoff water through bunding and by increasing the percolation of rainwater to recharge the groundwater-table through check dams, percolation tanks, ponds and other water-harvesting and soil-conservation structures. However, in most locations off-take of water for irrigation and domestic use is increasing, resulting in a ‘smaller than desired’ effect of interventions on the groundwater-table. This trend has become more important over time despite the implementation of various practices to harvest, conserve and use rainwater.

Indicators of groundwater availability include depth of groundwater, safe yield (sustainable level of harvest), number of wells, spatial and temporal availability, and yield. To increase land productivity it is important that the use of available groundwater in a given hydrological unit is optimised. For the sustainable management of groundwater resources, it is necessary to have information on how much water can be stored, and how much can be taken off for irrigation and domestic use. The potential or permissible withdrawal of water is a function of groundwater recharge that in turn is a function of rainfall, runoff, evapotranspiration, percolation, and geological thresholds. The concept of safe yield needs to be evaluated on a watershed scale so that there is a balance between groundwater recharge and outflow (including pumping). To put the concept of safe yield into practice, the total numbers of open wells, tubewells and their depths and spacing need to be estimated and monitored for water status.

The depth of groundwater in wells is the most widely used parameter by researchers, development agencies and farmers for estimating the level and availability of groundwater (Moore, 1984; Khepar *et al.*, 2001; Wani *et al.*, 2003). But, several development agencies also use the number of operating or dry wells, and the area under irrigation as indicators of the water-table and quantity of available groundwater (Rao *et al.*, 1996).

Groundwater level measurements are often used as indicators to assess the impact of various soil and water conservation interventions on groundwater status. For example, in Adarsha watershed, Ranga Reddy district, Andhra Pradesh, ICRISAT monitored the water level in 62 open wells situated at different distances from water recharging facilities at fortnightly intervals. The results showed that after the construction of check dams and other soil and water conservation structures, the water level and yield in the open wells during the study period (1999–2002) improved significantly, particularly in open wells located near water-harvesting structures. The differences in groundwater levels in open wells near or away from check dams were relatively smaller during years of relatively low rainfall, but this difference grew during years of high rainfall, indicating the positive contribution of water-harvesting and recharging structures to increasing groundwater levels. This indicator showed a consistent pattern in groundwater levels during relatively low (1999, 2001 and 2002) and high rainfall (2000) years (Fig. 4.2). The effect of seasonal rainfall on groundwater levels in treated and untreated sub-watersheds is shown in Fig. 4.3. The groundwater level measured in the treated sub-watershed was higher than that in the untreated sub-watershed, where it fell steeply during low rainfall years. However, despite increased water withdrawal as farmers drilled more wells in the area, the treated sub-watershed maintained a higher groundwater level during the 2000–2002 seasons. This example shows how the selected indicator can be monitored at regular intervals to evaluate how improved catchment management contributes to increasing the availability of groundwater. The difference in groundwater levels between the two treatments can be used to estimate the impact of improved water management practices on groundwater availability.

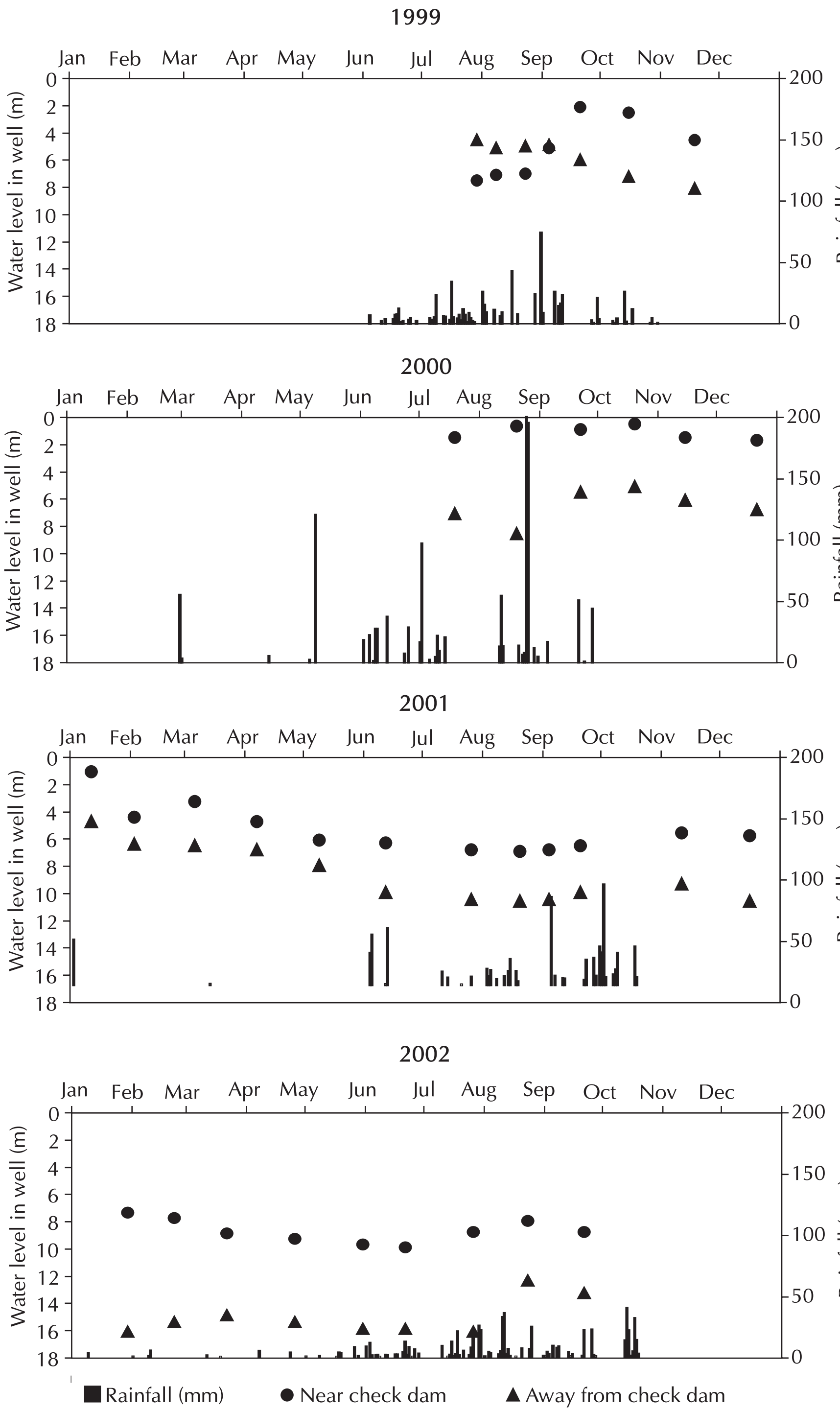

Fig. 4.2. The impact of check dam construction and soil and water conservation practices on groundwater levels at Adarsha watershed, Kothapally, India, 1999–2002 (ICRISAT, unpublished data).

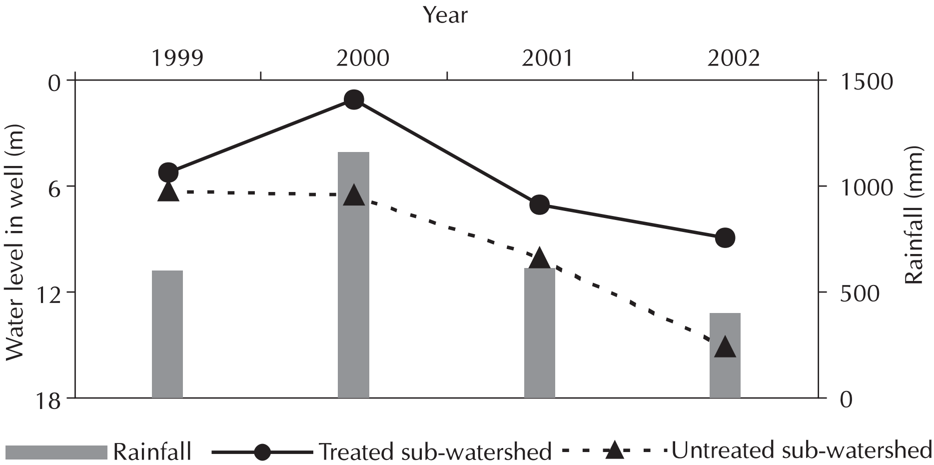

Fig. 4.3. The impact of integrated watershed management on groundwater levels at Adarsha watershed, India, 1999–2002 (ICRISAT, unpublished data).

Most of the existing groundwater indicators do not provide adequate information for planning and judicious management of groundwater resources. Moreover, simply monitoring changes in the water level in open wells or bore wells does not explain the extent to which changes in water levels are attributable to one or more of the following:

- Annual variations in rainfall and their effect on groundwater recharge and reduced runoff
- Increased off-take for irrigation resulting from increasing numbers of bore wells or deeper drilling of wells
- Increased off-take for domestic use.

The effect of variation in annual rainfall on groundwater recharge makes the relationship between annual or seasonal rainfall and groundwater levels quite complex. This requires a better understanding of the pattern of multi-annual fluctuations in the water-table and its relationship with variation in rainfall (Hazell *et al.*, 2001).

There is a clear need for more appropriate indicators of groundwater availability that can provide accurate information about its status. Such indicators need to provide enhanced information for management and planning, and adequate signals for tracking the long-term sustainability of groundwater resources (Farroukhi, 1995).

Recently watershed programmes have been adopting participatory methods to develop more effective indicators of groundwater availability. Farmers are being closely involved in monitoring groundwater levels and in deciding the equitable distribution of surface and groundwater. In some instances, participatory groundwater monitoring experiences in India have contributed towards the sustainable management of groundwater resources (APWELL, 2003). Preliminary survey results suggest that the participatory monitoring can be an effective way to equitably manage groundwater at the community level (Kerr, 2002). Most of the participatory groundwater monitoring research work is in the initial stages of testing. Its usefulness will depend on the outcome of such research.

Indicators for rainfall use efficiency

In this chapter, rainfall efficiency is defined as the economic yield or economic returns per millimeter of rainfall (for detailed reviews see Molden *et al.*, 2003). The underlying concept is to produce 'more crop per drop' of water or 'producing more with less water'. In addition to rainfall use efficiency as an indicator, other proposed sub-indicators include:

- The amount of water stored in the root zone divided by the total rainfall per growing season
- Crop transpiration divided by total rainfall
- Crop yield divided by total rainfall in a given growing season
- Gross margins divided by total rainfall (Barker *et al.*, 2003; Molden *et al.*, 2003).

Increasing rainfall use efficiency is crucial for rainfed farming and can be effected by the judicious use of external inputs such as fertilisers and by implementing soil and water conservation practices.

Rainfed production systems that do not use water efficiently result in irrecoverable loss of water resources, lost opportunities for higher crop yields, and the possible degradation of water quality (Samra, 1998). For example, in a water-deficit situation it is very important to use rainfall use efficiency as an indicator to assess the efficiencies of various NRM technologies. The data required to compute rainfall water use efficiency include: data on daily and annual rainfall; runoff; crop yields; evapotranspiration (measured or simulated value); outflow and inflow of surface and groundwater; and volume of water withdrawn for irrigation.

Water Quality Indicators

Water quality is generally defined by its physical, chemical, biological and aesthetic (smell or odour and appearance) characteristics. These quality parameters may differ with use (drinking, recreation, wildlife, industrial, agricultural or domestic). Like water availability, water quality is greatly influenced by NRM-based agricultural activities. Land and water management practices, tillage, and the use of fertilisers and plant protection chemicals all affect water quality. Several indicators have been proposed to characterise and monitor the physical, chemical, and biological characteristics that relate to water quality in its various uses (Table 4.3).

Water quality is high in undisturbed or natural ecosystems. Several soil processes are adversely affected by the conversion of lands under natural vegetation to agricultural production. Among these, the hydrologic cycle and cycles of carbon and plant nutrients are most relevant to the determination of water quality. The conversion of natural systems (under forest or grass) to agricultural land use reduces water quality due to the

Table 4.3. Selected water quality indicators for monitoring and impact assessment of natural resource management interventions.

Criteria	Water quality indicators
Physical/aesthetic quality	Odour Floating matter Colour Turbidity and clarity Dissolved solids Sediment load Suspended organic and inorganic materials
Chemical quality	pH (acidity/alkalinity) Salinity, electrical conductivity Dissolved oxygen Chemical oxygen demand Dissolved organic matter and organic nitrogen Dissolved load of chemical constituents (nitrate, phosphorus, fluoride, pesticides, toxic compounds, etc.) Heavy metals (copper, nickel, mercury, lead, chromium, cadmium, etc.)
Biological quality	Biomass Microorganisms Biological oxygen demand Pathogens (bacteria, algae, etc.) Phytoplankton and zooplankton Cyanobacteria

contamination of water with sediments, plant nutrients, and agricultural chemicals used in production systems. Studies in the humid tropical regions of Nigeria suggest that the quality of surface water is greatly influenced by agricultural operations (Lal, 1994). Water quality is significantly affected by land use and farming systems. The principal agricultural management practices that affect the quality of surface and groundwater include:

- Soil surface management including tillage methods and ground cover
- Crop residue management and the use of such crop residues as mulch, ploughing under, burning, or grazing
- Fertility management including type of fertiliser (inorganic or organic, soluble or slow-release), method of placement and time of application
- Crop rotations including cropping intensity, crop type, type of farming (commercial or subsistence) and use of chemicals to control insects and plant diseases
- Weed management including use of chemicals, cultivation and manual weeding (Angle *et al.*, 1984, 1993; Lal, 1994).

In general, farming practices that affect soil erosion also affect surface and groundwater quality (Lal, 1994; Evans, 1996).

The movement of sediment and associated agricultural pollutants (fertilisers, pesticides and amendments) into watercourses is the major offsite impact resulting from soil erosion. This not only results in the silting-up of dams, and disruption of wetland ecosystems, but also leads to the contamination of drinking water (Evans, 1996). It has been observed that pollution of surface and groundwater takes place even if the rate of soil erosion is not high, because significant amounts of agricultural chemicals can be transported off-site (Favis-Mortlock, 2002).

Water quality indicators associated with agricultural practices include: sediment load in runoff water, quality of runoff water, nitrogen (N) and phosphorus (P) concentrations and amounts in runoff water, and nitrate pollution of groundwater (Lal, 1994; Jones *et al.*, 1999; Thorburn *et al.*, 2003).

High levels of water pollution resulting from intensification of agriculture have negative effects on human and animal health that need to be accounted for in assessing the impact of agricultural practices and other NRM interventions. The World Health Organization guidelines for nitrate in drinking water recommended that the nitrate concentration be less than 50 mg nitrate/l or 11.3 mg nitrate-N/l (WHO, 1970). According to this recommendation, nitrate concentration in the range of 50–100 mg/l is acceptable, but a concentration of greater than 100 mg nitrate/l can be harmful. In 1980, the European Economic Community (EEC) recommended a maximum acceptable concentration of 50 mg nitrate or 11.3 mg nitrate-N/l unless waivers were granted by the member-state of the Union (EEC, 1980).

Among the plant nutrients, added N is of great concern because it is required in large amounts for crop production. Nitrogen is generally transported from soils into surface and groundwater by water runoff, erosion and leaching (mainly nitrate) (Foster *et al.*, 1982; Follett, 1989). In arable crop production systems, the nitrification of soil and fertiliser ammonium converts relatively immobile ammonium into highly mobile nitrate. That explains why the control or regulation of nitrification retards the contamination of surface and groundwater with nitrate by reducing the movement of nitrate in runoff water and through leaching (Sahrawat, 1989).

Singh and Sekhon (1976) studied the nitrate pollution of groundwater from N fertilisers and animal wastes on light-textured soils in Punjab where N fertilisers are intensively used to grow such cereal crops as maize and wheat. They found that in the Ludhiana district, 90% of the well water samples contained less than 10 mg/l nitrate-N. More importantly, the nitrate concentration of well water decreased significantly with depth, and correlated positively with the amount of fertiliser N added annually per unit area.

Monitoring the nitrate-N concentrations in shallow well water in Ludhiana in 1982 and 1988 revealed that the increase in fertiliser N consumption was associated with an increase of nitrate-N of almost 2 mg/l (Singh *et al.*, 1991). Bajwa *et al.* (1993) analysed 236 water samples from 21 to 38 m deep tube wells in different blocks of the Punjab where annual fertiliser-N consumption ranged from 151 to 249 kg N/ha. They found that 17% of the tube-wells in vegetable-growing areas contained more than 5 mg NO_3-N/l compared to 3% in the tube-wells located in rice–wheat and 6% in potato–wheat rotation areas. These results suggest that excess N not used by the crops moved to the groundwater with rainwater during the rainy season. These results drew attention to the need for rational use of fertiliser N to avoid nitrate pollution of surface and groundwater in porous soils.

Soil conservation practices such as landform configuration also help to conserve soil and reduce loss of N in runoff. For example, a study on Vertic Inceptisol at the ICRISAT farm in Patancheru, India (Table 4.4) showed that the BBF landform had less water runoff, soil loss and nitrate-N loss in water runoff than a flat landform during the 1998 rainy season (ICRISAT, unpublished).

Table 4.4. Impacts of improved land management (flat vs. broadbed-and-furrow (BBF)) on water runoff, soil and nitrate loss in Vertic Inceptisols, ICRISAT farm, Patancheru, India, 1998 (ICRISAT, unpublished data).

Parameter measured	Land management treatments	
	Flat	BBF
Water runoff (mm)	287	226
Soil loss (t/ha)	5.4	3.1
Nitrate-N loss (kg/ha)	13.3	9.3

Among the water quality indicators used to assess the impact of agricultural practices (Table 4.3), the most important and practical indicators of surface and groundwater quality include sediment load, odour or smell, dissolved load of chemical constituents (nitrate, P, pesticides, etc), turbidity and colour. These indicators are also simple and useful in decision-making. For example, waters with high proportions of suspended materials and foul smell are not considered suitable for domestic use, especially for drinking.

The contamination of groundwater with such chemicals as nitrate, phosphate, fluoride, basic cations (potassium, calcium, magnesium and sodium) and heavy metals (mercury, copper, nickel, lead, cadmium, chromium, etc.) is a problem. This contamination can be determined by chemical analysis of surface, shallow, or deep groundwater. Measurements of concentrations of the polluting chemical serve as quality indicators. The suitability of water for drinking, agricultural or other domestic use depends on several physical, chemical and biological properties and their acceptable concentrations or presence in the water (Lal, 1994). For example, long-term chemical analysis of rainwater samples from three locations on the ICRISAT farm showed that rainwater annually added significant amounts of N, sulphur, potassium, magnesium and calcium nutrients to the soil. This input of nutrients through rainfall offsets, at least partially, their removal by crops (Murthy *et al.*, 2000). The changes in water quality resulting from NRM interventions can also be compared to the threshold levels specified by the international water quality standards for chemical contaminants (Table 4.5).

The presence of such pathogens as bacteria, cyanobacteria and other algae or microorganisms has been found to be highly undesirable for the use of surface and groundwater for various domestic purposes. Little research has been reported on the contamination of both surface and groundwater with pesticides, but pesticide contamination of surface and groundwater is of great concern to human health.

Table 4.5. International water quality standards for some chemical constituents for human and livestock consumption (Lal, 1994).

	Concentration (mg/1000 ml)	
Chemical constituent	Human	Livestock
Nitrate	< 45	< 200
Ammonium	< 0.05	NA[a]
Chloride	< 400	< 1000
Calcium	< 200	< 1000
Barium	< 1.0	NA
Zinc	< 15	< 20
Molybdenum	NA	0.01
Lead	< 0.1	0.05
Arsenic	< 0.05	0.05
Selenium	< 0.01	0.01
Cadmium	< 0.01	0.01
Mercury	< 0.01	0.002

[a]NA = not available.

Application of Simulation Modelling

Hydrological models have been extensively used to assess surface and groundwater availability (Pathak and Laryea, 1992; Allerd and Haan, 1996; Sireesha, 2003). The models have been used to provide evidence of trends in the long-term availability of surface and groundwater. Pathak and Laryea (1992) used a water-harvesting model to estimate the probability of runoff and water availability in a tank. They also ran simulations using long-term data on rainfall, evaporation, soil characteristics and catchment area, to estimate the chances of adequate stored water being available for supplemental irrigation during drought stress periods in a growing season (Pathak and Laryea, 1992).

There is a direct link between soil conservation and the enhancement of surface and groundwater quality. This implies that without soil conservation practices water quality cannot be maintained. Research on water quality has focused on developing simulation models to evaluate suitable soil management practices that maintain surface and groundwater quality (McCool and Renard, 1990). Simulation modelling has an important role to play in the development of water quality indicators for monitoring and assessing water quality. Several water quality models (McCool and Renard, 1990; Williams *et al.*, 1994) have been used to generate information on how to solve a variety of complex water quality problems. It has been suggested that simple screening simulation models may be sufficient to identify pollution sources in surface and groundwater. On the other hand, rather comprehensive models may be required to compare the effects of various agricultural management practices on the transport of chemicals and pollutants by water runoff and sediment (Williams *et al.*, 1994).

For example, simulation models have been used to estimate the amount of nitrate-N in runoff water from the soil surface layer. The decrease in nitrate-N concentration by the volume of water flowing through a soil layer is simulated using an exponential function. In this way, an average daily concentration of nitrate-N can be obtained by integrating the exponential function to give nitrate-N yield, and dividing this value by the volume of water leaving the soil layer in runoff, lateral flow, and percolation. The amount of nitrate-N in surface runoff is estimated as the product of the volume of water and the average nitrate-N concentration. A provision is made in the model for estimating production of nitrate via nitrification and loss of ammonium via ammonia volatilisation. The loss of nitrate produced via denitrification is also taken into account under partial anaerobic or anaerobic conditions created by the water regime.

Simulation models have also been used to evaluate the impact of agricultural practices on environmental quality. For example, Kelly *et al.* (1996) simulated the long-term (30-year) impacts of different cropping systems and such NRM interventions as no-till, manure application, and cover crops on the tradeoffs between net returns and different aspects of environmental quality. Their study showed that no-till rotations provided the greatest returns, followed by conventional rotations. In terms of environmental impacts, no-till rotations dominated all other rotations with lowest N loss, and cover crop rotations had the best results in terms of soil erosion and P loss. However, since herbicides were used to control weeds in the no-till system, the pesticide index was very high, suggesting a trade-off between pesticide hazard and other environmental considerations. The authors also constructed an environmental hazard index to provide decision-makers with better information for analysing the trade-offs between potential chemical contamination of water bodies and net returns.

Recently, the combined use of geographic information systems (GIS) and mathematical modelling has been used to develop decision-support systems for quantifying:

- Runoff and movement of sediment, pesticides and nutrients
- Percolation and leaching of pesticides and nutrients to shallow groundwater
- The economic impact associated with crop management, land use, and other policy changes to improve water quality at the watershed and river basin levels (Lovejoy *et al.,* 1997).

Gardi (2001) evaluated the impact of a new agronomic framework protocol in a small watershed using combined applications of GIS and a crop-simulation model (CropSyst). It was found that the greatest leaching of nitrate occurred on coarser-textured soils. Erosion and herbicide effects on water quality were higher in sloping areas sown to spring–summer crops. It was concluded that the increase in row-crop cultivation, determined by European Union (EU) agricultural policy, represented the main adverse impact on water quality of the site studied.

Summary and Conclusions

With the impending freshwater scarcity in many regions of the world, water availability and issues relating to water quality are assuming increasing importance. Agricultural activities can affect the quantity and quality of surface and groundwater resources. Improved NRM practices are being developed and implemented to reduce the negative environmental outcomes of agricultural practices and to increase water availability and quality. Information reviewed in this chapter indicates that the use of fertilisers, especially fertiliser N in excess of that utilised by plants in intensive production systems on porous soils, has the potential to contaminate shallow and deep groundwater resources. Little information is, however, available on the contamination of surface and groundwater resources with pesticides and other agricultural chemicals. There is lack of sufficient data on biophysical indicators from tropical regions to fully assess the impact of agricultural practices and soil processes on water availability and quality.

Because of their simplicity, cost and effectiveness, commonly used water availability indicators include:

- Measurement of soil moisture using the gravimetric method
- The number of storage structures and their water levels to assess surface water availability
- Water levels in open wells, tube wells and piezometers, and duration of water pumping to determine groundwater availability.

Commonly used water quality indicators include:

- Aesthetic (smell, appearance, floating matter)
- Physical (sediment load, turbidity)
- Chemical (chemical constituents such as nitrate, fluoride, etc.) and
- Biological (presence of bacteria and pathogens, etc.) characteristics.

More importantly, unlike soil quality that takes a long time for observable changes to occur, water quality is extremely dynamic and needs regular monitoring.

Recent watershed research results reviewed in this chapter indicate that improved NRM interventions have the potential to decrease runoff and soil loss and increase surface and groundwater availability. However, there is a need to generate more empirical data on the impact of NRM technologies on water availability and the quality of surface and groundwater in different ecoregions, because these relationships are likely to be context- and location-specific.

Another important research area is understanding the relationships between soil management and water quality, especially in tropical regions where there is a shortage of such information (Karlen, 1999). When minimal empirical data is available, simulation models can be used to understand this relationship, and to provide information useful in developing indicators that consistently track impacts over time. More attention is needed to link technological options for water harvesting and use to regular monitoring of impacts on water budgets and quality of groundwater resources. In addition, threshold or tolerable limits in terms of the concentrations of major pollutants in natural waters need to be standardised.

Priority should be given to developing and applying simulation models that can effectively predict nitrate movements in surface water and its leaching into groundwater, and how this will be affected by agricultural and resource management practices. Such research can be helpful in developing ecofriendly and environmentally sound N management practices for intensive and high input-based agriculture (Moreels *et al.*, 2003).

Progress in generating information required to monitor the impacts of agricultural and management practices on water availability and quality in the developing regions has been slow and limited. The use of simulation modelling and remote sensing and GIS tools could help to bridge this gap and to develop useful decision-support systems. In addition to such biophysical factors as soils, climate, and land use, socio-economic and institutional factors and agricultural policies often play an important role in the management of water resources. Greater emphasis should therefore be given to integrated approaches that link socio-economic and biophysical information when assessing the impacts of NRM interventions on water quantity and quality (Faeth, 1993; Lal and Stewart, 1994; Shiferaw and Holden, Chapter 12, this volume).

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5 Biophysical Indicators of Agro-ecosystem Services and Methods for Monitoring the Impacts of NRM Technologies at Different Scales

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Introduction

Natural resources such as soil, water, air, and vegetation that provide the vital needs of humans and animals are in a perpetually dynamic state. Agricultural interventions typically involve opening closed natural systems that may have attained a certain equilibrium. Such products as food, feed, fuel, etc. are exported from the system resulting in more outflows than inflows. When this happens, unless outflows are complemented by external inputs, resource productivity will gradually decline. Land degradation is a commonly used term to describe this situation and refers to the productivity loss and/or diminishing ability of land to provide such essential ecological services as groundwater recharging, carbon fixation and storage, detoxification of harmful compounds, and water purification.

In order to minimise the process of degradation and to maintain productive capacity and ability to provide ecosystem services for present and future generations, various natural resource management (NRM) options have been developed and implemented.

Socio-economists and natural resource experts have long struggled to assess the broader economic and environmental impacts of NRM technologies. This has been a difficult task because such technologies are not separately developed and marketed as divisible component inputs like seeds. Typically NRM practices are developed in an integrated approach to improve biophysical conditions and are used in conjunction with other yield-enhancing inputs. Hence, the direct economic benefits derived from such technologies are not always evident and are generally attributed to such other inputs as improved seeds. The new paradigm of integrated natural resource management (INRM) aims to provide multi-disciplinary solutions in a coordinated manner to achieve livelihood and sustainability objectives. However, the full social impact of INRM cannot be measured directly using conventional methods of economic evaluation (Shiferaw *et al*., Chapter 2, this volume).

Therefore, appropriate qualitative and quantitative indicators of biophysical impact on varying spatial and time scales are needed. A good indicator must be sensitive enough to show temporal and spatial changes, predictable, measurable and interactive. Assessing NRM impacts will need new methods, tools and multidisciplinary teams of experts to understand and accurately quantify the benefits. Some non-marketed agro-ecosystem services are especially difficult to recognise and quantify. Such tools as simulation modelling, geographic information systems (GIS), and satellite imaging, used in conjunction with traditional productivity-based techniques, are vital in estimating some NRM impacts. Productivity-based indicators (e.g. biomass and crop yields) at micro levels need to be complemented by indicators like the vegetation index at ecoregional levels using satellite images and GIS tools. Simulation modelling is also useful for verifying and extrapolating results to larger scales and for studying long-term effects.

Chapters 3 and 4, this volume, dealt with biophysical indicators for assessing soil quality and water availability and quality. This chapter presents indicators used to monitor changes in the flow of such other ecosystem services as biodiversity conservation, carbon sequestration and ecosystem regulation and describes tools and methods available to monitor and estimate the impacts associated with adoption of NRM technologies on various scales. This chapter first presents the criteria and indicators for monitoring NRM impacts related to various ecological functions and ecosystem services. The use of simulation models to estimate biophysical changes is then discussed. Following is a discussion of how remote sensing and GIS tools can be used to monitor spatial and temporal changes. The key issues and areas for future research are highlighted.

Indicators of NRM Impact

An indicator is a sign or signal that relays a complex message, potentially from numerous sources, in a simplified and useful manner. It can reflect the biological, chemical or physical attributes of ecological conditions. The primary uses of an indicator are to characterise current status and to track or predict significant change. With a foundation of diagnostic research, an ecological indicator may also be used to identify major ecosystem stress. Glave and Escobal (1995) proposed a set of verifiable and replicable indicators to assess changes in natural resource conditions, the ecological and economic structure, and ecological, economic and social benefits in the Andes. Munasinghe and McNeely (1995) suggested the index of biophysical sustainability, soil and water conservation, efficiency of fertiliser use, efficiency of energy use, and the productive performance of forests as important NRM indicators. Ramakrishnan (1995) introduced such additional indicators as management practices, biodiversity and nutrient cycles. Smyth and Dumanski (1993) stated that good indicators are measurable and quantifiable, such as the environmental statistics that measure or reflect environmental status or changes in resource conditions. Agricultural systems can be analysed at various hierarchical levels. For land evaluation and farming systems analysis, FAO (1992) distinguishes between cropping, farming, sub-regional, regional, and national systems. The precision level and the purpose of a given indicator will change if it is extrapolated to a higher scale and time step.

Indicators for assessing NRM technology impacts are selected according to data availability, data sensitivity to temporal and spatial change, and the capacity of the data to quantify the behaviour of given agricultural systems. Table 5.1 presents commonly used and potential indicators for monitoring NRM impacts.

Table 5.1. Indicators for monitoring biophysical and sustainability impacts of NRM interventions.

Criteria	Indicators
1. Biodiversity	• Species richness • Species diversity • Species risk index
2. Agro-biodiversity	• Index of surface percentage of crops (ISPC) • Crop agro-biodiversity factor (CAF) • Genetic variability • Surface variability
3. Agro-ecosystem efficiency	• Productivity change • Cost–benefit ratio • Parity index
4. Environmental services	• Greenery cover/vegetation index • Carbon sequestered • Reduced emissions of greenhouse gases • Reduced 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, etc.) • Soil chemical indicators (e.g. soil pH, organic C, inorganic C, total and available N, P and other nutrients, CEC, salinity, etc.) • Soil biological indicators (e.g. soil microbial biomass, soil respiration, soil enzymes, biomass N, diversity of microbial species, etc.)
6. Water availability and quality	• Quantity of fresh surface water available • Fluctuations in groundwater level • Quality of surface water and groundwater

Biodiversity indicators

Natural resource management affects biodiversity on various scales. Indicators are required to assess the impacts of NRM interventions on natural and managed ecosystems. Biodiversity has been most generally defined as the 'full variety of life on Earth' (Takacs, 1996). It is the sum total of different kinds of diversities such as species diversity within communities, genetic diversity, i.e. the variety of individuals within populations, and life-form, floristic, and functional diversities. Some believe that it has simply replaced the terms 'nature' or 'wilderness' (Chadwick, 1993). In fact, 'biodiversity' is now sometimes used to mean 'life' or 'wilderness' and has served on occasion as a catch-all for 'conservation' itself. Biodiversity provides many benefits. Its reduction influences the structure, stability and function of ecosystems and diminishes the flow of valuable ecosystem goods and services to humans (Erlich and Erlich, 1992). Some of these benefits come in the form of goods that can be directly valued and costed while other critical indirect benefits to humans are difficult to value and quantify (Freeman *et al*., Chapter 1, this volume; Shiferaw *et al*., Chapter 2, this volume). These benefits include such ecosystem services as air and water purification, climate regulation, soil formation, and the generation of moisture and oxygen.

When exploring indicators that might shed light on the conservation of biodiversity, it is essential to identify the types of indicators needed on various scales to determine whether conservation objectives are being met. Reid *et al*. (1993) provide a summary of 22 biodiversity indicators defined on three levels: genetic, species, and community diversities.

Biodiversity on any scale can be measured with flora, fauna and species diversity of different types. The term species diversity or biodiversity at first instance means the number of different species found in a given area, but it must take into account the relative abundance of all the different species. Indicators are needed to measure the outcomes related to such effects. Changes in biodiversity can be measured in terms of indicators for species richness, diversity, and risk index. Species richness and species diversity are often confused and used interchangeably, but mean different things (Spellerberg and Fedor, 2003).

Species richness

This refers to the total number of species per site or habitat and can be estimated by counting all species within the target area (Simpson, 1949). Although species richness is a measure of the variety of species, it should be used to refer to the number of species in a given area of sample (Spellerberg and Fedor, 2003).

Species diversity

This measures the total number of species (abundance) and their relative distribution, i.e. as the index of some relationship between number of species and number of individuals. Diversity indices that take the relative abundances of different species into account, therefore provide more information about community composition than simply species richness. Species diversity can be measured in several ways; commonly used indices are the Shannon Index, the Simpson Index and the Species Risk Index.

The Shannon Index (H) is based on probabilities of occurrence. It measures the average degree of uncertainty in predicting the species of a given individual selected at random from a community (Shannon and Weaver, 1963):

$$H = -\sum_{i=1}^{K} [P_i \ln(P_i)] \quad (1)$$

where $P_i = n_i/N$ is the number of sample observations in category i, n_i is the number of individuals in category i, and N is the total number of individuals in the sample.

The index varies from a value of 0 (for communities composed of a single species) to high values (for communities with many species). The larger the index, the greater the diversity. This index, based on communication theory, is also referred to as the Shannon–Wiener Index (in recognition of the work of Norbert Wiener from which Shannon built the index) and the Shannon–Weaver Index (in recognition of the mathematician Warren Weaver with whom Shannon co-authored his original book in 1949). The index combines the number of species (species richness) with the distribution of individuals among species to provide a quantitative measure of diversity in any habitat.

The Simpson Index (SI) measures the probability that two individuals randomly selected from a sample will belong to the same species (or some category other than species) (Simpson, 1949). The index can be computed as:

$$SI = \sum_{i=1}^{K} n_i(n_i - 1)/N(N-1) \quad (2)$$

where $0 \le SI \le 1$, n_i is individuals in species i and N is sample size (total number of individuals). With this index, 0 represents infinite diversity and 1, no diversity. In order to make the index more intuitive, it has been suggested to use $1-SI$ or $1/SI$ so that diversity increases with the index.

Moreover, when it is necessary to compare the degree of similarity in the abundance of different species in a given habitat, the evenness index (EI) can be calculated using H and S (Shannon and Weaver, 1963) as:

$$EI = H/\ln(S) \quad (3)$$

where S is an index of species richness (defined above).

When there are similar proportions of all species, EI will have a value of 1. When the abundances are very dissimilar, the value of EI increases to greater than 1.

The Species Risk Index combines information on endemic species within a community and on the status of that community in order to provide insights into the risk status of species, even in the absence of detailed threatened species lists. The index is calculated by multiplying the number of endemic species (per unit area) in a community by the percentage of the natural community that has been lost. Thus, an ecological community with many endemics that has lost a high proportion of its area would be ranked at high risk, while a community with few endemics or one that has experienced little conversion would be ranked at low risk (MacKinnon and MacKinnon, 1986; Reid *et al*., 1993).

Agro-biodiversity indicators

Agricultural biodiversity or agro-biodiversity embodies cultural and spiritual dimensions of biodiversity together with the practical and economic values of gaining sustainable rural livelihoods for poor people (Altieri, 1999). Agro-biodiversity can be defined much more broadly as the many ways in which farmers use the natural diversity of the environment for production. It includes farmers' choice of crops, and management of land, water, and biota (Brookfield and Padoch, 1994). It goes beyond the concept of species and genetic diversity of plants and animals to incorporate other aspects of the farming system related to agriculture, namely: genetic resources, crops and non-cultivated edible and non-edible beneficial plants, livestock, freshwater fish, beneficial soil organisms; and naturally occurring biological pest and disease control agents (insects, bacteria, and fungi). The concept also includes habitats and species outside farming systems that benefit agriculture and enhance ecosystem functions.

Natural resource management interventions can engender significant changes in the state of agro-biodiversity (Thrupp, 1998). Agro-biodiversity has therefore been used as an important criterion for agricultural sustainability (Table 5.1). There are no universally accepted indicators of agro-biodiversity. Some practitioners suggest that the index of surface percentage of crops (ISPC), crop agro-biodiversity factor (CAF), genetic variability, and surface variability factors can all be used as useful indicators to monitor changes in agro-biodiversity (McLaughlin and Mineau, 1995). The ISPC expresses the ratio between the number of crops that represent 50% of the cultivated area and the number of crops commercially cultivated. The CAF indicates the relationship between the number of major crops in a given area and the crops that are agroecologically adapted to the prevailing management systems. Genetic variability or diversity refers to variation in the genetic composition within or among species. Traditional Mendelian methods are insufficient to provide a detailed estimation of genetic variability. The process is too time-consuming and is restricted to phenotypic characters. Today this can be overcome by using DNA-based molecular techniques that provide more precise information on genetic variability (Noss, 1990). To some extent, genetic variability in agro-ecosystems can also be inferred qualitatively from the proportional area of a given cultivar within the total cultivated area of that crop. For example, agro-ecosystems where single varieties or hybrids occupy a large share of the cultivated area indicate limited genetic variation. Surface variability refers to the area covered by agricultural crops in a given agro-ecosystem (Merrick, 1990). For example, regions with a large number of crops with similar areal coverage will have higher surface variability than those dominated by only a few crops. How changes in agro-biodiversity can be used to monitor the sustainability related impacts of NRM technologies is illustrated using information on crop diversity and surface percentage of crops that represent aspects of the stability and balance of agricultural systems at the watershed level (Box 5.1). The examples given for two watersheds, Thanh Ha (Vietnam) and Kothapally (India), show how such quantitative indicators as ISPC, CAF, and surface variability of main crops have changed as a result of integrated watershed management interventions (Wani *et al*., 2003b).

Box 5.1. The impact of watershed management on agro-biodiversity.

In an operational scale watershed of the International Crops Research Institute for the Semi-Arid Tropics (ICRISAT) at Thanh Ha, Hoa Binh Province, northern Vietnam, a total of four different crops cover the agricultural surface, which represents a low diversity of commercially cultivated species grown. The CAF for the watershed is 0.25 indicating that only one-fourth of the potentially useful species is exploited. Cereals such as maize and rice together constitute 84% of the agricultural surface. These crops are largely cultivated as monocrops generating a very low ISPC.

Maize is the most extensive crop (83% of cropped area) and its production is based on hybrids bred from exotic or introduced genetic materials. In northern Vietnam fewer than five hybrids have produced more than 80% of maize in the last 15–20 years. Not only the number of prevailing hybrids in the ecosystem needs to be considered but also the parentage of such hybrids. In many cases few parental lines, particularly the male-sterile lines (female parent) are used in producing such hybrids, resulting in a narrow genetic diversity of cultivated hybrids, in contrast to the high genetic diversity found in traditional systems. Due to various NRM interventions in this watershed, the area under maize has declined from 380 ha to 148 ha while the area under groundnut, mungbean and soybean has increased from 18 ha to 250 ha changing the CAF from 0.25 in 1989 to 0.6 in 2002.

During 1998–2002, more pronounced impacts in terms of increasing agro-biodiversity were observed in a 500-ha micro-watershed at Kothapally, Ranga Reddy district, Andhra Pradesh, India. In this watershed the farmers grow a total of 22 crops, and a remarkable shift has occurred in the cropping patterns from cotton (200 ha in 1998 to 100 ha in 2002) to a maize/pigeonpea intercrop (40 ha in 1998 to 180 ha in 2002); thereby changing the CAF from 0.41 in 1998 to 0.73 in 2002.

Agro-ecosystem efficiency indicators

Agro-ecosystem efficiency can be approximated through various productivity and economic efficiency indicators. Crop yield is a land productivity indicator that reflects the efficiency of the system (soil, solar energy, water, etc.), with regard to genetic potential, ecological conditions, management, capital investment and labour use. It denotes the production of economic yield and total plant biomass from application of various inputs from a given parcel of land during a given period. It is used as a biological parameter for the evaluation of a system's behaviour and reflects its state at any given time. It is perhaps the best-known functional characteristic of agro-ecosystems and is widely used as a criterion for the assessment of both the biological and economic sustainability of agricultural systems. To assess the impact of NRM technologies, yield parameters sometimes converted in terms of economic returns serve as important indicators. Further, since yield is a final product that takes into account soil and other growing conditions, time-series yield data from a given system can directly indicate the dynamics and sustainability of the system.

At ICRISAT, Patancheru, operational watersheds have been maintained over the last 26 years and scientists have compared the productivity impacts of different NRM options on Vertisols (Wani *et al.*, 2003a). The best practice included: improved soil and water conservation options such as grassed waterways; land configuration (broadbed-and-furrow (BBF) on grade); integrated nutrient and pest management options; recommended varieties of maize intercropped with pigeonpea; plant population and crop husbandry. The farmers' traditional management practice included: rainy-season fallow; and flat-land cultivation with postrainy-season sorghum grown on stored soil moisture with application of 10 t/ha farmyard manure once in 2 years.

The productivity and sustainability impacts of NRM options were tested using time series yield data during 1977–2002 (Fig. 5.1) along with soil quality parameters. Crop yields increased under both management practices, but the annual productivity growth under improved management (77 kg/ha) is significantly higher than that under traditional management (26 kg/ha). The improved system with an average productivity of 4.7 t/ha has a higher carrying capacity (18 persons/ha) than the traditional system with 0.95 t/ha (4 persons/ha). Improved management is better able to respond to increasing population pressure while higher incomes enhance farmers' capacity to invest in more-sustainable practices.

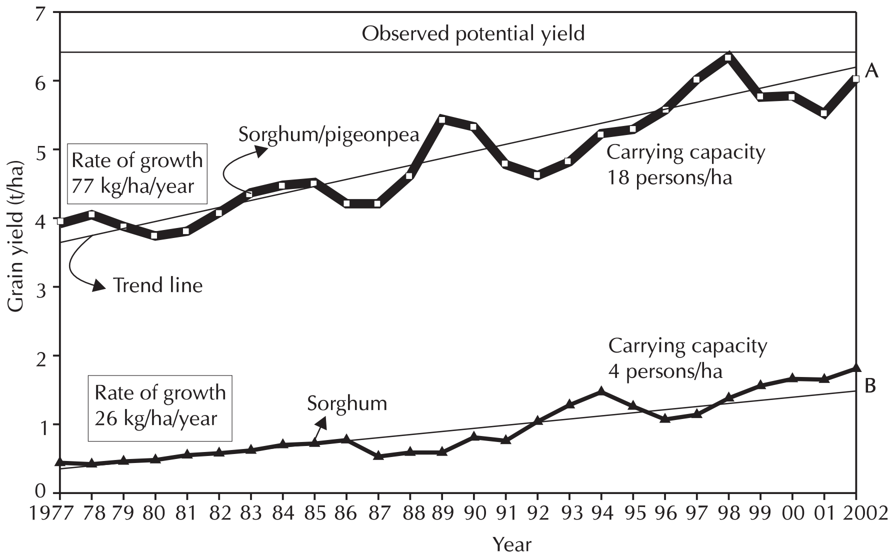

Fig. 5.1. Average grain yields under improved (A) and traditional (B) technologies on a Vertisol watershed at ICRISAT (1977–2002).

The potential yield can also be estimated for a fully optimised production situation using crop simulation models with a fixed limiting constraint such as soil-water availability. The gap between the potential yield that is often greater than that attainable under experimental conditions, and yields under farmers' growing conditions is often referred to as a 'yield gap'. In this sense, NRM impact can also be estimated in terms of the extent to which improved NRM succeeds in reducing the yield gap. The larger the reduction in the yield gap, the higher the success of the intervention in optimising production. Singh *et al*. (2002) used this approach to identify the soybean-growing districts where high yield gaps existed and to identify locations where the yield gaps could be bridged using improved NRM interventions to increase soybean productivity at the district level (Table 5.2). A similar approach was also applied in an operational-scale watershed to assess the potential of improved soil, water, nutrients and crop management options for soybean-based systems at ICRISAT (Singh *et al*., 1999).

Table 5.2. Simulated soybean yields and yield gap for the selected locations in India.

Location	Mean sowing date	Harvest date	Simulated yields (kg/ha) Mean	SD	Mean observed yield[a] (kg/ha)	Yield gap (kg/ha)
Raisen	22 Jun	11 Oct	2,882	1,269	–	–
Betul	19 Jun	08 Oct	2,141	603	858	1,283
Guna	30 Jun	14 Oct	1,633	907	840	793
Bhopal	16 Jun	08 Oct	2,310	615	1,000	1,310
Indore	22 Jun	10 Oct	2,273	939	1,122	1,151
Kota	03 Jul	16 Oct	1,165	936	1,014	151
Wardha	17 Jun	06 Oct	3,040	640	1,042	1,998
Jabalpur	23 Jun	11 Oct	2,079	382	896	1,183
Amaravathi	18 Jun	08 Oct	1,552	713	942	610
Belgaum	17 Jun	30 Sep	1,844	629	570	1,274

[a] Mean of reported yields during 1990–95.

Related to the productivity measure, various economic efficiency indicators like the benefit–cost ratio can also be computed to evaluate the efficiency of agroecosystems. Such indicators can be used to evaluate the economic feasibility of various cropping systems and sustainability enhancing NRM options (Lynam and Herdt, 1989; Tisdell, 1996). A simple economic productivity indicator like the benefit–cost ratio can be computed at the farm level to determine the economic benefits to farmers of adopting new management practices.

Another related economic indicator is the Parity Index that compares the relative efficiency of different crops or income-generating options in response to a given intervention. The relative index is computed as a percentage or ratio of the option that provides the highest net return. When data on benefits and costs are available, such simple agro-ecosystem efficiency indicators can be computed relatively easily. The challenge is in estimating the parity indices when some of the non-market benefits and costs are difficult to value. Application of environmental valuation methods can be useful approaches to estimate the efficiency of the system in such situations.

Environmental services indicators

Various environmental services such as groundwater recharging, reducing silt load and nitrate concentrations in the runoff water, carbon (C) sequestration in vegetation and in the soil, soil formation, reducing levels of greenhouse gases in the environment, etc. generated through NRM are very important but generally difficult to assess using conventional economic methods. Moreover, the benefits of the environmental services may occur off-site, i.e. far away from the point of NRM interventions.

Existing policies and legal frameworks in many developing countries are not able to properly value the environmental services provided by land-use systems and such ecosystem services as those generated by NRM investments. For example, the effects of deforestation, land degradation or environmental degradation on global warming and climate change are difficult to quantify and assess. Similarly, it is difficult to assess the effects of environmental improvements associated with NRM investment practices. Measurement problems and off-site effects complicate the process of monitoring such changes. However, with the advancement of science and technology, new methods and tools are evolving to quantify these environmental benefits. A good example is the measurement of C sequestration benefits from improved NRM, where some progress is being made at the global level. In 1997, the Kyoto Protocol to the United Nations Framework Convention on Climate Change established an international policy context for reduction of carbon emissions and increased carbon sinks in order to reduce global warming and effects on climate change. This has drawn attention to NRM practices that sequester more carbon from the atmosphere.

C sequestration in soils not only reduces atmospheric CO_2 concentrations but also improves the organic matter status and overall fertility of soils. There is great interest in C sequestration in soils and numerous strategies including technical and policy issues for increasing C in cultivated land have been identified (Bruce *et al*., 1999; Izaurralde *et al*., 2001; Pretty and Ball, 2001; Wani *et al*., 2003a; Smith, 2004). The application of nutritive amendments required for biomass production, including the chemical fertilisers that provide N, P, S, etc. (Vlek, 1990; Wani *et al*., 2003a) and organic amendments, and diversification of monocropped cereal systems through inclusion of legumes, all favour build-up of soil C and the improvement of soil quality (Wani *et al*., 1994, 2003a; Paustian *et al*., 1997). It is clear that soils can sequester C and reduce the atmospheric concentration of CO_2.

Several soil and crop management practices affect C sequestration in soil. Lal (1999) reviewed the role of various practices on C sequestration potential in soil (Table 5.3). According to him conservation tillage, regular application of compost at high rates, integrated nutrient management, restoration of eroded soils, and water conservation management all have a relatively high potential for sequestering C and enhancing and restoring soil fertility.

The level of C sequestered by agricultural, agroforestry, and agrihorticultural systems can be quantified using suitable biochemical methods based on data collected from long-term experiments. The amount of C sequestered

Table 5.3. Carbon sequestration potential of various land management practices under dryland conditions.

Management practice	C sequestration potential (t C/ha/year)
Conservation tillage	0.10–0.20
Mulch farming (4–6 t/ha/year)	0.05–0.10
Compost application (20 t/ha/year)	0.10–0.20
Integrated nutrient management	0.10–0.20
Restoration of eroded soils	0.10–0.20
Restoration of salt-affected soils	0.05–0.10
Water conservation management	0.10–0.30
Afforestation	0.05–0.10

Source: Lal (1999)

by vegetation is quantified by assessing biomass accumulation and the C content of the biomass using standard methods of C estimation. Carbon sequestered in soils is estimated by analysing samples from different soil profiles and calculating the stocks in the profile using the bulk density for a given depth and the area covered by a particular system under study. Following the Kyoto Protocol, C sequestered by agricultural and NRM systems, once quantified in C units, can now be valued in economic terms.

Using this approach, Bruce *et al.* (1999) recorded an annual soil C gain of 0.2 t/ha on pasture and rangelands in the USA following adoption of best management practices. In the SAT of India, Wani *et al.* (2003a) evaluated the effect of long-term (24 years) improved management of Vertisols on C sequestration and reported a difference of 0.3 t C/ha/year attributable to NRM. Under improved soil fertility (60 kg N and 20 kg P/ha/year) and land management (BBF to drain excess water) and cropping systems (maize/ pigeonpea intercrop), the soils contained 46.8 t C/ha in 120 cm soil profile as compared to farmers' traditional management practices that contained 39.5 t C/ha. This amounts to a gain of about 7.3 t C/ha over the 24-year period.

Growing knowledge on the C-sequestration benefits of NRM options and the possibilities for C trading have opened new opportunities for C-based rural development in many poor regions where the relative returns to agricultural land use are low. However, several hurdles remain in harnessing such initiatives for community development. For other environmental services, more work is needed in the area of quantification and policy development.

Simulation modelling for the estimation of biophysical changes

Simulation models are mathematical representations of various processes of soil, plant and climate systems in the form of computer programs that describe the dynamics of crop growth in relation to the biophysical environment. These models usually operate in daily time steps. They require soil, climate, crop, and management data as inputs and produce output variables describing the state of the crop and the soil at different points in time. The models are used to evaluate soil and crop management options for a given environment, to extrapolate the results of management strategies over time and space, and to study the long-term effects of NRM on productivity, soil quality, and the environment. Before the models are used to do this, they must be validated with observed field data for the specific soil–plant processes to be evaluated. There are several kinds of simulation models available in the literature, each with its own strengths and weaknesses. Selection of a model depends on its strengths, the purpose for which it is used, and the availability of input data in a given environment for model operation. Table 5.4 provides a summary of different types of simulation models.

Table 5.4. Simulation models and their potential application.

Acronym	Extended name	Purpose/simulation
APSIM	Agricultural production systems simulator	Effect of agronomic management practices on crop productivity and changes in soil properties
APSIM–SWIM	Agricultural production systems simulator – soil water infiltration and movement	Effect of agronomic practices on crop productivity and soil processes using SWIM module
CENTURY	-	Change in nitrogen (N), organic carbon (C), phosphorus (P), and sulphur (S) in the soil due to changes in agronomic management of various land-use systems
CERES–RICE	Crop estimation through resource environment synthesis for rice	A component model of DSSAT v3.5
DSSAT v3.5	Decision support systems for agrotechnology transfer, version 3.5	Effect of agronomic management practices on crop productivity and changes in soil properties
PERFECT	Productivity, erosion, runoff functions to evaluate conservation techniques	Effect of various conservation techniques on runoff, soil erosion and crop productivity
RothC–26.3	Rothamsted Carbon model, version 26.3	Carbon changes in the soil in response to various land and crop-residue management practices
SCUAF	Soil changes under agroforestory	C and N changes in soils in response to land clearing and agronomic management of agroforestory systems
SIMOPT2–MAIZE	A simulation-multi-criteria optimisation software for maize	Optimise productivity and N losses using CERES–MAIZE model
WATBAL	A simple water balance model	Estimate the soil moisture regimes of a site from readily available climatic data

The sustainability of production, soil quality and other environment resources are the major impact factors of NRM. Detailed empirical research over a period of time and space is required to quantify the impacts of improved management on these desirable outcomes. However, such long-term studies are costly and time-consuming; simulation models provide a cost-effective and efficient complementary approach to long-term field experimentation for *ex ante* analysis of the long-term impacts of NRM options. These models have often been validated on a plot or field scale. On a watershed scale, the models can be integrated with GIS to study spatial variability effects on crop production and the state of natural resources, enhancing their capability for up-scaling and user-friendly mapping. Thus, the models are useful when undertaking temporal trend analyses, and when incorporating a spatial component to assess the NRM impact on various processes governing sustainability. For example, considering past trends and current management practices using simulation models, Fisher *et al*. (2002) assessed the long-term (25–50 year) impact on crop yields of climatic change including the occurrence of droughts. In the following section, examples and approaches for assessing the impact of NRM using simulation models and GIS are discussed.

Impacts of land surface management on runoff, soil erosion and productivity

Runoff, soil loss and nutrient depletion are the major agents of human-induced land degradation (Pathak *et al*., Chapter 3, this volume; Sahrawat *et al*. Chapter 4, this volume). Freebarin *et al*. (1991) used the results of two long-term field experiments to develop coefficients for soil processes and to validate the PERFECT model for two sites in Australia. Then they used the model to assess the impact of various management practices such as crop/fallow sequences, tillage, and effects of various amendments that modify soil physical processes. Long-term (100^{+} years) simulated results showed the decline in yields associated with soil erosion and removal of the previous season's crop stubble from the field. Singh *et al*. (1999) used DSSAT v3.5 to assesses the impact of two land surface configurations on surface runoff and yields of soybean and chickpea using experimental data (2 years) and historical weather data (22 years). It was found that in most years BBF decreased runoff from the soil, but had a marginal effect on yields of soybean and chickpea. The decreased runoff was associated with an increase in deep drainage and reduced soil loss. Wani *et al*. (2002) used a simple WATBAL model (Keig and McAlpine, 1974) along with GIS to assess the available soil moisture and excess runoff water available for harvest at the district level.

Nelson *et al*. (1998) used the APSIM model to evaluate the sustainability of maize crop management practices in the Philippines using hedgerows to minimise land degradation. Intercropping maize with hedgerows was used to assess the long-term sustainability of maize production due to reduced soil erosion. In the absence of hedgerows, continuous maize cultivation turned out to be unsustainable in the long term, although the inclusion of a fallow period slowed the productivity decline by spreading the effect of erosion over a larger cropping area.

Impact of nitrogen management on leaching

Field experiments conducted in environments with highly variable climates may give misleading results, as the years in which they are conducted might not represent the long-term average. In such cases, simulation models provide a rigorous mechanism to assess the long-term risks of specific management options. Verburg *et al*. (1996) using the APSIM–SWIM model assessed the long-term (33 years) impact of different irrigation management strategies and N application on sugarcane yield and nitrate leaching. Alocilja and Ritchie (1993) used the SIMOPT2–MAIZE model to investigate the trade-offs between maximised profits and minimised nitrate leaching. Thornton *et al*. (1995) took the analysis a step further by linking it to GIS with spatial databases of soils and weather to analyse the influence of N management on crop yield and leaching at the regional level. Such a linkage not only allowed an analysis of the spatial variability due to different soil types and weather across the region, but also the temporal variation associated with changes in weather.

Singh and Thornton (1992) simulated the effects of various nutrient management strategies on N leaching from rice fields in Thailand using the CERES–RICE model. The results obtained from a 25-year simulation suggested that on well-managed clayey soils, medium- to high-input agriculture can be highly productive and environmentally sustainable. Leaching losses were considerably higher on sandy soils than on clay soils. The N loss was inversely related to the depth of urea incorporation and could be minimised by deep placement.

Production systems and soil quality

A number of cropping systems simulation models incorporate the simulation of soil processes such as soil water dynamics, decomposition and mineralisation of added crop residues and organics, with simulation of N fixation by legumes, thus providing the opportunity to evaluate yield responses to application of organic matter and the integration of legumes. Probert *et al*. (1998) used the APSIM for simulating the performance of hypothetical chickpea–wheat rotations on clay soils in Queensland, Australia. The simulation results indicated that soil organic matter (SOM) and N steadily declined over 25 years under continuous wheat cropping without N fertiliser application, whereas the integration of chickpea into the rotation considerably reduced the soil fertility decline. Similar results were obtained by Bowen and Baethgen (1998) using the DSSAT models to assess the long-term sustainability impacts of various cropping systems in Brazil. A continuous maize–fallow system without fertiliser application caused maize yields to decline gradually over 50 years, whereas a green-manure–maize–fallow system was able to sustain yields over the same period.

Menz and Grist (1998) applied the SCUAF model to evaluate the impact of vegetation burning and changing the length of the fallow period in shifting cultivation systems in Indonesia. The results were used to assess the economic viability of different management options in terms of returns from rice cultivation. It was concluded that although more-intensive cultivation carried a future yield penalty, systems with extended fallow periods were unable to overcome the more immediate economic gains to be made from intensive cropping.

Shepherd and Soule (1998) developed a farm simulation model to assess the long-term impact of existing soil management strategies on productivity, profitability, and sustainability of farms in western Kenya. The model linked soil management practices with nutrient availability, crop and livestock productivity, and farm economics. A wide range of soil management options was simulated, including crop residue and manure management, soil erosion control measures, green manuring, crop rotations, and N and P fertiliser application. The dynamic model was applied for Vihiga district in western Kenya, and was used to assess the sustainability of the existing systems using three household groups (farms) in the area. It was shown that the low and medium resource endowment farms had declining SOM, negative C, N and P budgets, and low productivity and profitability. The high resource endowment farms, on the other hand, had increasing SOM, low soil nutrient losses and were productive and profitable. This approach showed the dangers of relying on nutrient balances of an 'average' farm-type. The authors concluded that when the required capital is available, farmers can invest in NRM options that improve profitability without sacrificing long-term sustainability.

Carbon sequestration

Conducting long-term experiments could also be used to monitor the changes in soil C contents associated with NRM investments. Alternatively, soil C simulation models can also be used to simulate the impact of NRM interventions on C sequestration in soils on farm and catchment scales. The most commonly used models are RothC–26.3 (Coleman and Jenkinson, 1996) and CENTURY (Parton *et al*., 1987). More recently DSSAT v3.5 (Gijsman *et al*., 2002) and APSIM softwares have also incorporated soil C balance subroutines to simulate soil C change along with analysis of crop productivity. The simulation approach avoids long-term experimentation and the models can be validated using empirical data along with known biochemical relationships in the soils. Probert *et al*. (1998) used the CENTURY and APSIM models to examine the effects of tillage, stubble management and N fertiliser on the productivity of a winter-cereal–summer-fallow cropping system in Australia. Both models predicted that for this continuous cereal cropping system there would be a decline in SOM (organic C = SOM/1.72).

Furthermore, the C stocks at regional or ecoregional levels can be calculated using GIS and measurements of C at benchmark sites for a given soil series and management system. Velayutham *et al*. (2000) calculated C stocks in India using information on soil series and measurements at benchmark locations that were extrapolated using GIS techniques.

Monitoring Spatial and Temporal Dynamics of Agro-ecosystems

Natural resource management interventions result in multi-faceted biophysical impacts including the establishment of vegetation cover, reduction in soil loss, increase in the number and spatial coverage of water bodies, changes in water quality, and groundwater recharge. These changes can be monitored over space and time. Remote sensing and GIS are the most suitable tools for monitoring such spatial and temporal dynamics. By providing synoptic and repetitive coverage at regular intervals, remote sensing offers high potential for monitoring observable changes. Remote sensing refers to making an observation on a feature or phenomenon without being in physical contact with it. In nature, every object reflects and/or emits a fraction of incident radiant energy that makes it possible to derive coded information that will help to remotely sense the condition of the resource under study. *In situ* air and/or spaceborne spectral measurements are made to detect various natural and/or cultural features. GIS is a tool used to store, retrieve, analyse and integrate spatial and attribute data. The system helps to generate development plans by integrating information on natural resources with the ancillary information, and to develop a decision-support system.

Impact assessment of NRM technologies/interventions often involves the evaluation and monitoring of changes in selected indicators at a reference site. For this purpose, the reference site needs to be characterised in terms of its natural resources and environmental conditions. Remote sensing holds very good promise for providing information on changes in land use/land cover, quality of surface water, vegetation cover and dynamics of degraded land, which can in turn be used as indicators of agricultural sustainability. Since NRM is implemented on various scales ranging from plot/farm to watershed and river basin, impact assessments also need to be made using a database with a matching spatial scale. In this context, spaceborne/airborne spectral measurements with varying spatial resolution, ranging from about 1 km (geo-stationary satellites) to the sub-metre level (Quickbird–II mission), provide the desired details of terrain features that enable assessment of the impact of diverse biophysical NRM impacts. How spaceborne multispectral data could be used to monitor the spatial and temporal dynamics of agro-ecosystems is discussed below. A synthesis of different satellite systems used in monitoring biophysical dynamics of agro-ecosystems is given in Appendix 5.1.

Land-use change and intensification

Gemini and Apollo space photographs were used to map land use/land cover in the late 1960s (Aldrich, 1971), but operational use of spaceborne multispectral measurements for land use/land cover mapping only began with the launching of the Earth Resources Technology Satellite (ERTS–1), later named Landsat–1, in July 1972 (Anderson *et al*., 1976). Subsequently, data from other satellites in the Landsat series, along with the Satellite pour observation de la terre (SPOT) and the Indian Remote Sensing Satellite (IRS–1A/–1B/–1C/–1D) have been operationally used to collect information on land use/land cover on various scales ranging from regional to micro-watershed level (Landgrebe, 1979). The utility of spaceborne multispectral data in the detection of changes in land-use patterns is illustrated by an example from a micro-watershed of Ghod catchment in Maharashtra, India. The Linear Imaging Scanning Sensor (LISS–III) aboard IRS–1C/–1D, and Landsat–5 Thematic Mapper (TM) data for the period 1985/86 and 1999/2000 were used to generate agricultural land use maps (not shown) and data (Table 5.5). The area estimated from analysis of satellite data revealed that compared with 166 ha during the period 1985/86, the area under postrainy-season cropping had increased to 251 ha during 1999/2000. A similar trend was observed in the spatial extent of other land uses.

Table 5.5. Impact of NRM on land use in gd24 micro-watershed, Ghod catchment, Maharashtra, India, during 1985/86 to 1999/2000.

	Area (ha)	
Land use	1985/86	1999/2000
Rainy season *(Kharif)*	192	193
Postrainy season *(Rabi)*	166	251
Double crop	144	243
Fallow	158	99
Forest	6	6
Scrubland	256	177
Barren/rocky	411	360
Water bodies	0	4
Built-up	0	0
Total	1,333	1,333

Vegetation cover

Amongst various biophysical parameters relevant to NRM impact assessment, vegetation density and vigour, and above ground biomass can be detected from spaceborne spectral measurements. Higher reflection in the near-infrared region (NIR) and considerable absorption in the red region (R) of the spectrum of green plants enables their detection using remote-sensing techniques. Absorption in the red region is due to the presence of chlorophyll in plant leaves, while reflection in the NIR region results from the inter-cellular space of plant leaves. Various vegetation indices – normalised difference vegetation index (NDVI), transformed vegetation index (TVI), and soil-adjusted vegetation index (SAVI) – can be derived from spectral measurements that are related to biomass, vegetation density and vigour, and crop yield. The NDVI is most commonly used as a surrogate measure of the vigour and density of vegetation, and is computed from spectral measurements in the red (0.63–0.69 μm) and near-infrared (0.76–0.90 μm) region as follows:

$$\mathrm{NDVI} = \frac{\mathrm{NIR} - \mathrm{R}}{\mathrm{NIR} + \mathrm{R}} \tag{4}$$

where NIR is spectral responses of vegetation in the near infrared and R for red regions of the spectrum. Index values can range from – 1.0 to 1.0, but vegetation values typically range between 0.1 and 0.7. Higher index values are associated with higher levels of healthy vegetation cover. NDVI can be used as an indicator of change in relative biomass and greenness.

The utility of NDVI for assessment of vegetation development is illustrated in Table 5.6 for a micro-watershed in the Ghod catchment, Maharashtra, India. Soil and water conservation interventions resulted in the establishment of vegetation cover during the period 1985/86 to 1999/2000, that could be monitored through temporal NDVI images. As is evident from Table 5.6, the area under the three NDVI ranges (0.20–0.39, 0.40–0.59 and 0.6) has increased substantially (National Remote Sensing Agency, 2001a). This shows that the area under various levels of vegetation cover has increased from 1985 to 1999.

Table 5.6. Vegetation dynamics in gc3b micro watershed in Ghod catchment, Maharashtra.

NDVI range	Area (ha) 1985/86	Area (ha) 1999/2000
<0.0	1,519	1,312
0.00 – 0.19	936	859
0.20 – 0.39	329	469
0.40 – 0.59	117	227
>0.60	96	130
Total	2,997	2,997

Monitoring changes in surface water resources

Because of its characteristic absorption feature in the near-infrared region of the electromagnetic radiation, surface water is easily detected in remotely sensed images. The high transmittance of incident radiation in the blue region (0.45–0.52 μm) enables the discrimination of clear water from turbid water. The turbidity causes most of the incident radiation in the blue region to reflect, resulting in a higher spectral response. Moore and North (1974) and Adam *et al.* (1998) used optical and microwave sensor data to delineate floodwater boundaries. Lathrop and Lillesand (1986) used Landsat–TM data to assess water quality in Southern Great Bay and West Central Lake, Michigan, USA. The temporal change in the spatial coverage of reservoirs after NRM interventions has been studied in the Ghod catchment (Fig. 5.2). While the water spread in the reservoir was about 3 ha in 1985, it increased to 16 ha by 1999 following the implementation of soil and water conservation measures (National Remote Sensing Agency, 2001a).

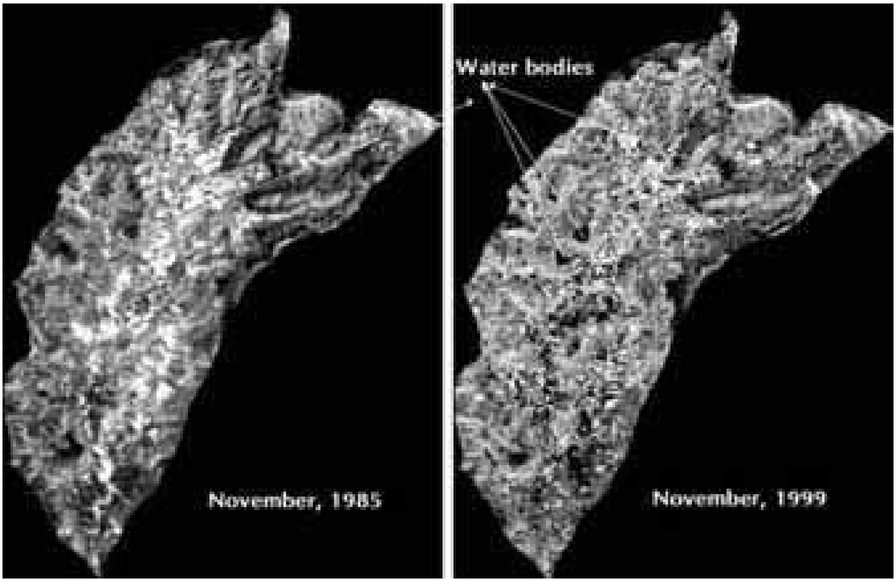

Fig. 5.2. Changes in surface water resources in gc1f micro-watershed, Ghod catchment, Maharashtra, India.

Monitoring the dynamics of degraded lands

Natural resource management interventions in degraded land areas often result in improvements in soil quality and gradual improvement in vegetation cover. Spaceborne multispectral images have been extensively used to inventory and study the dynamics of eroded lands (Wu *et al*., 1997), salt-affected soils (Dwivedi *et al*., 2001), waterlogged areas (Wallace *et al*., 1993), areas of shifting cultivation (Dwivedi and Ravi Sankar, 1991) and the land affected by tanneries' effluents (National Remote Sensing Agency, 1999). The following examples illustrate the use of Earth Observation Satellite data in this endeavour.

Eroded lands

Investment in soil conservation measures in a given area, generally, results in reduced soil loss, reduced soil erosion, and improved soil moisture status, and vegetation cover/biomass. The extent of land degradation is directly related to ground cover that can be quantified using remote sensing data. An illustrative example of eroded lands in the 'rg2h' mini-watershed of the Ramganga catchment, Uttaranchal Pradesh, northern India, during the periods 1985/86 and 1999/2000 is shown in Fig. 5.3. The figure shows that

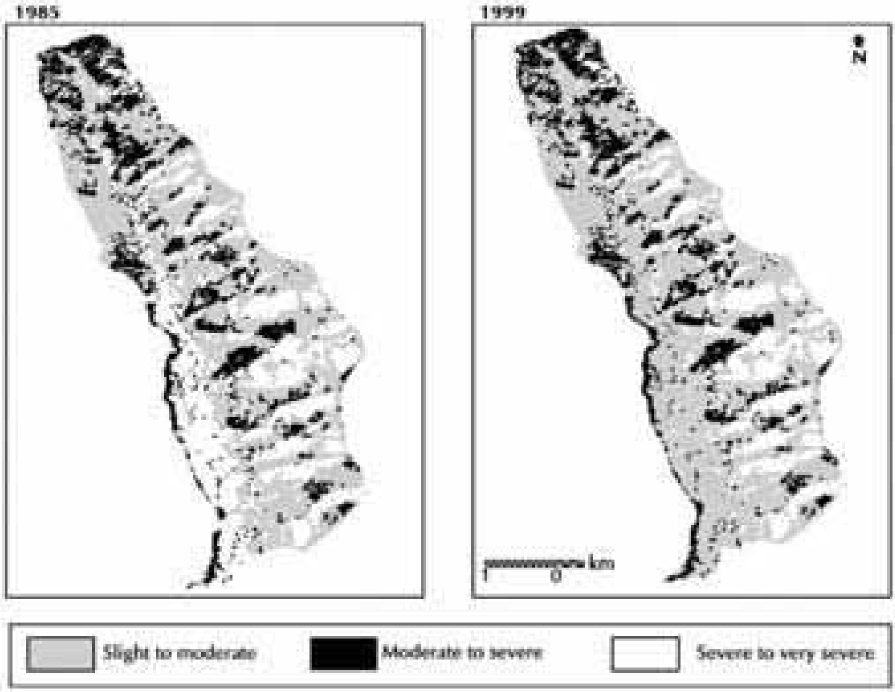

Fig. 5.3. Change in spatial extent and distribution of eroded lands in rg2h micro-watershed, Ramganga catchment, Uttaranchal, northern India.

there has been substantial shrinkage in the spatial extent of moderately eroded lands with concomitant increase in the slightly eroded category (National Remote Sensing Agency, 2001b). In 1985 an estimated 691 ha of land suffered due to moderate soil erosion. By 1999, this had been reduced to 457 ha while the slightly eroded category expanded to 1128 ha from 901 ha in 1985.

Waterlogged areas

Waterlogging in arid and semi-arid regions with alternate wet and dry periods leads to the development of soil salinity. By virtue of the very low response of water in the near-infrared region of the spectrum, the detection of waterlogged areas, especially those with surface ponding or a thin film of water at the surface from remote sensing images is easy. Figure 5.4 shows an example from Mahanadi Stage-1 command area in Kendrapara district, Orissa, eastern India. Gentle slopes and the presence of lenses of clay that act as a hydrological barrier, and irrigation by flooding have contributed to the development of waterlogging. There has been an appreciable increase in the spatial extent of both seasonally and perennially waterlogged areas. Whereas an estimated 389 ha of land were found to be subject to seasonal waterlogging in 1985, by 1999 this had risen to 442 ha.

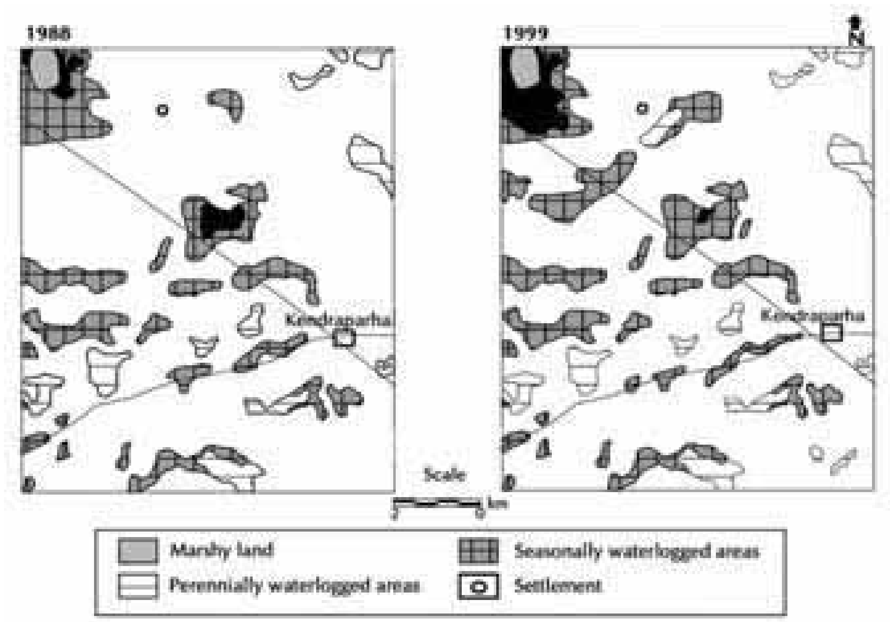

Fig. 5.4. Dynamics of waterlogging in part of Kendrapartha district, Mahanadi Stage-I command, Orissa, India.

Summary and Conclusions

Assessing the multi-dimensional impacts of NRM interventions – especially in non-tangible environmental services – is not an easy task. Monitoring selected indicators through direct observation during and after project implementation or through simulation modelling is a useful approach that will enhance options for evaluating the impacts of NRM interventions. Difficulties on various scales could be overcome through the application of such available tools as GIS and remote sensing. Off-site impacts on ecological functions and ecosystem services such as the effects on water quality, land quality, siltation, groundwater recharge, and C sequestration can also be assessed by systematic monitoring using remote sensing and ground-truthing measurements.

In this chapter various indicators and tools that can be used to monitor the impacts of NRM interventions were presented. They focused on biophysical indicators for ecosystem services and discussed various tools used to generate data on such indicators. Agro-biodiversity and agro-ecosystem efficiency indicators can be applied on different spatial scales. The impacts of NRM technologies on C sequestration and other ecosystem services can be either measured directly through long-term studies or simulated using agro-biological simulation models. The latter approach is becoming increasingly popular as long-term experimentation and monitoring become either impossible or highly costly. However, the approach requires climatic and agronomic data to estimate potential impacts by calibrating the models to specific local conditions.

Remote sensing in conjunction with *in situ* observations/measurements (ground-truthing) offers tremendous potential in providing timely information on the spatial extent and temporal behaviour of various indicators on scales ranging from micro-watersheds to regional/ecoregional levels. Remote sensing methods are being used to monitor changes in land resource conditions, vegetation dynamics, surface water resources, and to assess changes in levels of land degradation. In the future, the impact of NRM on such environmental services as C sequestration and groundwater recharging could also be monitored or derived from satellite images as new satellites equipped with an array of sensors are launched. On a watershed scale, crop simulation models and water balance models can be important tools for evaluating the biophysical impacts of proposed interventions. Several indicators including those for agro-biodiversity and agro-ecosystem efficiency could also be useful at the micro-watershed level.

Such recently launched satellites as Resourcesat–1(IRS–P6) with varying spatial resolution ranging from 56 m from Advanced Wide Field Sensor (AWiFS) to 23 m from LISS–III to 5.8 m from LISS–IV offer unique opportunities to monitor biophysical impact indicators on different spatial scales. Integrating panchromatic data with 2.5-m and 1-m spatial resolution from such future Earth observation missions as Cartosat–1 and Cartosat–2, will further enhance the value of data from the Resourcesat–1 satellite.

Despite the technological advances and the impressive progress made in the last few years, there will be a need for future research to enhance and develop methods and indicators to assess NRM impacts on ecoregional scales. Such indicators will complement and enhance economic approaches for evaluating the impacts of NRM interventions, especially on larger spatial scales. Methods and indicators for the quantification of various difficult-to-quantify environmental services and for monitoring such non-quantitative impacts as effects on implementation processes, policies and institutional arrangements, changes in social capital, and capacity building and empowerment of local communities will also need attention in future research.

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Appendix 5.1. Brief summary of remote sensing satellites and their characteristics.

Satellite	Owner	Launch	Sensors[a]	Spectral range (mm)	Resolution (m)	Swath (km)	Revisit (days)
Cartosat–1	India	Expected launch 2004	PAN	0.55–0.75	2.5	30	6–7
Cartosat–2	India	Expected launch 2006	PAN	0.55–0.75	<1	10	
IKONOS–II	USA	1999	PAN	0.45–0.9	1	11	1–4
			MSS	0.45–0.52, 0.52–0.6 0.63–0.69, 0.76–0.9	4		
IRS–1A and 1B	India	1988 and 1991	LISS–I	0.45–0.52, 0.52–0.59 0.62–0.68, 0.77–0.86	72.5	148	22
			LISS–II	0.45–0.52, 0.52–0.59, 0.62–0.68, 0.77–0.86	36.25	74	
IRS–1C and 1D	India	1995 and 1998	WiFS	0.62–0.68, 0.77–0.86	189	810	5
			LISS–III	0.52–0.59, 0.62–0.68 0.77–0.86	23.6	142	24–25
				1.55–1.70	70.8	148	
			PAN	0.50–0.75	5.8	70	
Landsat–1	USA	1972	MSS	0.5–0.6, 0.6–0.7, 0.7–0.8, 0.8–1.1	82	185	18
Landsat–5	USA	1984	MSS	Same as Landsat–1	Same as Landsat–1		
			TM	0.45–0.52, 0.52–0.6 0.63–0.69, 0.76–0.9 1.55–1.75, 2.08–2.35	30	185	16
				10.4–12.5	120		
Quickbird–II	USA	2001	MSS	0.45–0.52, 0.52–0.6 0.63–0.69, 0.76–0.89	2.5	17	1–3.5
			PAN	0.45–0.9	0.61		

Appendix 1. Continued.

Satellite	Owner	Launch	Sensors	Spectral range (mm)	Resolution (m)	Swath (km)	Revisit (days)
Resourcesat–1	India	2003	LISS–IV	0.52–0.59, 0.62–0.68 0.77–0.86	5.8	23.9 (MX) 70 (mono)	5
			LISS–III	0.52–0.59, 0.62–0.68 0.77–0.86, 1.55–1.70	23.5	141	24
			AWiFS	0.52–0.59, 0.62–0.68 0.77–0.86, 1.55–1.70	56 (nadir) 70 (pixel end)	740 (combined) 370 (each head)	5
SPOT–4	France	1998	MLA	0.5–0.59, 0.61–0.68 0.79–0.89, 1.58–1.75	20	60	26
			PLA	0.61–0.68	10		

[a] Sensors: AWiFS = Advanced Wide Field Sensor, LISS = Linear Self-Scanning Sensor, MSS = Multi Spectral Scanner, MLA= MSS Linear Array, PAN = Panchromatic, PLA = Panchromatic Linear Array, TM = Thematic Mapper, WiFS = Wide Field Sensor.

11 Assessing Economic and Environmental Impacts of NRM Technologies: An Empirical Application Using the Economic Surplus Approach

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Introduction

This chapter addresses methodological and empirical complexities in assessing the impact of crop and resource management research through a concrete case study. It develops an applied economic surplus analysis of welfare gains, using farm survey data to measure farmer benefits from increased yields, reduced unit costs, and higher income. The environmental aspects of natural resource management (NRM) research impacts present special challenges in measurement across time and space. Farmers' perceptions of long-term environmental changes are highlighted as a means to augment or substitute for narrower quantitative indicators.

The case of groundnut production technology (GNPT) in central India illustrates the methodological and empirical issues in estimating research payoffs to NRM research investments. The GNPT was developed for the semi-arid tropics (SAT), a region usually characterised by water scarcity, low soil fertility and land degradation. Impact analysis of GNPT presents estimated costs and benefits using the principle of economic surplus and complements this with a detailed account of both quantitative and qualitative information provided by scientists and experts, including farmers.

Groundnut production technology (GNPT)

The research and development team that developed the GNPT package aimed to raise groundnut production by generating research information on various groundnut crop production components and integrating them into a 'package' of technology options. The technology package that was developed in 1986 integrates crop and resource management options detailed in Table 11.1. These components can be divided into five broad categories: land, nutrient, water, insect and pest management, and improved varieties.

Table 11.1. Technology components of the groundnut production technology (GNPT).

Component		Improved package (GNPT)	Local practice
C1	Land management Seedbed	Raised bed and furrow (RBF)	Flat
C2	Nutrient management		
	Farmyard manure	5–12 t/ha	10 t/ha
	Ammonium sulphate	100 kg/ha	Diammonium phosphate: 100 kg/ha
	Single superphosphate	300–400 kg/ha	Murate of potash: 100 kg/ha
	Zinc sulphate	10–20 kg/ha every 3 years	20 kg/ha every year
	Ferrous sulphate	2–3 kg/ha	–
	Gypsum	400 kg/ha	200 kg/ha
C3	Water management	Furrow or sprinkler to improve efficiency of water use	Flood
C4	Disease and pest management (effective control of insects, diseases and weeds, seed dressing/treatment)	Bavistin, dimethoate, monocrotophos	Need based
	Seed dressing	Thiram, Bavistin or Dithane M 45	Thiram
C5	Seed		
	Improved variety	ICRISAT varieties	Local varieties
	Seeding rate	125–150 kg/ha	120–125 kg/ha

During 1987–1991, International Crops Research Institute for the Semi-Arid Tropics (ICRISAT), through its Legumes On-Farm Testing and Nursery (LEGOFTEN) Unit, was an active partner with the Indian Ministry of Agriculture and other agencies in identifying and demonstrating appropriate technology options for increased groundnut production. The team reviewed all available and relevant research information and carefully identified production constraints in the major oilseed-producing regions of India. This package was thoroughly discussed with the national agricultural research service (NARS) and State Departments of Agriculture. This collaboration in a technology exchange programme provided ICRISAT with an opportunity to confirm the suitability and viability of the GNPT concept in farmers' fields. Although some components of the package (i.e. improved varieties, fertilisers, seed dressing) were already being used by farmers, ICRISAT's value addition took the form of information on appropriate timing and dosage rates of inputs.

The two new essential innovations introduced were land and water management. The land management component of the GNPT entails preparation of raised-beds and furrows (RBF) for groundnut production (Fig. 11.1). Compared to the practice of traditional farmers, who used 1–2 harrowings to sow groundnut on flat land, the RBF technologies were designed to reduce soil erosion, provide surface drainage, concentrate organic matter and fertiliser application, and reduce soil compaction around plants. Over a period of time, the concept of RBF was modified to suit the requirements of the farmers into a narrow-bed and furrow configuration, i.e. a bed of 75 cm, with ridge and furrow systems. The water management component was introduced to improve water use efficiency through furrow and sprinkler irrigation.

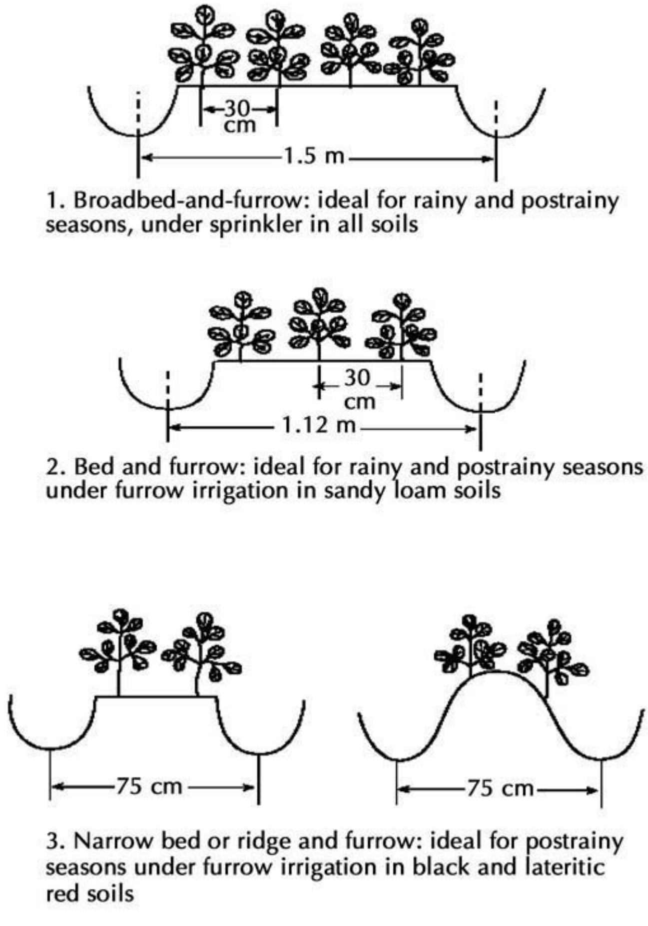

Fig. 11.1. The raised-bed and furrow (RBF) method of groundnut cultivation.

Groundnut Production and Markets in India: Background

India is the world's second largest producer of groundnut after China. Groundnut is one of the most important food and income-generating oilseed cash crops for smallholder farmers in semi-arid India. About 80% of the groundnut crop is rainfed, and is grown in southern, western, and parts of central India during the southwest monsoon. The remaining 20% is irrigated. Groundnut is mostly cultivated in red sandy soils (Alfisols) in many states, but it is also grown in shallow to medium-deep black soils in some parts of the country.

Groundnut yields in India vary widely depending on the production system (Freeman *et al.,* 1999). Rainfed groundnut yields roughly 0.9 t/ha, while the irrigated crop yields about 1.6 t/ha. Important improved groundnut cultivars include TMV 2, SB 11, CG 2, JL 24 and J 11, although these have never completely replaced the local cultivars. After the introduction of GNPT by LEGOFTEN, the area under groundnut production in India increased from 6.84 million ha in 1987/88 to 8.67 million ha in 1991/92 and groundnut production increased from 5.88 million t in 1987/88 to 7.07 million t in 1991/92. Rainy-season groundnut yields increased from 700–1000 kg/ha to 1.5 t/ha; postrainy season-yields rose from 2 to 4 t/ha, and summer yields rose from 1 to 3 t/ha after the introduction of GNPT.

Groundnut demand increases were driven by population growth, although the increase was moderated by rising prices. About 80% of Indian groundnuts are crushed for oil, and groundnut remains the vegetable oil of preference; but its share in the vegetable oil market is declining as consumers shift to such cheaper alternatives as rapeseed, sunflower, and imported palm oil. Large quantities of the groundnut meal produced in India are traded. Groundnut oil is thinly traded because in India substantial quantities of the oil produced are domestically consumed.

Methods for Research Evaluation

The unique empirical challenges of NRM impact assessment include both problems of measurement, and the attribution of research impacts. An impact analysis begins by measuring research benefits. Information on the actual cost of research and development (R&D) and technology transfer is combined with the stream of benefits based on the rate of technology uptake or levels of adoption. The approach quantifies those impacts that were amenable to quantification, while systematic documentation describes those that were difficult to quantify. For a five-component package like GNPT, the research evaluation includes measurement of the stepwise adoption of various technology options, estimates of on-farm benefits, and the relative significance of specific components among quantifiable variables. For the non-quantifiable impacts, researchers and farmers are important sources of detailed descriptions that may serve as a basis for evaluating as many effects as possible, or qualitatively understanding associated research impacts.

Research impacts documentation

The practical measurement of research impacts necessarily involves tracking and understanding the process based on detailed description by both researchers and research beneficiaries. In the absence of hard facts or documented data, detailed descriptions are an important way to understand the basis for estimates of costs and benefits associated with economic and environmental effects.

Because post-project long-term monitoring of GNPT was not undertaken, a systematic process of documentation was crucial for the evaluation process in order to carefully delineate various types of impacts: market and non-market, on-site and off-site, as well as intra- and inter-temporal effects. The implications of these aspects for impact assessment also require the analysis of counterfactuals for non-market effects. Additionally, the complexity of estimating impacts considering economic vs. environmental effects is recognised when some effects are already reflected in yield gains, but some environmental effects are non-quantifiable and do not relate to markets.

Data

Information was collected through farm interview surveys using a structured questionnaire, focus group meetings and participatory rapid rural appraisals, together with interviews with researchers on technical aspects of GNPT. Data on the following aspects were collected from farmers for the 1994/95 crop season:

1. Size of holding, total sown area, irrigated and non-irrigated areas
2. Land use and cropping pattern
3. Cost of groundnut production
4. Input and output data
5. Crop yields and prices
6. Farmer perceptions of sustainability issues and the constraints to adoption of GNPT.

Information on adoption trajectories for different technology options was collected, including:

1. Total groundnut area
2. First year of adoption of different GNPT components
3. Extent of adoption of different GNPT components in the first year
4. Extent of adoption during the period 1992–1994
5. Modification in technology components, if any.

District-level data for area and production were compiled from the Maharashtra State Department of Agriculture records, and disaggregated data below the district level were obtained from the Office of the Agricultural Development Officer (ADO) in each district. Rates of adoption obtained from the survey were also crosschecked with the ADO. Price data were re-collected from seed dealers and several traders dealing with the GNPT components. Estimates of elasticities used earlier estimates by Murty (1997), Radhakrishna and Ravi (1990) and ACIAR (1992), and were validated using expert opinion.

The sample

Multi-stage stratified random sampling (using size of holding and intensity of groundnut cultivation as the basis for stratification) was used to select a representative group of groundnut farmers in order to assess the adoption and impact of different GNPT components. The technology was originally targeted at eight states in the Indian SAT: Andhra Pradesh, Gujarat, Karnataka, Madhya Pradesh, Maharashtra, Orissa, Tamil Nadu, and Uttar Pradesh. However, only in Maharashtra did government and non-government agencies follow up with the dissemination of technologies, and the State Ministry of Agriculture recommended the full GNPT package. Since the objective was to assess the adoption and evaluate the impact of the package, the evaluation of its impact therefore focused on Maharashtra.

The first and second stages of sampling involved stratification by the intensity of groundnut cultivation, while the last stage was stratified by size of holding. In the first stage of sampling, all districts growing groundnut were stratified into high and low intensity categories by the total area sown to groundnut. Two districts each from the top 50% and lower 50% intensity groups were selected at random. In the second stage of sampling, each selected district was stratified into three groups of *talukas* (sub-districts) by tercile of area sown to groundnut (high, medium, or low). Similarly, villages in each *taluka* were subdivided into three strata, also by tercile of groundnut sown area (details in Joshi and Bantilan, 1998). In the last stage of sampling, farm households were grouped into large (>4 ha), medium (1–4 ha) and small (<1 ha) categories according to size of farm holding. The final sampling units were identified through random selection of farmers in randomly selected villages in selected *talukas*. The final sample included 355 farm households.

Estimating the adoption pathway

Many crop and resource management technology packages that include several components are adopted component by component in step-wise patterns (Byerlee and Hesse de Polanco, 1986; Traxler and Byerlee, 1992). Establishing an accurate picture of adoption patterns among groundnut farmers can be complex. The five components of the GNPT package can be combined into ten pairs, ten triples, five quadruples, and one set of all five (Table 11.2). The adoption pattern can be established from the survey data by analysing farmers' responses when asked whether they practised different GNPT components. If the answer was yes, the farmer was asked to recall the first year of adoption for different components. Two additional questions were useful: 1. the extent of adoption of different GNPT components in the first year; and 2. the extent of adoption during the last 3 years ending in 1994. Several components of the technology package were already known and had been adopted even before the introduction of the package, and farmers were free to choose and adopt any of its subsets. Hence, adoption sequences were evaluated by tracking discrete subsets of options available to the farmer, for example, all subsets that included at least the land management option (shown as shaded components in Table 11.2). A systematic approach to tracking multiple technology adoption entailed measuring all subsets of technology components that included: 1. at least one option (say, land management); 2. two specific options (say, improved variety and land management); and 3. all options (full adoption).

Table 11.2. All possible combinations of the five components[a] of the groundnut production technology (GNPT) package.

One component adopted	Two components adopted	Three components adopted	Four components adopted	All components adopted
C1	C1C2	C1C2C3	C1C2C3C4	C1C2C3C4C5
C2	C1C3	C1C2C4	C1C2C3C5	
C3	C1C4	C2C3C4	C1C2C4C5	
C4	C3C4	C1C3C4	C2C3C4C5	
C5	C2C3	C1C3C5	C1C3C4C5	
	C2C4	C1C4C5		
	C2C5	C1C2C5		
	C1C5	C2C3C5		
	C3C5	C2C4C5		
	C4C5	C3C4C5		

[a]See Table 11.1 for a description of the components.

Farm survey data also served to estimate and project the adoption patterns of different GNPT components over time. By fitting a logistic function to data on the first year of adoption and data for the period 1989–95, the proportion of farmers affected by GNPT could be projected. The logistic function is defined as:

$$A_{it} = \frac{C_i}{(1 + e^{-(a+bt)})} \tag{1}$$

where A_{it} is the percentage adoption of the i^{th} component of the GNPT in the t^{th} year; C_i is the adoption ceiling of the i^{th} component; b is the rate of adoption; and a is the constant intercept term.

Research benefits and costs

Estimation of market benefits

Underlying the empirical application of the measurement of GNPT impacts is the principle of economic surplus, described in detail in Alston *et al.* (1995) and Swinton (Chapter 7, this volume). This principle is based on the idea that improved technologies enhance productivity or reduce the groundnut producers' unit cost of production, which translates into an outward shift in the producer's supply curve. Considering the conventional, comparative-static, partial equilibrium, closed economy model of supply and demand in the groundnut commodity market, and assuming simple linear demand and supply equations, a parallel supply shift (k) may be expected to occur due to a measurable reduction in unit cost of production when farmers adopt the GNPT technology package. As a point of reference, Fig. 11.2 shows the supply shift from S_0 (without GNPT) to S_1 due to measured unit cost reduction (*ae*) with the adoption of GNPT. For each cropping season, the change in the groundnut consumer surplus (ΔCS) and producer surplus (ΔPS) can be calculated using the formulae

$$\Delta CS = P_0 Q_0 Z (1 + \tfrac{1}{2} Z \eta) \quad (2)$$

$$\Delta PS = (J - Z) P_0 Q_0 (1 + \tfrac{1}{2} Z \eta) \quad (3)$$

where P_0 and Q_0 are the base groundnut price and quantity; $Z = -(P_1 - P_0)/P_0$; k is the unit cost reduction (equal to distance *ae* in Fig. 11.2); $J = k/P_0$; $(P_1 - P_0)$ is the change in market price; and η is the absolute value of the price elasticity of demand.

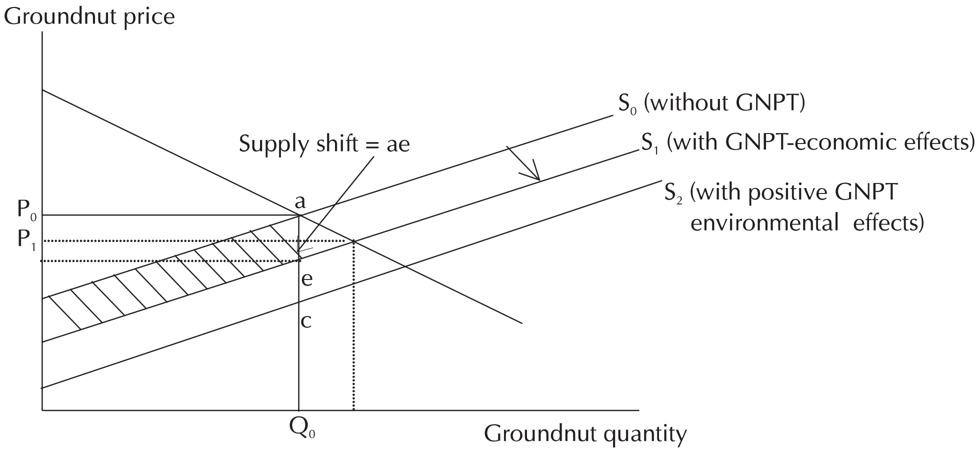

Fig. 11.2. Measurement of economic and environmental benefits due to adoption of groundnut production technology (GNPT) components.

Equations 2 and 3 can be used to calculate the empirical market benefits from adoption of the technology package. Annual gains are computed over the horizon the benefit is expected to accrue at actual adoption levels. The above estimation process only covers benefits accruing due to measurable market effects.

Computing the value of a supply shift

By custom, the magnitude of a supply shift (distance *ae* in Fig. 11.2) is measured by the change in unit cost of production and referred to as '*k*' (following Alston *et al.*, 1995). Establishing the actual supply shift (k) for adoption of GNPT involves understanding the unit cost reduction resulting from adoption of each of the possible GNPT options available to the farmer. This complex procedure can be overcome by categorising discrete subsets of options, among the whole range of 31 GNPT component mixes identified in Table 11.2.

Estimates of the *k*-shift in the supply function can be derived by using information available from on-farm trials. For analysis of the GNPT package, Pawar *et al.* (1993) provided results from trials managed by farmers and supervised by researchers. Different sets of technology options under on-farm trials presented alternative scenarios, namely:

- With and without improved package: This allowed comparison of the improved package of the GNPT, including improved varieties, RBF method, and other management practices, with the local package (full adoption)
- With and without RBF: This set compared only the effects of RBF with the flat land method of groundnut production, keeping the remaining components of the improved technology at their recommended level (at least RBF)
- With and without improved management practices: This option considered the use of improved varieties and compared the improved package of management practices with the traditional management package (i.e. partial adoption involving management practices only holding the effect of improved varieties).

The calculation of the supply shift *k* involves the use of the on-farm input and output data generated for each of the above scenarios. In particular, unit cost of production (Rs/t) was calculated based on total input cost and corresponding yield levels. Pairwise comparison of the unit cost incurred for the improved options versus the benchmark package generated a supply shift estimate for each scenario.

Inclusion of environmental impacts in the evaluation of NRM research benefits

In the process of examining the inclusion of environmental impacts in the evaluation of NRM research, it is useful to conceptualise specific scenarios detailing the nature of impacts by considering whether or not: 1. the effects of the technology intervention can be valued using conventional markets; 2. the effects are on-site or off-site or both; and 3. they have dynamic effects. Following this idea, Lubulwa and Davis (1997) identified four types of impact:

1. *On-site market impacts*. These impacts are specific to the site targeted by the technology intervention, do not have downstream effects, and can be evaluated using conventional markets. One example is exploitative farming systems that do not adequately replenish nutrients extracted during agricultural production. This activity has negative impacts as it reduces soil depth, degrades soil structure, decreases aeration, and increases salinity. The effects are on-site and may also have dynamic effects on crop productivity. These impacts are reflected in declining crop yields and can be valued using markets for the relevant crops.
2. *Off-site market impacts.* This represents off-site effects at locations different from where the technology impacts are targeted (e.g. downstream effects). Using the same example above, downstream effects that can be valued using markets include silting of rivers, reduced capacity for water storage, lowering water-table levels and the high costs of dredging irrigation canals.
3. *On-site non-market impacts.* This type of impact is specific to the site targeted but is not reflected in the marketplace. A good example is the slash and

burn practice used by farmers to expand cultivation area. A major impact of this practice is the loss of ecological biodiversity at the slash and burn site, but this impact cannot be valued using conventional markets. Contingent valuation or other similar techniques would be needed to value such an impact.

4. *Off-site non-market impacts.* This type reflects impacts that affect non-targeted locations as well as future generations. Water purification, carbon sequestration, and reduced flooding are all examples of down-stream benefits resulting from upland watershed management.

Systematic process documentation of the research and impact pathways is necessary in order to understand the source of the impact and quantify the nature of the impact. More importantly, this process documentation enables identification of those variables that have market impacts and those that have non-market environmental impacts. The measurement of environmental effects in monetary terms within the context of the principle of economic surplus draws from changes in the social marginal cost of production (supply curve) and the demand for the marketed product. Figure 11.2 illustrates the measurement of a positive environmental effect as an additional supply shift resulting from the reduction in environmental damage or positive environmental effects caused by a specific option. In this case, cost-reducing research will shift the supply curve further from S_1 to S_2 thereby reducing the marginal cost by '*ec*'. The total cost reduction effect is represented by the sum of the supply shift due to cost reduction of the technology and a further shift caused by environmental effects. Thus, marginal environmental benefits are accounted for in the total unit cost reduction that is estimated as $ac = ae + ec$. This process adjusts the benefit calculations for implicit price changes. If, however, the effect of the resource management technology is negative, the supply curve S_2 shifts backwards reflecting the environmental damage and corresponding increase in cost. The following section details the analysis of market and non-market impacts of GNPT.

Research cost

Data on research costs can be based on project report documents and historical evidence, as well as on interviews and discussions with the scientists and extension staff who were directly involved in conducting research, on-farm trials, and technology transfer activities. The annual cost of developing and packaging the GNPT, plus the cost of its diffusion and dissemination were estimated by using the formula:

$$GNPTRC = C_{ic} + C_{nars} + C_{ext} \qquad (4)$$

where *GNPTRC* is the annual research and technology transfer cost of all components; C_{ic} is the annual research and overhead costs incurred at ICRISAT; C_{nars} is the annual research and other costs at the NARS; and C_{ext} is the annual cost of extension incurred by the technology transfer department of NARS.

Evaluation of Economic and Environmental Benefits

Farm-level benefits of the GNPT: quantitative estimates

Accounts of actual on-farm practices by representative farmers derived from the sample survey gave estimates of the benefits realised by farmers that include yield gains, cost saving and higher incomes (source: survey data of 1994/95 crop season):

1. The average groundnut yield among adopters was 2.2 t/ha, an increase of about 38% over the 1.6 t/ha among non-adopters
2. The unit variable cost of groundnut production under improved management was Rs3.86/kg in compared to Rs4.58/kg under local practices, a saving of about 16%; and
3. Net incomes among adopters averaged Rs21,470/ha in contrast to Rs15,580/ha among non-adopters, a gain of about 38% for the adopters.

Note that these estimates were obtained without accounting for the possibility of selection bias, an aspect that warrants consideration in future research.

On-farm trial data also provide estimates of the yield gain and unit cost reduction effects of GNPT. The value of the unit cost reduction is summarised for the three subsets chosen for this analysis based on on-farm trials detailed in Table 11.3:

a. k_1 = Rs1,198/t is achieved with the improved GNPT package (including improved varieties, RBF method, and other management practices), compared with the local package (full adoption)
b. k_2 = Rs564/t is achieved with the improved package of management practices compared with the traditional management package (with use of improved varieties in both cases), i.e. partial adoption involving management practices only, holding the effect of improved varieties.
c. k_3 = Rs270/t comparing the effects of RBF with the flat land method of groundnut production, keeping the remaining components of the improved technology at their recommended level (one component). This estimate is assumed to measure the unit cost reduction due to RBF.

Table 11.3. Cost of production and yield of groundnut under on-farm trials with different technology options, Maharashtra, India, 1987–91 (adapted from Pawar *et al.,* 1993).

Technology components		Yield	Cost	Unit cost
Management	Variety	(t /ha)	(Rs/ha)	(Rs/t)
Improved	Improved	3.49	6990	2002.86
Improved	Local	1.97	5990	3040.61
Local	Improved	2.56	6570	2566.40
Local	Local	1.74	5570	3201.15

By the nature of the measurable market effects listed above, the total value of the supply shift is only partially accounted for by taking these estimates of unit cost saving from adoption of the GNPT package instead of the existing practice.

Benefits as described by farmers in surveys and focused group interviews

Farmers described the additional benefits in a pilot survey (1999–2000), participatory rural appraisals and focus group interviews (Box 11.1).

Box 11.1. Welfare changes due to the adoption of groundnut production technology (GNPT) components, based on farm survey, participatory rural appraisals and focus group meetings (Bantilan *et al.*, 2003).

1. Raised-bed and furrow land configuration (RBF) improved soil moisture conservation (75% of survey respondents).
2. RBF was perceived to improve field drainage (75% of survey respondents).
3. RBF saved nutrients and water (28% of survey respondents).
4. Reinvestment in agricultural implements and inputs brought long-term stability to the farming system in the villages.
5. Stability of the farming system increased farmers' options in making decisions about cropping pattern (cash vs. subsistence crops) or investing in production vs. investing in schooling, housing, household assets.
6. The GNPT options were observed to have spillover effects beyond groundnut production. The RBF method was found applicable to such other crops as chillies, soybean, pigeonpea, chickpea, sunflower, mustard and some vegetables. Application of micronutrients to selected crops was also becoming popular where farmers had learned about the GNPT package.
7. Assets acquired for GNPT are being used for other crops, and have enabled cultivation in other seasons.
8. The community has become more socially inclusive, with greater interaction between members of different social categories. Respondents attributed this to a direct consequence of GNPT adoption, as it made landowner farmers more dependent on tribal and landless labour for longer periods throughout the year.
9. Credit rating of the village has risen.
10. Due to the newly found visibility conferred by GNPT adoption successes, the Maharashtra Government targeted the village for special development programmes (e.g. rural sanitation, wasteland development, integrated mother and child development).
11. Empowerment – a general improvement in self-esteem, confidence, ability to innovate were expressed in an increased diversity of crops cultivated, greater choice of investments, and greater access to credit, information, and government agents.
12. Higher pod yields with GNPT generated on-farm employment in shelling, especially for women. The overall labour requirement was about 12% higher with the GNPT than with the existing local practices.
13. For the marginalised groups (tribals and landless labourers), year-round employment ensured adequate food and nutrition for all members of the household.
14. Increased labour demand replaced out-migration of labour by in-migration.

Delineating market and non-market impacts

Table 11.4 summarises the overall impacts of GNPT adoption and delineates the market and non-market impacts in columns 2, 3 and 4. Yield-increasing or cost-reducing benefits cited in column 2 can be measured and directly included in the economic surplus calculations. Quantifiable measurements of these indicators give an initial basis for estimating the parallel *k*-shift in the supply function.

Table 11.4. Analysis of market and non-market impacts of groundnut production technology (GNPT).

Component	Market impacts	Non-market impacts	Environmental effects
C1 Land management			
RBF seedbed	Yield gains Saves 20% of input cost compared to conventional flat system	• Agricultural sustainability • Reduces soil erosion • Reduces water logging • Helps move salts to furrows, and from furrows to drains • Conserves soil moisture during deficit rain • Concentrates organic matter and fertiliser application • Reduces soil compaction, providing loose and well-aerated soil for growing crop • More soil depth for better development of root mass	+ (Greater yield stability, increased water availability off-site and in future, enable cultivation in other seasons) – (Off-site increase in soil salinity)
	Change in labour demand	• More labour required • Reduces drudgery for women in weeding operations (labourers sit in furrows and weed) • Efficient use of tractor and field machinery; interculturing with tractor/bullock implements • Less power requirement for land preparation in successive years	
C2 Nutrient management			
Farmyard manure	Increase in groundnut yields	Improves soil physical properties and soil health	+ (Increase carbon content)

Continued

Table 11.4 Continued.

Component	Market impacts	Non-market impacts	Environmental effects
Ammonium sulphate	Increase in groundnut yields	Environmental effects	+ (Checks soil alkalinity) – (Causes water pollution)
Single super-phosphate	Increase in groundnut yields	Environmental effects	+, –
Zinc sulphate	Increase in groundnut yields	Environmental effects	+, –
Ferrous sulphate	Increase in groundnut yields	Environmental effects	+, –
Gypsum	Increase in groundnut yields	Environmental effects	+, –
C3 Water management			
Sprinkler irrigation	Reduced unit cost due to enhanced water use efficiency	Positive environmental effects due to reduced pest incidence Efficient water utilisation through GNPT offers potential long-term benefits, particularly in increasing water availability off-site and in the future	+
C4 Disease and pest management			
Fungicidal seed treatment	Good quality seeds reduce yield loss and increase employment potential		+, –
Herbicides and pesticides	Reduced yield losses	Negative health effects Adverse effects on water quality	– (Skin allergies)
C5 Seed			
Improved variety	Increase in yields	Conserves biodiversity, checks insect pest incidence	+
Seed rate	Increase in yields	Check insect pest infestation	+
Sowing–dibbling	Yield increase due to good and uniform plant population Increase in employment	Increase drudgery on women	–
Seed dressing	Increased yield	Check insect pest infestation	+

Some non-market impacts may also be indirectly reflected in the calculation of economic benefits to the extent that they affect improvement in yields or unit cost reduction. For example, improvement in the soil physical properties listed in column 3 may be reflected in enhancing groundnut yields. But, there are some indirect or long-term benefits that are difficult to measure as shown in columns 3 and 4 of Table 11.4. These include agricultural sustainability resulting from enhanced biodiversity and health effects. Ideally the value of these impacts can be obtained by seeking appropriate relationships between a chosen GNPT technological intervention and environmental effects. Finding a unique equation or a functional relationship that can be used to quantify, in physical terms, the effect on human health or air quality or other environmental impacts of each component could be difficult. For example, while soil health is believed to improve with the GNPT's land and nutrient management interventions, there are no data or models to measure the specific effects on soil health (J.V.D.K. Kumar Rao, personal communication, 2004). Nevertheless, descriptions of the likely environmental effects of GNPT interventions by Pawar *et al.* could help in impact assessment (1993; and C.S. Pawar, personal communication, 2004):

- The natural acidity of ammonium sulphate checks the alkalinity of the soil. This is a positive effect in alkaline soils, but excess applications of ammonium sulphate can also result in negative environmental effects
- Pollution levels are high with local practices of fertiliser application
- Water quality can be reduced when excess nitrogen is applied to crops
- Micronutrients like zinc sulphate and ferrous sulphate help maintain the yield potential of the soil. Zinc sulphate is used to rectify the zinc deficiencies of the crop. Ferrous sulphate is used to rectify iron deficiencies incurred by waterlogging
- Herbicides and pesticides, if used in large quantities, can cause severe damage to the environment; exposure can also trigger skin allergies in farmers. Prior to the introduction of GNPT, farmers applied excess quantities of pesticides due to lack of awareness. ICRISAT educated the farmers about appropriate dosages and safe handling procedures, thereby mitigating negative environmental effects and farmer health risks.

Listing the positive and negative effects, in Table 11.4, aids in the analysis of market and non-market impacts of the GNPT management options. It records the market impacts representing yield gains or reduced yield losses and changes in unit cost from adoption of GNPT components. The inventory of non-market effects is substantial. The RBF land management appears to have had significant positive environmental effects resulting to greater long-term yield stability, increased water availability off-site and in the future. Agricultural sustainability was enhanced through reduced soil erosion and reduced waterlogging during periods of heavy rain. The other components including nutrient management, disease and pest management and water management improved the soil physical properties and soil health. The environmental benefits included increased carbon content and checked soil alkalinity. Negative effects (environmental costs) from water pollution arose from the use of ammonium sulphate and other micronutrients and pesticide runoff. When pesticide use exceeded recommended levels, it also caused adverse health effects. Finally, although increased groundnut yields increased incomes, denser planting and groundnut shelling created added drudgery for women.

Table 11.4 illustrates how a qualitative understanding of the nature and direction of the impacts can provide a basis for determining the range of possible conditions that would simulate potential benefit levels. In this case it is important to understand the source of the impact, the nature of an impact, and the relationship between an impact and those variables that can affect current, potential, or future producers and consumers. Even though the effects on the environment are complex, the identification and understanding of GNPT effects narrows the field remaining for evaluation. Table 11.4 highlights how the conventional calculations that exclude environmental effects can skew measures of the full technology impact.

Approximations of Economic and Environmental Effects

This section applies the approach discussed above to estimate the total gains due to GNPT technology. Estimates of basic parameters are explained and procedures are illustrated.

Production, price and elasticities

a. The annual base level of groundnut production was 151,280 t in the four selected districts of Maharashtra (average during 1988–1990; source: ICRISAT District-Level Database)
b. The base groundnut price was Rs6533/t (average groundnut price in Maharashtra during 1988–1990, source: ICRISAT District-Level Database)
c. The price elasticity of demand was 0.5 and price elasticity of supply was 0.1 (Radhakrishna and Ravi, 1990).

Research lags

On the estimation of the research lag (i.e. the period of investment required before benefits were realised), the survey indicated that GNPT adoption first took place in 1989. A research lag of 12 years was measured from the time of initial research started in 1974 to the introduction of the technology in farmers' field in 1986 and a further lag of 3 years before first year of actual adoption.

Adoption estimates

Using the methodology introduced in the earlier section on adoption, the survey data covering the period 1989–1994 were used to develop the adoption pathway for GNPT (Fig. 11.3). The results above confirm the situations of partial adoption and step-wise adoption. They indicate that different technology components of GNPT are adopted in a step-wise process of adopting improved varieties, nutrient management, soil management, and other components of the package depending upon: 1. information about the technology, 2. the availability of necessary resources or inputs, 3. marginal returns to the technology, 4. risks, and 5. the suitability of technology traits.

The logistic function was used to estimate the adoption curve and predict the future path, e.g.:

$$A_t = \frac{40}{(1 + e - ^{(-2.6 + 0.69t)})} \quad \text{for adoption of at least RBF} \quad (5)$$

$$A_t = \frac{98}{(1 + e - ^{(-3.2 + 0.34t)})} \quad \text{for adoption of at least improved varieties} \quad (6)$$

Similar estimates can also be obtained for any selected component or subset of GNPT.

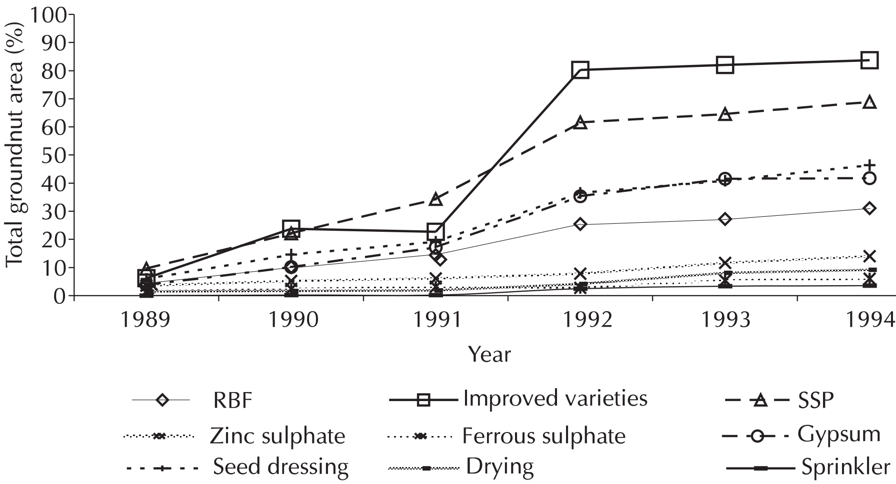

Fig. 11.3. Adoption patterns of groundnut production technology (GNPT) components in Maharashtra.

Figure 11.4 depicts the adoption path for the RBF component, estimated using the logistic function, showing a consistent increase in adoption of the RBF. Because this adoption path reflects those households adopting RBF (some of whom did not adopt other GNPT components), it overestimates adoption of the full package.

Farmers who adopted the concept of RBF but lacked appropriate implements did not strictly adhere to making beds 1.5-m wide. This illustrated an important dimension of crop and resource management technologies: farmers adapt technologies to meet special needs, changing the technologies in the process.

Among the other GNPT components, the adoption rate of improved groundnut varieties rose dramatically from 6% in 1989 to 84% in 1994. The adjusted rate of adoption of improved varieties was higher for those farmers practising the RBF method. The accelerated adoption of improved varieties may be attributed to the dissemination of information on GNPT. At the time of the survey in 1994 the sprinkler method of irrigation was yet to be

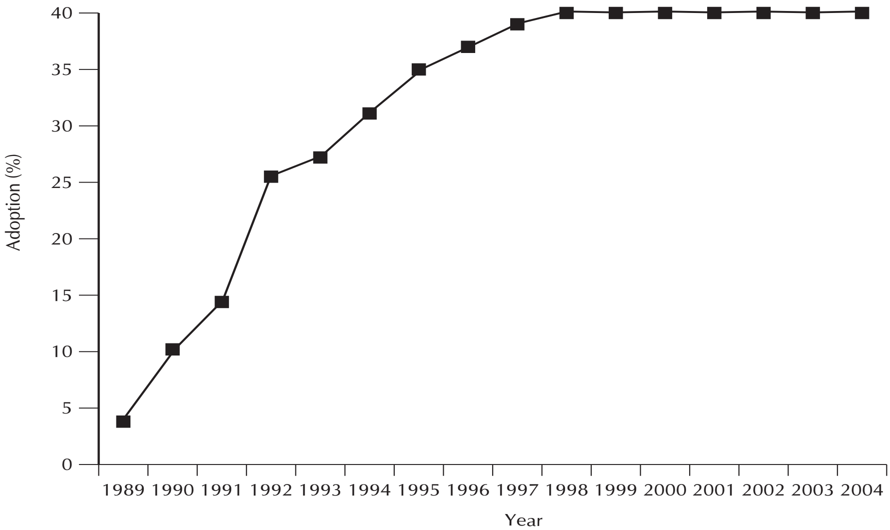

Fig. 11.4. Adoption of raised-bed and furrow (RBF) of groundnut production technology (GNPT) in selected districts of Maharashtra, India, 1989–1995 (projected to 2004).

adopted by the majority of groundnut cultivators. By the late 1990s, the use of sprinkler irrigation in Maharashtra had been substantially enhanced by government subsidies.

Research cost estimates

The estimated cost of research and technology transfer is detailed in Table 11.5. The annual cost of ICRISAT, C_{ic}, was estimated as:

$$C_{ic} = SAL_{ic} + OPR_{ic} + OVR_{ic} + OFD_{ic} \tag{7}$$

where SAL_{ic} is the annual salary of the research team; OPR_{ic} is the annual operational expenses required to undertake GNPT development, packaging, and diffusion; OVR_{ic} is the annual overhead cost at the Institute; and OFD_{ic} is the annual cost incurred to conduct on-farm trials and demonstrations in farmers' fields.

The salary of the research team at ICRISAT, SAL_{ic}, is considered to include the salaries of all those associated with the research project (SAL_i), each weighted by the proportion (w_i) of their time devoted to developing and packaging the GNPT, that is,

$$SAL_{ic} = \sum_{i=1}^{n} w_i * SAL_I \tag{8}$$

This annual salary cost was estimated at US\$34,900. The operational cost (OPR_{ic} = US\$12,215) of developing and packaging the GNPT was assumed to be 35% of the salary. This assumption is based on historical norms at ICRISAT. The overhead costs (OVR_{ic}) are usually considered to be half of the research expenses (Byerlee, 1996); this figure (US\$47,115) was based on research resource allocations to different research projects at ICRISAT. Since the technology components were packaged and recommended for groundnut, pigeonpea, and chickpea, the research and packaging costs for GNPT was proportionately distributed. The share of groundnut in the total area of the three crops was used as a basis for allocating research costs to GNPT (US$45,600).

Table 11.5. Annual research and technology transfer cost (US$) of groundnut production technology (GNPT), 1974–2000.

Component	Year	Cost (US$)
Research		
Salary	1974–86	34,900
Operations	1974–86	12,215
Overheads	1974–86	47,115
NARS	1974–91	9,500
Technology transfer		
Packaging/on-farm trials	1987	24,000
On-farm trials	1988–90	20,000
On-farm trials	1991	10,000
State expenses	1992–2000	7,500

The NARS was involved in packaging the technology and conducting on-farm trials. To assess this cost, several researchers who worked for the NARS were consulted. It was determined that, on the basis of NARS participation in the development and packaging of the technology, the NARS incurred a cost of about US$4560 (approximately 10% of ICRISAT's total cost). Similarly the cost of on-farm research and technology transfer activities (OFD_{ic}) undertaken through the LEGOFTEN Technology Transfer Network, which started in 1987, was proportionately allocated. The expenses incurred in technology transfer (C_{ext}) through the Maharashtra Department of Agriculture during the post-LEGOFTEN period were calculated using the share of groundnut in total area in the State as no separate documentation exists on resource allocation for each commodity or technology.

The technology packaging and its transfer started from 1987 through the LEGOFTEN programme. The initial budget for this programme (1987 and 1988) was met through ICRISAT's core funds, and later (1989–1991) through financial assistance from the International Fund for Agricultural Development (IFAD). In the first year, when different components of technology were integrated, the cost of GNPT (US$24,000) was computed on the basis of the proportionate area under groundnut. In subsequent years, the total budget allocated to LEGOFTEN was distributed (US$20,000) to represent the GNPT package that was apportioned according to the number of on-farm trials conducted on groundnut. The budget of the State Department of Agriculture for GNPT extension activities during 1987–1991 was also met through the LEGOFTEN programme. The expenses incurred in technology transfer through the state departments of agriculture during the post-LEGOFTEN period were calculated using the share of groundnut in the total cropped area in the state, as no separate information on resource allocation to each commodity/technology is documented. On the basis of the salary, operations, and overheads, the annual technology transfer cost during the post-LEGOFTEN period was calculated to be US$7,500. This cost was considered from 1992 until 2000. Since the research and technology transfer costs incurred by ICRISAT, NARS, and the state departments of agriculture were rough estimates based on available ICRISAT Annual Reports and interviews with scientists involved in the project, a sensitivity analysis was also performed by increasing the cost of research and technology transfer by 10–20%. The results revealed that the internal rate of return (IRR) is rather insensitive to changes in costs of research and technology transfer.

Supply shift

The unit cost of production (Rs/ton) was calculated based on total input cost and corresponding yield levels. Pairwise comparison of the unit cost incurred by GNPT enhanced options vs. the traditional practice generated supply shift estimates for each scenario. For the three scenarios described in the previous section, three levels of on-farm unit cost reduction were taken: k_1 = Rs1,198/t, k_2 = Rs564/t, and k_3 = Rs270/t.

Table 11.6 presents the stream of research and technology transfer costs and market-based research benefits using the unit cost reduction estimates (k_1, k_2 and k_3) above, levels of adoption represented by Fig. 11.4, price, quantity and elasticity estimates. It also gives the estimated net present value, IRR, and benefit–cost ratio under three different scenarios. As noted earlier, the estimate using the adoption path for RBF gives an upper bound of the benefit levels. (A lower bound can be estimated using the adoption pathway of the GNPT component that has been adopted least, i.e. at a ceiling level of 15% based on the data.)

The analysis revealed that the IRR of GNPT was 25.3% if the total package of the GNPT is adopted. The total net present value of information from the research and technology transfer programme on GNPT was estimated to be US$3.45 million. The benefit–cost ratio was 9.37, which means that every US$1 invested in developing and disseminating GNPT produced an average benefit of US$9.37 throughout the period.

Given the environmental effects recorded from the analysis above (largely positive but also partially negative), two different scenarios of positive and negative net environmental effects were simulated. Because the major impacts were felt to be captured by the effects on marketable crop yields, the sensitivity analysis scenarios involved modest levels of change: a 10% increase in unit cost reduction from the base level of full GNPT package adoption, and a 5% decrease in unit cost reduction from the base level. The analysis revealed that positive environmental effects that might further increase the unit cost reduction could result in a benefit–cost ratio of 9.73 and an IRR of 26.17. The second scenario of a negative environmental effect by a marginal rate of 5% could reduce the benefit–cost ratio to 8.26 and result in reducing the IRR to 24.95. Negative environmental effects would have to increase the social value of unit production costs by 79% for the benefit–cost ratio to fall to the break-even level of 1.0. Such an increase in units costs is implausibly high, given the dominantly beneficial environmental effects reported by farmers and focus groups. None the less, these simulations show the sensitivity of research impacts when environmental effects are considered.

Table 11.6. Market-based cost and benefit streams for research and technology transfer of the groundnut production technology (GNPT) package.

	Cost (US$'000)		Benefits (US$'000)		
Year	ICRISAT	NARS	Full package	Partial package[a]	Land mgt (RBF)[b]
1974	45.6	4.56	0	0	0
1975	45.6	4.56	0	0	0
1976	45.6	4.56	0	0	0
1977	45.6	4.56	0	0	0
1978	45.6	4.56	0	0	0
1979	45.6	4.56	0	0	0
1980	45.6	4.56	0	0	0
1981	45.6	4.56	0	0	0
1982	45.6	4.56	0	0	0
1983	45.6	4.56	0	0	0
1984	45.6	4.56	0	0	0
1985	45.6	4.56	0	0	0
1986	24.0	4.56	0	0	0
1987	20.0	4.56	0	0	0
1988	20.0	4.56	0	0	0
1989	20.0	4.56	162.57	76.15	36.42
1990	10.0	4.56	460.62	215.75	103.19
1991	0.0	7.50	650.29	304.59	145.68
1992	0.0	7.50	1,151.56	539.39	257.97
1993	0.0	7.50	1,228.33	575.34	275.17
1994	0.0	7.50	1,404.45	657.84	314.63
1995	0.0	7.50	1,580.57	740.33	354.08
1996	0.0	7.50	1,670.89	782.64	374.31
1997	0.0	7.50	1,761.21	824.94	394.54
1998	0.0	7.50	1,806.37	846.09	404.66
1999	0.0	7.50	1,806.37	846.09	404.66
2000	0.0	7.50	1,806.37	846.09	404.66
2001	0.0	0.00	1,806.37	846.09	404.66
2002	0.0	0.00	1,806.37	846.09	404.66
2003	0.0	0.00	1,806.37	846.09	404.66
2004	0.0	0.00	1,806.37	846.09	404.66
2005	0.0	0.00	1,806.37	846.09	404.66
Internal rate of return (IRR) (%)			25.26	19.15	13.50
Net present value (US$ '000)			3,452.94	1,389.06	453.45
Benefit–cost ratio			9.37	4.39	2.10

[a]Partial = management practices only.
[b]Land mgt (RBF) = raised-bed and furrow.

Summary and Conclusions

This chapter principally illustrates an empirical estimate of economic surplus using the case of GNPT developed by ICRISAT and its partners in the Indian NARS. The case study illustrates the critical importance and use of qualitative information in understanding the additional environmental and long-term effects due to the adoption of NRM technologies.

To quantify the returns to investment on research and technology exchange, three aspects were examined:
1. Benefits (both economic and environmental) accruing from the research and technology exchange programme
2. Adoption rates and the spread of different components of GNPT
3. Research and technology exchange cost involving research partnerships among international and national research programmes as the extension sector.

Economic surplus and distribution of welfare gains were estimated by assuming a parallel shift in supply function due to investment in the research and technology development. Internal rates of return, net present values and benefit–cost ratios were computed under three options:
1. Full adoption of the GNPT package
2. Adoption of only management practices
3. Adoption of only land management (RBF) with other practices remaining the same.

Because environmental effects were not measured in monetary terms, two sensitivity analyses were carried out under scenarios related to net positive and negative environmental effects.

The survey results show that farmers initially adopted parts of the crop and resource management package, and adapted the technology options according to their needs, convenience, and resource endowments. Logistic growth functions were estimated to describe the rate of adoption of each GNPT component. The adoption analysis illustrates the nature and dynamics of adoption of NRM technologies.

The estimation of benefits accruing from GNPT involved computation of welfare gains based on yield gains and/or reduction in unit production costs. The inclusion of qualitative environmental effects encompassed impact dimensions not captured via the measurable reduction in unit cost or yield gains due to lack of quantifiable or long-term data. The difficulty of quantifying many environmental costs and benefits challenged the approach to incorporating these effects into cost–benefit analysis. The environmental effects were characterised by systematically tracking both individual and interaction effects of GNPT components. Thorough analysis is based on systematic documentation coupled with reasonable estimates of economic effects.

Environmental effects can have a large overall impact. The results show that if environmental effects reduced fully accounted unit costs by just 10% more than market effects, the net present value of the GNPT would increase by US$0.4 million and the IRR would increase by 1%. Clearly, environmental effects in the assessment of NRM options cannot be ignored. As stated by Winpenny (1991), the environment is not free, even though there may not be a conventional market for its services. In the context of decisions based on cost–benefit analysis, it is important to understand the source of the impact, the nature of an impact, and the relationship between an impact and those variables that can affect current, potential, and future consumers and producers. This means that valuing as many effects as possible and plausible, narrows the field remaining for pure judgement.

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12 Assessing the Economic and Environmental Impacts of Conservation Technologies: A Farm-level Bioeconomic Modelling Approach

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Introduction

Along with degradation of the productive resource base, widespread poverty and population growth are major concerns for sustainable intensification and agricultural development in many poor regions of the world. The relationship between population pressure, poverty and environmental degradation has been a subject of debate and controversies for many years, with an upswing in the debate over the last 30 years (Boserup, 1965; Cleaver and Schreiber, 1994; Tiffen *et al*., 1994; Grepperud, 1996; Templeton and Scherr, 1999). Earlier studies on technology choice among smallholders in Ethiopia found that low or negative initial returns to conservation technologies could undermine investments in such practices (Shiferaw and Holden, 1998). Some evidence indicated that population pressure, poverty and land scarcity may even encourage removal of conservation structures (that occupy productive lands) introduced in the past through food-for-work programmes.

Although the empirical evidence is mixed and less conclusive (Templeton and Scherr, 1999), there is emerging consensus on the potential nexus between population density, poverty and land degradation in some less-favoured areas where poverty is associated with poor policies, and lack of access to markets and improved technologies (Reardon and Vosti, 1995; Heath and Binswanger, 1996). Under such conditions, poor land users often lack the wherewithal to invest in sustainability-enhancing options and could be caught up in a self-reinforcing nexus that may lead to worsening poverty and resource degradation. However, significant research and development effort is directed towards evolving options for improved natural resource management (NRM) to enhance the productivity and sustainability of production systems. Nevertheless, the basic question remains about the potential of technological and policy options to lift the poor out of poverty, and to what extent these options could

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actually contribute to sustaining the resource base and livelihoods under conditions of high population density and high risk of land degradation.

Bioeconomic models are suited to evaluating the potential impact of new technologies and policy options on rural livelihoods and the environment (resource conditions) at different temporal and spatial scales (Holden, Chapter 8, this volume). The integration of biophysical and socio-economic conditions into the local economy is an enhancement of earlier econometric approaches, since it allows more-precise simulation of household investment decisions and simultaneous assessment of the welfare and environmental impacts in a dynamic setting – a more suitable approach to assessing NRM impacts. The objective of this chapter is to illustrate how a multiperiod bioeconomic household-level model, in which changes in resource quality have feedback effects on future land productivity, can be used to explore the economic and environmental impacts of NRM technologies and policies. This model is used to test the influence of land scarcity and asset poverty (e.g. oxen and labour) on incentives to undertake sustainability investments. The integration of agroecological and socio-economic information has provided useful insights regarding the potential of alternative policy instruments and the impacts of new technologies. The model incorporates important features of the biophysical system and its dynamics along with market characteristics in the rural economy. The choice of crop and livestock production activities and NRM technology investments are jointly determined. The model is developed in Generalised Algebraic Modelling System (GAMS) using data from Andit Tid, in the central highlands of Ethiopia, an area inhabited by poor smallholder farmers and characterised by high population density, rugged topography, steep slopes, and severe problems of soil degradation.

The results show how land scarcity could drive conservation investments, while poverty in vital assets such as oxen and labour could deter investments in land and water management. The welfare and environmental impacts are very modest but are highest when the conservation technology does not reduce short-term crop yields. Otherwise, the level of adoption of these technologies and their effects on poverty and soil degradation are significantly reduced even when family labour is not limiting. This contributes to worsening the conditions of the poor and continued degradation of the resource base. For credit-constrained households the increased fertiliser use associated with improved credit availability may substitute for conservation effort. The following part of the chapter offers an overview of the case study area and important biophysical and socio-economic aspects included in the model, then the basic structure of the bioeconomic model is presented. This is followed by presentation and discussion of the simulation results. The final part highlights the major findings and policy implications.

The Biophysical and Socio-economic System

The study area (Andit Tid) is located in North Shewa, in the central highlands of Ethiopia, approximately 60 km north of Debre Berhan, along the main road from Addis Ababa. This implies that market access is fairly good. The area is characterised as a low-potential, cereal–livestock zone and suffers from severe soil degradation. Given the high altitudes, the land falls in two altitude zones: *Dega* zone (<3200 m asl) and *Wurch* zone (>3200 m asl). There are two distinct rainfall and growing seasons, the *Meher* (June–December, 1056.8 mm rainfall), and the *Belg* season (January–May, 315.4 mm rainfall).

Barley is the main crop, followed by wheat, horse bean, and field peas. Lentils and linseeds are also commonly grown. The cropping pattern depends on the local agroclimatic zone (see Table 12.1). Crop production mainly depends on organic fertilisers, while the use of mineral fertilisers is limited by lack of credit and the low profitability of applying it to some crops. Most of the production takes place in the low altitude zone but barley is grown also in the higher altitude zone in the *Belg* season. The major crops during the main growing season (the *Meher*) are barley, wheat, faba beans, field peas and lentils, in the low-altitude zone. In the *Belg* season, barley is grown in the high altitude zone, and lentils and field peas in the low altitude zone. Droughts are not common during the *Meher* season but can occur in the *Belg* season. Hailstorms and frost may damage crops during the *Meher* season.

Table 12.1. Crops grown in the different seasons and local agroclimatic zones.

	Cropping zone	
Season	Low altitude	High altitude
Main season (*Meher*)	Barley, wheat, faba beans, field peas, lentils, linseed	Fallow
Short-rainy season (*Belg*)	Field peas, lentils	Barley

The two dominant soil types are Andosols and Regosols. Andosols are dominant in the high-altitude zone while Regosols are common in the lower-lying areas. The Regosols are the most important and intensively cultivated soils. Andosols are mainly used to grow barley and are relatively rich in organic matter. Steep slopes and intensive cultivation increase the risk of soil degradation. An estimated 75% of the land area is steeply sloped (>25%). Soil erosion rates are very high and an estimated 21% of the agricultural land has shallow soils (<30 cm) and 48% medium-deep (30–60 cm) soil (Yohannes, 1989).

Cattle and sheep are the predominant types of livestock but goats, equines and chickens are also common. There are strong crop–livestock interactions in the system. Crop residues are typically used as animal fodder. Oxen provide traction power to cultivate land and thresh crops. Animal manure is used to enhance soil fertility and for fuel. Fodder shortage is a constraint to livestock production. High population density and land scarcity increase competition between crop and livestock production. Sale of small stock (sheep, goats and chickens) complements both household consumption and crop–production activities.

Some conservation technologies were introduced through food-for-work programmes in the early 1980s. With the termination of programme benefits in the early 1990s, farmers have been selectively removing soil conservation structures from their plots (Shiferaw and Holden, 1998). The removal seems to be accelerated when structures occupy productive land and increase land scarcity, or when they do not contribute to increasing short-term yields. How poverty affects this process and the potential economic and environmental impacts from such NRM investments are not well understood. Farm households possess usufruct rights to land. Following the land reforms of 1975 and frequent land redistributions thereafter, landlessness is uncommon, and land is fairly distributed according to family size (see Table 12.2). This means that livestock wealth is often a better indicator of household wealth and wealth differentiation. The oxen rental market is underdeveloped (Holden and Shiferaw, 2004) and ownership of traction power is an important asset that determines crop income. When the necessary traction power is lacking and rental markets are imperfect, land ownership by itself may not necessarily translate into better living conditions for the household. Typically, households lacking traction power either rent out land to households with two or more oxen, or depend on relatives with oxen to cultivate their lands. Hence, local communities often use oxen ownership as a wealth indicator. Therefore oxen ownership along with farmland and family labour endowments were used as proxy indicators for household poverty. Future work will need to extend this through use of other more-relevant poverty indicators.

Production remains largely subsistence based. The small towns in the vicinity, inhabited mainly by local traders, serve as markets in the area. Owing to the difficult terrain, there are significant transportation costs to some of the accessible output markets in the area. Farm-gate prices are adjusted for such transaction costs based on local estimates. Some markets for labour, land and livestock exist within the village or in the nearby towns. The labour market is largely inactive, but the local wage rate varies seasonally depending on local demand. Labour may be hired in cash, in kind (fixed output share) or in exchange for traction power. Formal institutional credit is largely unavailable. Hence, the basic model does not include credit, but this assumption was relaxed to assess the effect of credit policy on welfare and sustainability investments. Off-farm income options are mainly limited to local agricultural wages and self-employment in petty trade within the vicinity.

Along with biophysical and experimental data collected by the Soil Conservation Research Project (SCRP), socio-economic data mainly collected in 1994 and complemented in 1998 were used to formulate and develop the model. The availability of on-site biophysical and socio-economic data made it possible to assess technology and policy impacts using a multi-period bioeconomic model. In 1994, about 26% of the households had no oxen, 15% had one ox, and 56% had two oxen. Less than 5% of households were landless, mainly young families awaiting land allocation by the State. Table 12.2 shows the basic characteristics and resource endowments of the different household groups. For better simulation of the biophysical system and variations in land quality, land was classified into eight different soil depth and slope classes (Table 12.3 and Fig. 12.1).

Table 12.2. Basic farm household characteristics in Andit Tid, 1994.

Variables	Household type[a] No ox	One ox	Two+ oxen	Average
Family size	2.80	5.80	7.20	6.10
Consumer units	2.60	5.17	6.58	5.55
Labour units	1.53	2.78	3.98	3.23
Own farm size (*Timad*)[b]	5.55	7.68	11.00	9.05
Operated crop area	3.30	5.08	8.84	6.73
Own cultivated area	3.00	4.79	7.80	6.07
Rented-in land	0.30	0.28	1.04	0.66
Rented-out land	1.55	0.18	0.10	0.31
Tropical livestock units (TLU)	1.45	3.52	7.10	5.10
Oxen	0	1.00	2.30	1.53

[a] The sample size was 10 households with no oxen, 30 households with one ox, 40 households with two or more oxen.

[b] Land areas are measured in *Timad*, approximately 0.25 ha.

Table 12.3. Land area (in *Timad*) by farm household category, soil type, soil depth and slope classes.

Soil type	Codes	Soil depth class (cm)	Slope class (%)	Household category No ox	One ox	Two+ oxen
Andosols (A)		All	All	2.03	2.82	4.02
	A0–30 cm (1)	0–30	0–20	0.91	1.26	1.80
	A30–60 cm	30–60	0–20	0.57	0.78	1.12
	A>60 cm	>60	0–20	0.32	0.44	0.63
	A0–30 cm (2)	0–30	>20	0.24	0.33	0.48
Regosols (R)		All	All	3.52	4.88	6.98
	R0–30 cm (1)	0–30	0–20	1.62	2.25	3.21
	R30–60 cm	30–60	0–20	0.86	1.19	1.69
	R>60 cm	>60	0–20	0.31	0.44	0.62
	R0–30 cm (2)	0–30	>20	0.73	1.01	1.44

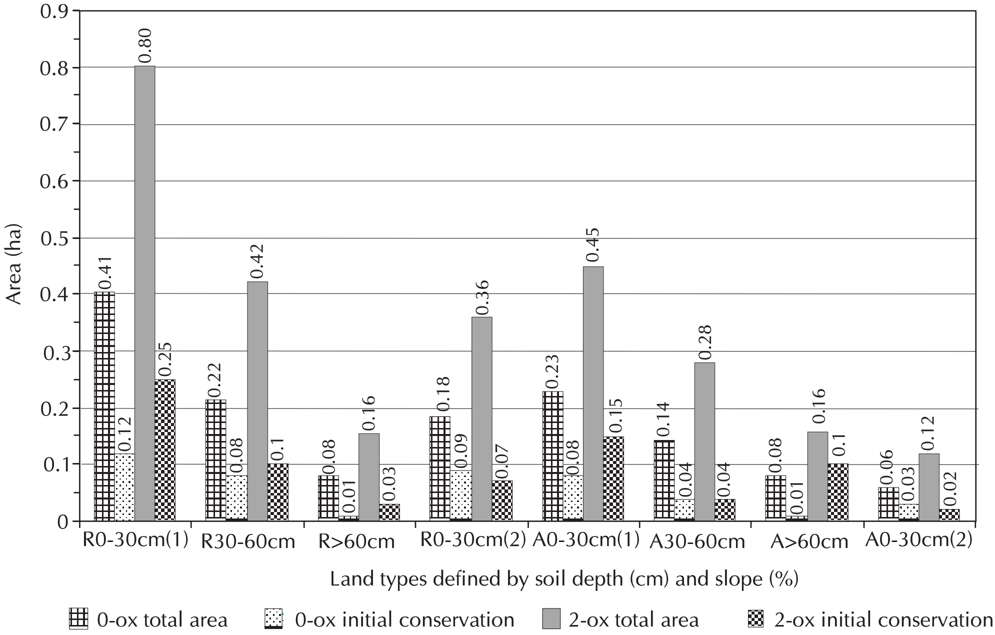

Fig. 12.1. Cultivable land and initial level of conservation by household group and land type (as defined in Table 12.3).

The Bioeconomic Model

Understanding farm households' incentives and constraints to intensification of land use, technology choice and investment behaviour, and analyses of the resulting pathways of development requires integration of biophysical and economic modelling approaches at the household level (Ruben *et al.*, 1998). The bioeconomic model developed here uses a non-separable farm household model (de Janvry *et al.*, 1991) as a basis. Production, consumption and investment decisions are jointly determined in each period. This results from imperfections in input and output markets that introduce divergence between selling and buying prices (price bands). In such situations, decisions are constrained by market imperfections, and household attributes and asset endowments will affect production and investment choices. The on-site costs of soil erosion and nutrient depletion are endogenous in the model and their future land productivity impacts influence the choice of land management practices. Off-site effects are not accounted for, but the model allows quantification of soil erosion and runoff that may also affect water bodies and other farmers in the vicinity.

The farm household maximises the discounted utility (*DU*) subject to resource supply, market access and subsistence consumption constraints:

$$DU = \sum_{t=1}^{T} U_t (1+r_t)^{-(t-1)} \tag{1}$$

The utility function is specified as:

$$U_t = \frac{1-\mu}{\left(\frac{FY_t}{SY_t}\right)^{\mu-1}} + \mu - 1 \qquad (2)$$

where FY_t is the full income of the household (as defined in Equation 4). SY_t is the subsistence (poverty line) level of full income estimated based on the annual poverty line income (Dercon and Krishnan, 1996) of Birr 528 (US$1 = Birr 6 in 1993/94) per consumer unit (CU) and minimum consumption of leisure time in the area. The utility function (U_t) has an elasticity of marginal utility of income (also called flexibility of money) equal to $-\mu$. The curvature of the utility function has a relative risk aversion coefficient equal to μ. The marginal utility of income estimated for different countries ranges from –3 at low levels of per capita income to –1.1 at higher levels (Bieri and de Janvry, 1972). A value of $-\mu = 3$ was used. As defined, the utility function attains a negative value when income is less than subsistence, a zero value when income is just equal to subsistence, and a positive value when income is higher than subsistence consumption. This provides a good indicator of the welfare impacts of conservation investments

Based on Holden *et al*. (1998) the rate of discount r is endogenous in the model and is determined by the level of income and asset endowments:

$$r_t = \frac{FV_{t+1}}{PV_t} - 1 \qquad (3.1)$$

$$PV_t = z + \beta \left\{ \frac{(FY_t - \Sigma_s w_{st} L_{st}^c)}{YCU_t} \right\} \qquad (3.2)$$

where PV_t is the present value equivalent of future income (FV_t) the household is willing to accept instead of waiting for one more year. The PV_t is assumed to be dependent on the level of income per CU in each period (YCU_t); the value of β is determined from an econometric model estimated for farmers in Ethiopia (Holden *et al.*, 1998). The upper and lower bounds of PV_t are estimated based on survey data and the highest and lowest discount rates found for households. Based on average incomes, the value of z is calibrated at levels consistent with the highest and lowest bounds for different household groups. In this way, an increase in household income increases the present value equivalent of future income, and reduces the rate of discount, indicating the household's ability to trade-off current consumption to improve future livelihoods. If the income level falls, the opposite would occur. The effects on technology choice and investments are estimated by solving the model for upper and lower bounds on the discount rates.

Household full income is given by:

$$FY_t = \sum_{g=1}^{G}\sum_{c=1}^{C} A_{cgt}\left\{p_{ct}y_{cgt}\left(x_{cgt}\right)-\sum_{i=1}^{I} e_{icgt}x_{icgt}\right\}+ \sum_{V=1}^{V} L_{vt}\left\{p_{vt}y_{vt}(x_{vt})-\sum_{i=1}^{I} e_{ivt}x_{ivt}\right\}+\sum_{s}^{S} w_{st}(L_{st}^{c}+L_{st}^{of}) \tag{4}$$

A_{cgt} is the area of crop c produced on land type g in year t. L_{vt} is production of units of livestock v in each period. x_t is a vector of inputs used in production of a unit of crop c in land type g and livestock v in year t. p is the per unit price of crops or livestock and e is the per unit input cost. y_{cgt} is the yield function for the production of crop c and y_{vt} is the yield function for livestock v. In year t, family home time (leisure) in each season (s) is L^c_s while w_s is the seasonal reservation wage (after transactions costs). The seasonal off-farm labour supply is L^{of}_s.

Linkages between the economic and biophysical system

The key equations that link the biophysical system with the economic behavioural model are embedded through the production functions that include the effect of changes in soil quality. Change in the soil nutrient stock is the cumulative outcome of positive and negative processes. Use of organic and mineral fertilisers adds soil nutrients, while soil erosion depletes both rooting depth and soil nutrients. The cumulative change in the available nutrient stock affects crop yields in the following years. Depending on the cost of abating soil degradation through conservation and/or fertiliser use, this creates the economic incentive to adopt new sustainability-enhancing practices. The change in the soil nitrogen (N) stock is given by:

$$N_{t+1} + N_t - \delta\left[N_t - \eta\,(SE_{\,t})\right] - \eta\,(SE_{\,t}) \tag{5}$$

where SE_t is the period t rate of soil erosion, δ is the share of soil N mineralised in each period and η is the N composition of the soil. Based on the advice of agronomists, an enrichment ratio of 2 for eroded soil and an annual mineralisation rate of 1% for soil N were used. The change in plant-available soil-N due to soil erosion and nutrient depletion from period to period (dN) is computed as:

$$dN = \delta(N_t - N_{t+1}) \tag{6}$$

where δ is as defined above. The cumulative reduction in plant-available N is included in the production function (Equation 7.2) to influence crop yields in each period. Since incorporating the effect of phosphorus (P) depletion on land productivity requires additional data on P-fixation, conversion of stabile P to labile P, and the total P-stock in the soils, the model currently includes only the effects of depletion of rooting depth and soil-N on crop yields.

Crop yield (y_{cgt}) for crop c on land type g in period t is estimated in two steps. Firstly, the intercept term (y^{int}) representing the depth–yield relationship without fertiliser use was estimated econometrically as a function of soil depth (SD_t) and soil type (ST) based on the SCRP time-series collected at the site (Shiferaw and Holden, 2001). Secondly, responses to N and P were estimated from Food and Agriculture Organization of the United Nations (FAO) fertiliser-response studies (Ho, 1992) and the soil productivity calculator (Aune and Lal, 1995) as a function of fertiliser nutrients and the cumulative change in the available soil-N (dN_t). Hence, the intercept term and the yield function are given as:

$$y_{cgt}^{int} = f(SD_t, ST) \tag{7.1}$$

$$y_{cgt} = f(y_{cgt}^{int}, dN_t, N_t, P_t) \tag{7.2}$$

where N_t and P_t are nitrogen and phosphorus available to plants.

The rate of soil erosion (SE_t), and hence the change in soil depth for each land type, in each period depends on the soil type (ST), slope (SL), rainfall (RF), land management or conservation technology used (K), and the type of crop grown (c):

$$SE_t = f(ST, SL, RF_t, K_t, c_t) \tag{8}$$

The parameters of Equation 8 were obtained from the SCRP experiments at the site or were estimated based on plot-level survey data. In return, soil erosion affects soil depth in each period such that:

$$SD_t = SD_{t-1} - \varphi SE_t \tag{9}$$

where φ is the conversion parameter. Hence, the soil depth trajectory depends on the initial soil depth and the cumulative level of soil erosion. Most of the model *parameters* were exogenously determined. These parameters include input and output prices, wage rates, seasonal working days (excluding religious holidays), population growth rate, activity-wise resource requirements, nutrient content of local foods, and household asset endowments. Given the objective function and a set of resource availability and market constraints, the model determines optimal values of *variables* that represent crop–livestock production, consumption and conservation investments.

Other model variables and constraints

Major activities in the model include production of six crops on eight land types with ten levels of fertiliser use [diammonium phosphate (DAP) and urea]; two land management options; two cropping seasons; consumption, storage and selling of crops; allocation of family labour (over ten seasons) for production, conservation, off-farm employment (constrained) and leisure;

seasonal labour hiring; production, selling and consumption of livestock; buying of agricultural products for consumption; buying of livestock feed (crop residues); and constrained local markets for renting in/out land and oxen. The model constraints include limits that the use and sale of available resources (e.g. land, seeds, labour, fertiliser, oxen power, food, animal feed and liquidity) could not exceed total household endowments:

$$\Phi A_t - X_t^b \leq X_t^w \tag{10.1}$$

$$X_t^s \leq X_t^w - \Phi^w A_t \tag{10.2}$$

where A_t is a vector of the level of activity, Φ is a vector of total and Φ^w owned resource requirement per unit of activity A, X^w is a vector of owned resources, X^b is a vector of bought (hired) resources, and X^s is a vector of sold or out-rented resources. Available resource supplies can be increased through participation in markets (10.1). According to local norms, the model assumes that labour may be hired in cash, in kind (fixed output) or in exchange for traction power. Land can be in-rented in cash or in kind (fixed output), the price depending on its quality. The model also allows in-renting or out-renting of oxen in exchange for labour or cash. When the family resource stock is nil (e.g. fertiliser), all the demand will be met from markets. When markets exist, resources not used in production can also be sold, but the amount used and sold cannot exceed available supplies (10.2). The overall cash and credit constraint is specified as:

$$P_t^b X_t^b + (1+\gamma) X_{t-1}^{cr} - P^s X_t^s \leq X_t^{lq} + X_t^{cr} \tag{11}$$

where P^b is the buying and P^s selling price, X^{lq} liquidity at hand and X^{cr} is the level of credit (with interest rate γ) received during each period. When liquidity is non-existent, all purchases will be financed from available credit and sale of resources (inputs or products). When credit is not available, cash expenditures cannot exceed cash income from sales. The interest and the principal from the credit used in the previous period [$(1+\gamma)X_{t-1}^{cr}$] should be paid back during the next period. Consumption requirements were specified as:

$$\lambda \left[X_t^w + X_t^b\right] \geq \Omega \tag{12}$$

where λ is a vector of nutrient composition of owned (X_t^w) and purchased (X_t^b) foods and Ω is the biologically determined nutritional requirement for carbohydrates, fats and proteins. Households can use markets to meet resource demand (10.1) and consumption requirements (12) but buying activities for inputs and products include a price band of 5–10% over farm-gate selling prices. All purchases are also subject to a cash constraint given in Equation 11. The model also allows for the import of commonly consumed crops not grown in the area. Taste and food diversity constraints reflecting observed consumption choices were also imposed. Consumption of grains could also include savings from previous production. The consumption requirements depend on family size and CUs. The production balance in each year for consumed products is given as:

$$Q_{cons} + Q_{seed} + Q_{sold} + Q_{stored} = Q_{Tot} \tag{13}$$

This indicates that the total production is consumed (Q_{cons}), used as seed (Q_{seed}), sold (Q_{sold}) and/or stored (Q_{stored}) for subsequent periods. Likewise, family labour is allocated to different activities seasonally as follows:

$$L_{st}^{c} = L_{st} - (L_{st}^{f} + L_{st}^{of}) \tag{14}$$

This shows that family labour in year t and season s (L_{st}) is used on-farm (L_{st}^{f}), off-farm (L_{st}^{of}), and the residual consumed as leisure (L_{st}^{c}). Off-farm employment is constrained to average levels estimated from the survey for different household groups. Other constraints include restrictions on crop rotations such that cereals follow land sown to legumes in the previous period. Accounting equations include land, crop and technology-specific soil erosion; cumulative changes in soil depth; and cumulative changes in conservation investments. Changes in the stock of animals was specified for each type as:

$$LV_t = (1-\theta - m)LV_{t-1} + LVR_{t-1} + LV_t^{b} - LV_t^{s} \tag{15}$$

where LV_t is adult livestock in period t, θ is the culling rate, m is the mortality rate, LV_{t-1} is the closing stock in the previous period, LVR_{t-1} is young stock of certain ages in the previous period reared into adult animals in period t and LV^{b} and LV^{s} are animals bought and sold during the period. Production and rearing of young stock is given as:

$$(1-m)kLV_t^{f} = LVR_t + LVR_t^{c} - LVR_t^{s} \tag{16}$$

where LV_t^{f} is female animals of reproductive age, and k is the litter size per reproductive female. The total number of newborns, adjusted by the mortality rates (m), is reared (LVR), consumed (LVR^{c}) or sold (LVR^{s}) within the year. The detailed structure of the model is presented in Holden and Shiferaw (2004).

Scenarios for analysis of technology and policy impacts

The bioeconomic model was used to simulate the adoption and potential impact of two types of land and water management options introduced into the area by the SCRP and the Ministry of Agriculture. These technologies were developed based on graded soil–stone bunds to enhance water infiltration, and drainage of excess water, and to reduce soil erosion. Farmers indicated that the structures occupy productive land and reduce yields in the initial period, especially on steeper slopes. In order to assess how this will affect adoption of these technologies, we specified two stylised versions of the technology. Type I is when the initial effect of area loss from adoption of the conservation methods is negligible and short-term yields are unaffected, and Type II is when loss of productive land and other undesirable effects may reduce initial yields with conservation by 5–10% depending on the slope. The Type I situation may arise if conservation improves soil fertility or raises relative returns to fertiliser use and offsets the negative effect of area loss. The Type II situation may arise when positive effects are negligible or when negative outcomes are dominant. Both are very likely and valid scenarios. Even if Type II conservation has a short-term yield penalty, it could still be attractive in the long term as crop yields exceed those without conservation. The length of time needed for this to occur will depend on the interaction between existing soil depth, the level of soil erosion and the type of crop grown. However, with a positive discount rate, delayed benefits may not create incentives for small-scale farmers to adopt these technologies. The model captures these relationships and impacts on welfare outcomes and the condition of the resource base.

Furthermore, depending on slope, adoption of these technologies is estimated to require 100–120 working days/ha while annual maintenance requires 15–20 days/ha. The model also allows removal of some of the existing conservation structures installed through food-for-work programmes and mandatory polices of the past. Figure 12.1 shows the area of land under different categories and the existing level of conservation in the initial year. Removal is assumed to require 25% of the labour need for construction. The decision to remove will depend on the availability and opportunity cost of family labour, the ability to pay for hired labour, the scarcity of land, and the expected returns from removal or maintenance of the structures. The expected return will in turn depend on the crop grown, the soil type and the slope of the land.

The two variants of the technology (Type I and II) are nested in the model for two household groups: without oxen (poor households), and with a pair of oxen (less-poor households). Since farm and non-farm employment opportunities are limited, it is hypothesised that the relative availability of land and oxen assets will be crucial for household welfare while the relative abundance of family workforce relative to land will contribute to increased conservation investments. In order to capture this complex relationship, each of the two household groups are further disaggregated into two sub-groups depending on the relative endowment of land and labour resources within the household at the initial period. Hence, four modelling scenarios are developed: without oxen and land-scarce, without oxen and land-abundant, with two oxen and

Table 12.4. Household sizes in the selected scenarios at the initial period.

	Households with two oxen		Households without oxen	
	Land-scarce[a]	Land-abundant[b]	Land-scarce[b]	Land-abundant[a]
Family size	7.2	4.2	7.2	2.8
Worker units	4.0	1.5	4.0	1.5
Consumers units (CU)	6.6	3.0	6.6	2.6
Land (Regosols)[c]	6.98	6.98	3.52	3.52
Land (Andosols)	4.02	4.02	2.03	2.03
Total farm size	11.00	11.00	5.55	5.55
Total farm size per capita	1.53	2.62	0.77	1.98

[a] These are average values for the group from the study area.
[b] Labour endowments are adjusted to explore the effect of changes in land–labour ratios.
[c] The land areas are in *Timad* (approximately 0.25 ha).

land-scare, and with two oxen and land-abundant. Table 12.4 shows the major attributes and cumulative asset endowments of these four household groups. The model uses the detailed land classification shown in Table 12.3. The multi-period model, written in GAMS, is solved for t = 5 years. The 5-year model has about 25,700 variables and solves within 1–2 hours using present-day Pentium-4 computers.

Simulation Results

As stated earlier, the bioeconomic model allows a simultaneous evaluation of the level of technology adoption and the associated effects on productivity, human welfare and sustainability. The optimised model provides extensive results on the crop–livestock economy, marketed surplus, conservation investments, consumption levels and changes in soil depth and soil erosion. The main focus here is on adoption of NRM technologies and productivity and environmental impacts. The differential conservation adoption patterns and the resulting livelihood and resource conservation outcomes for the different household groups are discussed. The level of conservation investments is reported for the different land types at varying endogenous rates of discount.

Adoption of NRM technologies

Households with a pair of oxen

Boserup (1965) hypothesised that intensification of land use and investments to enhance land productivity will be limited when land is more abundant than labour. This suggests that labour-scarce families with large farms will have lower incentives to increase the intensity of labour use and other inputs per unit of land to enhance its productivity. This may particularly be the case if land markets are imperfect and surplus land cannot be sold or leased out to others. These simulations also indicate that when land is more abundant than labour, the land users lack sufficient incentives to make significant erosion control investments (see Tables 12.5 and 12.6). The level of investment in conservation and soil fertility management is much larger for land-scarce households than for land-abundant households. When conservation does not incur a short-term yield penalty (Type I), the land-scarce households make significant conservation investments in all land types except the steep slopes that are mainly used for grazing. While labour-scarce households adopt conservation practices on a maximum of one-third of the different land types, the land-scarce (labour-rich) households are able to adopt conservation on up to 97% of the area of some land types (Table 12.5).

Compared to the land-scarce household, the short-term welfare impact of soil degradation in terms of future productivity decline is relatively less for the land-abundant household. Even if soil erosion increases on untreated lands, households with relatively abundant land will have enough land to

Table 12.5. Livelihood and environmental impacts for households with two oxen: when conservation technology does not take land out of production.

Household type	Household welfare	Average net income (Birr/consumer unit (CU))[a]	Adoption of conservation practices (% total area) terminal period: Regosols 0–30 cm (1)	Regosols 30–60 cm	Regosols >60 cm	Regosols 0–30 cm (2)	Andosols 0–30 cm (1)	Andosols 30–60 cm	Andosols >60 cm	Andosols 0–30 cm (2)
Land-abundant[b]	4.521	1305.8	32.4	26	25.8	22.2	2.6	17.9	0	0
Land-scarce[b]	2.076	708.6	78	97.6	93.5	97.2	0	96.4	57.1	0
Land-abundant[c]	2.68	1296.4	32.4	26.0	25.8	22.2	2.6	17.9	0	0
Land-scarce[c]	1.208	702.8	78.1	97.6	93.5	95.9	0	96.4	57.1	0
Land-abundant[d]	2.68	1296.4	100	100	100	100	1.3	100	0	0
Land-scarce[d]	1.208	702.8	70	57	100	100	0	100	80	0

[a] In 1993/94, US$1 = Birr 6. Current rates are about US$1 = Birr 8.6.
[b] Low discount rate (ranges: 0.25 to 0.26 for land-scarce, and 0.21 to 0.22 for labour-scarce households).
[c] High discount rate (ranges: 0.57 to 0.58 for land-scarce, and 0.50 to 0.51 for labour-scarce households).
[d] Percentage of the initial area of treated land maintained at the end of the terminal period (high discount rate).

Table 12.6. Livelihood and environmental impacts for households with two oxen: when conservation takes 5–10% land out of production.

Household type	Household welfare	Average net income (Birr/consumer unit (CU))[a]	Adoption of conservation practices (% total area) terminal period: Regosols 0–30 cm (1)	Regosols 30–60 cm	Regosols >60 cm	Regosols 0–30 cm (2)	Andosols 0–30 cm (1)	Andosols 30–60 cm	Andosols >60 cm	Andosols 0–30 cm (2)
Land-abundant[b]	4.511	1304.0	0	0	0	0	0	0	0	0
Land-scarce[b]	2.03	700.8	84.6	0	0	13.2	3.6	0	0	0
Land-abundant[c]	2.674	1293.8	0	0	0	0	0	0	0	0
Land-scarce[c]	1.1857	695.0	64.5	0	0	13.1	0	0	0	0
Land-abundant[d]	2.674	1293.8	0	0	0	0	0	0	0	0
Land-scarce[d]	1.1857	695.0	0	0	0	0	0	0	0	0

[a,b,c,d] Refer to footnotes to Table 12.5.

maintain their current welfare levels. The limited effect of degradation on their welfare reduces the incentive to mitigate the externality, especially when the rental value of land does not increase with conservation investments. A labour-scarce household with relatively abundant land will cultivate some of the land and rent out the rest. The incentive to treat out-rented land with conservation investments depends on the expected economic benefits. It was found that village land rentals markets do not reflect the value of conservation investments but do reflect land quality aspects that affect its productivity. This means that land of the same quality (whether or not treated with conservation measures) has the same rental value and that there is no short-term economic incentive for the land 'owner' to invest in conservation. Therefore the model does not choose conservation on out-rented plots. This result would have changed if the rental value of land decreases due to soil degradation as in share-tenancy arrangements. Future work may need to assess such effects. Shortage of labour relative to land also means that the labour-scarce household may have to hire-in labour in order to install labour-intensive conservation investments. The cumulative effect of scarcity of labour and land abundance is lower soil conservation effort for the labour-scarce household.

For Type I conservation technologies, it was also found that the labour-scarce households maintain much of the initial conservation (except those on deep soils where erosion effects are low or on marginal lands used for grazing) previously installed on their lands through programme benefits, while the land-scarce households dismantle most of the initial conservation (Table 12.5).

The investment gap and resulting impacts on the welfare of households and the resource base are even more pronounced for Type II conservation technologies that could take some land out of production and reduce initial crop yields (Table 12.6). In this case, both types of households quickly dismantle the existing conservation structures, especially in plots where the perceived risk of erosion is low. However, land-scarce households eventually install them on shallow soils where their effect on productivity is high and hence conservation benefits are large (Table 12.6). The re-investment on some plots seems to occur in later years as welfare levels improve from livestock production and storage of surplus grains. This may not be the case if risk were to be included in the model (Holden and Shiferaw, 2004). Compared to Type I technology, in the 5-year period considered here the overall conservation investment is highly reduced. The households may not, however, have removed the initial conservation investments if a longer planning horizon and a lower discount rate were used (although this may not be a realistic assumption). Moreover, since the discount rates are high and a longer time period is required for conservation benefits to have appreciable effects on productivity, the upper and lower bound discount rates in both cases did not show significant differences in household conservation investments.

Table 12.7. Livelihood and environmental impacts for households without oxen: when conservation technology does not take land out of production.

Household type	Household welfare	Average net income (Birr/consumer unit (CU))[a]	Adoption of conservation practices (% total area) terminal period: Regosols 0–30 cm (1)	Regosols 30–60 cm	Regosols >60 cm	Regosols 0–30 cm (2)	Andosols 0–30 cm (1)	Andosols 30–60 cm	Andosols >60 cm	Andosols 0–30 cm (2)
Land-abundant[b]	3.371	1049.4	22.2	37.7	13.7	0	35.2	28.7	13.7	0
Land-scarce[b]	–2.937	434.0	67.6	95.3	87.5	0	39.1	62.9	87.5	0
Land-abundant[c]	1.901	1027.2	22.2	37.7	13.7	0	35.2	28.7	13.7	0
Land-scarce[c]	–1.921	425.0	22.2	95.3	87.5	0	39.1	65	87.5	0
Land-abundant[d]	1.901	1027.2	74	100	100	0	100	100	100	0
Land-scarce[d]	–1.921	425.0	0	0	0	0	0	0	0	0

[a,b,c,d] Refer to footnotes to Table 12.5.

Table 12.8. Livelihood and environmental impacts for households without oxen: when conservation technology takes 5–10% land out of production.

Household type	Household welfare	Average net income (Birr/consumer unit (CU))[a]	Adoption of conservation practices (% total area) terminal period: Regosols 0–30 cm (1)	Regosols 30–60 cm	Regosols >60 cm	Regosols 0–30 cm (2)	Andosols 0–30 cm (1)	Andosols 30–60 cm	Andosols >60 cm	Andosols 0–30 cm (2)
Land-abundant[b]	3.364	1048.4	0	0	0	0	0	0	0	0
Land-scarce[b]	–3.06	430.0	22.2	0	0	0	39.1	0	0	0
Land-abundant[c]	1.901	1025.6	0	0	0	0	0	0	0	0
Land-scarce[c]	–1.99	421.8	22.2	0	0	0	39.1	0	0	0
Land-abundant[d]	1.901	1025.6	0	0	0	0	0	0	0	0
Land-scarce[d]	–1.99	421.8	0	0	0	0	0	0	0	0

[a,b,c,d] Refer to footnotes to Table 12.5.

Households without oxen

The corresponding results for the two household groups without oxen are presented in Tables 12.7 and 12.8. Under Type I technology, it was also found that the relative abundance of labour and scarcity of land improves the likelihood of sustainability investments. However, compared to households with a pair of oxen, the level of adoption of conservation is reduced, so the productivity and sustainability impacts of improved NRM options are relatively diminished. When the household is poor both in oxen and land, large family sizes put high pressure on the household's ability to meet subsistence needs. While the lack of oxen for ploughing compels the household to rent out land, imperfections in food markets and the presence of price bands work in the opposite direction. Under pressure from conflicting market influences, the household in-rents some traction power to grow a portion of its subsistence needs and rents outs some of its land. It spends about 15% of the available working time on hiring-in oxen for traction. However, meeting the consumption requirements of a large household becomes difficult unless the surplus labour finds some employment off-farm; the household allocates the allowable maximum 25% of the available labour time in activities that include petty trade and employment within and outside the village to earn supplemental income. If the labour market is missing, the model becomes infeasible, indicating that the household is simply unable to meet its subsistence needs unless external assistance (e.g. food aid) is provided. If sufficient off-farm employment is available, labour-rich households without oxen are more likely to reduce on-farm labour and work more off-farm, which may further depress investment in conservation. When off-farm employment is limited (as in this case) the household invests labour to install Type I conservation technologies (see Table 12.7). These investments occur on prime agricultural land where conservation benefits are high while steep slopes [R0-30cm(2) and A0-30cm(2)], mainly used as pasture for livestock, are left without conservation.

When the household is poor in both oxen and labour, the relative abundance of land and shortage of labour discourages conservation investments. The household will hire-in some traction power and labour seasonally to produce part of its subsistence, but will rent out the remaining land annually without conservation. Since fewer workers also mean smaller CUs, the household with relatively abundant land is able to meet its subsistence needs although it invests relatively less in conservation practices. Imperfections in land, oxen, labour and credit markets jointly constrain labour- and oxen-poor households from investing in conservation while compelling them to rent out part of their land assets to labour- and oxen-rich households within the village. If the revenue from land rentals declines because of soil degradation (i.e. rental markets reflect the value of soil conservation), and if labour, oxen and credit markets function well, the labour-scarce household is likely to use much of its land for itself or rent out it after undertaking conservation investments. Currently there is no credit for conservation, and fertiliser credit is minimal and unreliable (see below on the effect of credit). Both selling and long-term leasing of land are illegal in Ethiopia. Along with productivity-enhancing technical change, lifting such restrictions could enhance the value of land and the efficiency of land rental markets. Empirical evidence in Africa and elsewhere shows that under favourable policies (e.g. secure land rights) and market conditions, and when sustainability investments provide high relative returns, smallholders are unlikely to ignore the sustainability impacts of current land-use decisions (user costs) (Tiffen *et al.*, 1994; Heath and Binswanger, 1996; Templeton and Scherr, 1999; Holden *et al.*, 2001). These are important policy constraints that need to be tackled to encourage land investments in Ethiopia.

As expected, labour-scarce households maintain more of the initial conservation measures than land-scarce households. The situation is very different for Type II conservation technologies (Table 12.8). In this case, both households remove the conservation structures on their plots. Only land-scarce and labour-endowed households allocate some labour for conservation. Hence, the level of conservation adoption is minimal and the attained impact on the quality of the resource base is very limited mainly because exploitative traditional agricultural practices with high levels of soil erosion (up to 40 t/ha) continue (Shiferaw and Holden, 2001).

Economic and sustainability impacts

The above results have clearly shown the roles of land and labour scarcity in household conservation investment decisions. It was hypothesised that the endowment of traction power and farmland will largely determine the welfare impacts of new technologies. Households that are poor in land and oxen can therefore be expected to attain the lowest level of welfare. The discounted utility (welfare) and the average net income per CU for the different scenarios are presented in Tables 12.5–12.8. The results show that adoption of NRM practices is very minimal for Type II technologies. This means that the farmer will largely use existing practices and the welfare and environmental impacts from such interventions will be minimal. Comparison of the welfare and income differences under Type I and II technologies can therefore reveal the economic impacts associated with adoption of improved NRM practices. For example, the land-abundant household attains a welfare level of 4.521 under Type I, which declines to 4.511 under Type II where no adoption has occurred, representing a discounted welfare gain in 5 years of 0.22%. Similarly the average net annual income per CU has shown a slight increase of about Birr 2 (0.15%), which amounts to about Birr 10 in 5 years. These are direct benefits associated with the reduction in soil degradation from adoption of the conservation technology. It is to be noted that the best NRM technology simulated (Type I) does not enhance yield; it only reduces soil erosion while yields remain unchanged in the initial years. The economic gain would have been more pronounced had the NRM technology also contributed to growth in crop yields. Moreover, in all the scenarios simulated, the better-off households with two oxen attain the poverty line level of welfare ($U_t>0$ and $DU>0$) under both technology alternatives. Oxen-

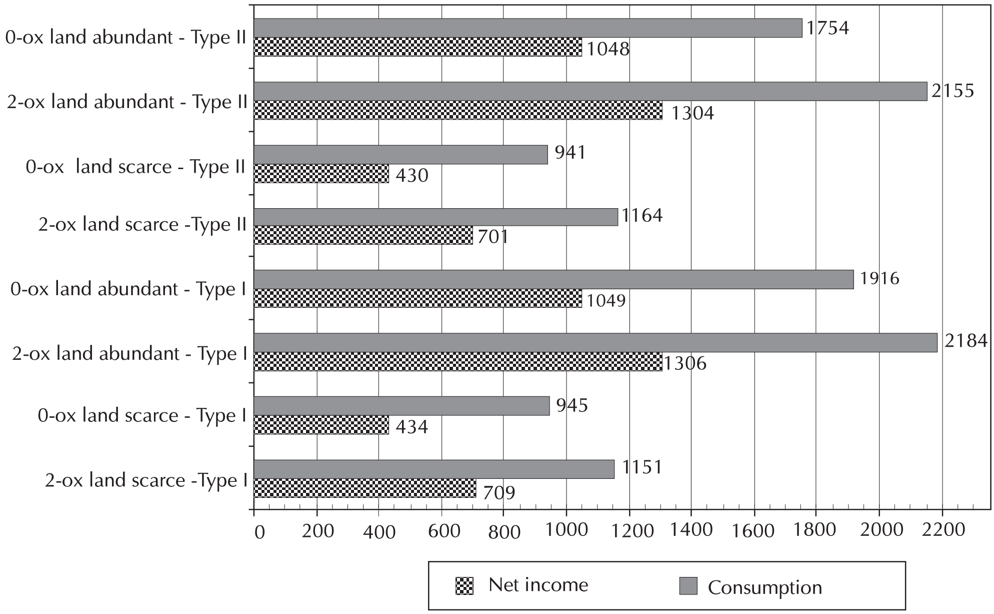

Fig. 12.2. Average annual net income (Birr) and consumption (kcal) per consumer unit by household group (US$1 = Birr 6 in 1993/94).

owning and land-rich households with fewer consumers attain the highest level of welfare. Households without oxen attain the poverty-line welfare level only when land is not scarce and the number of consumers is limited. A combination of land and oxen poverty along with insufficient off-farm employment opportunities makes the household unable to attain the poverty-line full income. Hence, these households are unable to escape poverty ($U_t<0$) even when Type I conservation is used. This probably explains why many poor households in the area also have small-sized families.

The income and consumption outcomes (at low discount rates) are depicted in Fig. 12.2. Using the annual poverty line income of Birr 528/CU and subsistence calorie requirement of 840 kcal/CU, the results show that all household groups attain the subsistence level of consumption but not the poverty-line net income. Land-scarce households without oxen fall far short of this level of income even though they meet their subsistence level of calorie consumption.

It will be useful to assess the level of economic gain from adoption of improved conservation practices. The gain in household net income attained per unit of land area conserved can be estimated from comparison of the net income with and without adoption of Type I technologies. For example, the average household annual net income for land-scarce and two-oxen households with adoption of Type I technologies is about Birr 51 higher than that without adoption. This amounts to about Birr 36/ha/year of conserved land. If irreversibility in soil degradation is assumed, the perpetual on-site net gain from adoption of conservation practices amounts to Birr 72 to 180/ha

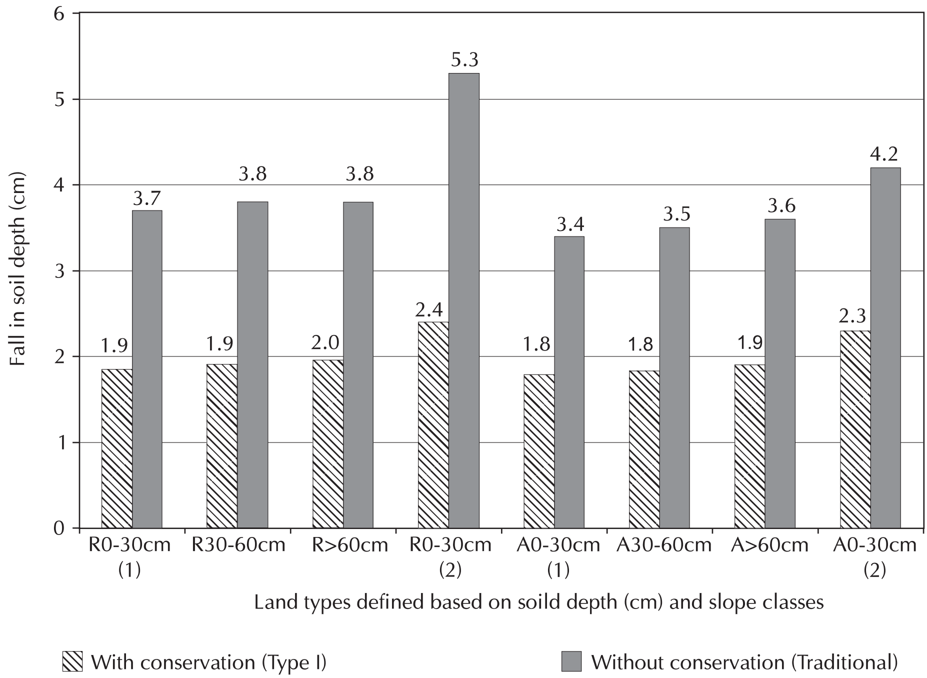

Fig. 12.3. Reduction in soil depth in 10 years: land-scarce 2-ox household (land types as defined in Table 12.3).

using the farmer's high (50%) and low (20%) discount rates. Adoption of high-yielding varieties and other options is likely to increase the net farmer benefits from conservation.

In order to show the long-term environmental or sustainability impacts of adopting improved management practices, the model was solved for a planning horizon of 10 years under Type I and traditional practices. The results are shown in Fig. 12.3. The fall in soil depth under Type I conservation technology is about half of that under traditional management. Depending on the soil and land type, soil depth declines by about 1.8–2.5 cm with conservation, but this increases to 3.2 to 5.4 cm under traditional management. As was shown in Equation 7.2, crop yields depend on many variables including the use of organic and inorganic fertilisers. Figure 12.4 shows the effect of soil degradation on crop productivity under differing levels of fertiliser use. If farmers do not use chemical fertilisers, barley yields decline by about 175 kg/ha without conservation (No Cons), while this loss falls to less than 50 kg/ha with conservation (Cons). This indicates that, depending on the relative returns, farmers have the option of using fertilisers to replace lost nutrients or of investing in conservation practices to mitigate the effect of soil degradation. Policies for fertiliser or conservation subsidies have been used to achieve productivity and/or sustainability objectives. Since fertiliser price subsidies are no longer popular policy options, it could be useful to investigate how the credit constraint might affect farmers' conservation choices. This is explored further in the following section.

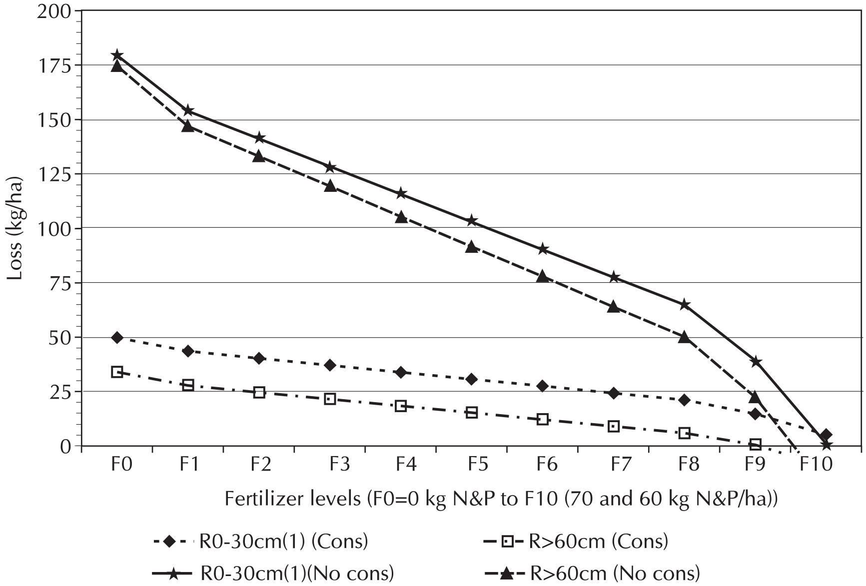

Fig. 12.4. Fertiliser use and decline in barley (*Meher* season) yields in 10 years under alternative land management practices (Type I vs. traditional).

Effect of credit for fertiliser

As discussed earlier, credit was not included in the base model presented thus far. What happens if the Ethiopian government increases allocation of credit for conservation and production purposes in the future? Availability of credit and fertiliser use are critical ingredients for stimulating adoption of improved technologies. Several earlier studies have shown that subsidised credit may increase fertiliser use (especially when profitable varieties are available) and may discourage investment in soil conservation (Shiferaw *et al.*, 2001; Holden and Shiferaw, 2004). When cheap credit is available, high levels of fertiliser use can easily replace lost nutrients and reduce the need for soil conservation. The same effects can be expected from fertiliser price subsidies. As Fig. 12.4 shows, under high levels of fertiliser use, the relative productivity benefits of conservation disappear and crop yields will be similar to those without conservation. We find that for credit-constrained households, increased availability of input credit could discourage investment in conservation. This is demonstrated using results for the poor and land-scarce household group (Table 12.9). As the availability of input credit improves, the level of conservation investment declines progressively, even when Type I conservation technology is available. With Type II conservation technologies, access to credit seems to entirely wipe out all the incentives for conservation. In this case fertiliser use becomes more economical than soil conservation to counter soil degradation. The decrease in sustainability investments occurs while short-term welfare improves because of increased fertiliser use and improved land productivity. It is not clear, however, for how long fertilisers

Table 12.9. The effect of access to credit on smallholder conservation investments (land-scarce households without oxen).

Techno-logy type	Credit level[a]	Fertiliser use (kg)[b]	Regosols				Andosols			
			0–30 cm (1)	30–60 cm	>60 cm	0–30 cm (2)	0–30 cm (1)	30–60 cm	>60 cm	0–30 cm (2)
I	No credit	D=150, U=62	67.6	95.3	87.5	0	39.1	62.9	87.5	0
	Limited	D=235, U=134	65.2	84.8	25	0	39.1	35	62.5	0
	High	D=250, U=147	32.1	41.9	25	0	39.1	35	12.5	0
II	No credit	D=150, U=62	22.2	0	0	0	39.1	0	0	0

[a] The 'limited' credit level is specified as Birr 400 from formal (12% interest) and Birr 250 (60% interest) from informal sources (US$1 = Birr 6 in 1993/94). The 'high' level of credit is twice that of the 'limited' level.
[b] D = diammonium phosphate (DAP), U = urea.

can be used to mitigate the effect of soil degradation. Agronomists argue that a minimum soil depth is essential for crop production and that once soil erosion reduces the rooting depth below a given threshold level, the marginal productivity of fertiliser use may decline. This indicates that as soil degradation increases, more fertiliser may be required to compensate for losses and to sustain crop productivity. This trade-off could be tackled through interlinkage of credit supply with conservation requirements (Holden and Shiferaw, 2004), a policy that could foster win–win economic and environmental outcomes.

Conclusions and Policy Implications

In resource-poor regions with high population pressure, sustainable use of land and other resources has become an important policy and development problem. Improved NRM interventions are important to reverse soil degradation and sustain agricultural productivity. Several recent studies have posited a nexus between poverty and the ability to undertake sustainability investments, especially when markets are imperfect. Bioeconomic models that interlink biophysical information with behavioural economic models at different spatial scales in a dynamic perspective are most suited to the analysis of NRM impacts and to determine how poverty in certain assets affects investment decisions. Using data from the Ethiopian highlands, it has been shown how a non-separable bioeconomic household model can be used to track these relationships and impacts, and how the effect of technology and policy changes affecting NRM can be evaluated simultaneously in terms of economic efficiency (the incentive to adopt the technology), welfare (poverty effects) and sustainability (resource conditions). The model is formulated for four stylised household groups and captures production, biophysical diversity and market conditions in the area. The results show that when land is relatively abundant, households are unlikely to carry out labour-intensive conservation investments. An increase in family labour coupled with scarcity of land, however, increases the incentive to invest in conservation, especially when opportunities for off-farm employment are limited and profitable conservation technologies are available. In this case, higher adoption of resource management practices leads to positive impacts on household welfare and sustainability of resource use.

It is also found that poverty in labour and traction power forces households to rent out land to other relatively better-off households. Under the existing system of usufruct rights to land in Ethiopia, sustainability investments that do not affect short-term crop yields do not affect the rental value of land. In this case, the oxen- or labour-poor households rent out land without conservation because the returns from renting are the same. This points to the need for new policies and interventions that would improve the efficiency and effectiveness of land rental markets and create incentives for land users to consider the future productivity impacts of current land-use decisions (user costs).

The economic incentive to invest in conservation drastically decreases when the new technologies increase scarcity of land and decrease crop yields in the short term (Type II). This scenario seems to explain the extensive removal in the study area of conservation measures introduced in the past. Unfortunately, better options that provide short-term benefits to the poor are rarely available and the only reasonable way to encourage investments in such practices is to provide some targeted subsidies (e.g. cost-sharing). However, when farmers are able to perceive the consequences of soil degradation and use-rights are secure, they are able to adopt Type I conservation technologies without additional incentives. Only labour-scarce households and those without the necessary traction power are unable to make significant investments due either to the relative abundance of land or to the high opportunity costs of labour.

The direct economic gains from the adoption of Type I technologies are quite modest. The average annual income gain is estimated at about Birr 36/ha, which translates to an increase in annual income per consumer of Birr 10 in 5 years. This is partly because the nature of the technology simulated in this case does not improve yields. Higher benefits can be expected if conservation also enhances land productivity. But the low return to available conservation technologies is a major factor that makes conservation investments less attractive than competing alternatives (e.g. off-farm employment or livestock production). This suggests the need to develop NRM technologies that provide attractive economic gains along with sustainability benefits. Land-scarce households without oxen even failed to attain the poverty-line income. The level of conservation adoption and its impact is lowest for land-abundant households. Adoption of conservation measures did not arrest soil degradation, but did provide substantial benefits in terms of maintaining soil depth and improved crop productivity. The decline in soil depth with conservation is half of that under traditional practices, but the yield reduction is less than one-third of that without conservation. Fertiliser use could also reduce yield losses. There is some evidence that increased fertiliser credit may substitute for conservation effort. This may require cross-compliance types of policies that link fertiliser credit with conservation requirements.

Nevertheless, evaluation of economic and environmental impacts will not be complete until the added social benefits are compared with the research and development (R&D) costs incurred in generating and delivering these technologies on a larger scale. When these costs are low and the associated economic and sustainability benefits are high, improved social efficiency from such NRM investments can be expected.

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16 Towards Comprehensive Approaches in Assessing NRM Impacts: What We Know and What We Need to Know

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Introduction

The deliberate use of ecosystems by mankind to meet food, feed, industrial, and social and environmental needs inevitably alters the natural ecosystem functions and services. While flux is inherent to ecological systems and their evolution, the natural resource base is currently facing unprecedented human pressure due to population growth and rising consumer demand that follows rising incomes. This human pressure creates a growing need to improve the productivity of existing natural resources and to counter processes that deplete their productive capacity. Governments around the world have responded to the degradation of the natural resource base with projects aimed at sustaining productivity levels and environmental quality. The rising proportion of research funds directed at natural resource management (NRM) at the Consultative Group for International Agricultural Research (CGIAR) is indicative of strong interest (Kelley and Gregersen, Chapter 15, this volume). The increase in funding brings with it an increased need for accountability, *ergo* the urgency of improving impact assessment of NRM investments.

The introduction to this volume summarised the special difficulties in measuring the impacts of agricultural technologies that are designed to enhance the sustainability of natural resources needed for human survival. The technologies themselves are diverse; they range from genetic improvements that allow crops to grow in inhospitable places to conservation practices that reduce soil loss and water pollution. Although a few NRM innovations boost farm revenues (e.g. via enhanced yields), most of the benefits to resource users come from cost-savings, reduced vulnerability to risk (e.g. yield stability) and the avoidance of declining productivity. Examples of such technologies include biologically based soil fertility management, soil and water conservation, water harvesting, integrated pest management, water-saving irrigation, minimum tillage, agroforestry and forest management, rangeland management, and biodiversity conservation. These innovations when adopted provide significant positive environmental and sustainability benefits both on-site and off-site.

The special characteristics of NRM technologies mean that a balanced economic impact assessment must be able to measure environmental and sustainability impacts above and beyond what would have occurred in their absence, a task that has often been ignored in impact assessments heretofore (Nelson and Maredia, 1999). As outlined in the introductory chapter, comprehensive NRM impact assessments pose special problems for establishing the counterfactual, measuring environmental effects, placing a value on those effects, and integrating the final results into a unifying framework.

This book has focused squarely on addressing the methodological challenges for evaluating the impacts of NRM. The preceding 15 chapters have presented and discussed the key issues, challenges, indicators, and valuation and evaluation methods. The sections that dealt with methodological advances were further enriched through case studies that illustrate how impact evaluations can integrate economic and environmental impacts. As agricultural research and development enters a new era through harnessing biotechnology and integrating genetic and resource management, diverging perspectives are emerging on how future impact assessments need to be carried out. The book has highlighted some of these views and outlined areas for future research.

This concluding chapter synthesizes the conceptual, methodological and empirical issues for evaluating the impacts of NRM technology and policy interventions. The intention is to highlight the salient features raised across the chapters and offer some insights on the key lessons, policy conclusions, knowledge gaps, and areas that need further research.

What We Know: The State of the Art in NRM Impact Assessment

Substantial experience has now been gained in applying economic impact assessment methods to productivity-enhancing agricultural research. Measuring changes in economic surplus associated with improvements in agricultural technologies is the most commonly used method in evaluating social net gains from research investments. Alston *et al.* (1995) and Maredia *et al.* (2000) provide a good review of best practices for *ex post* impact evaluation of the economic impacts of agricultural research programs. Despite extensive work on environmental valuation and benefit–cost analysis, there is a dearth of literature on methods for valuation of ecosystem services from NRM technology. Likewise, there are few empirical studies of the social impacts of NRM. Recent years have witnessed a gradual shift in the evaluation literature towards looking at the non-productivity related environmental and sustainability impacts of crop and resource management interventions (e.g. Traxler and Byerlee, 1992; Pingali *et al.*, 1994; Gumtang *et al.*, 1999; Gupta and Abrol, 2000; Pretty *et al.*, 2000). Improved methods are now being developed for comprehensive evaluation of the economic welfare impacts of agricultural interventions – including the productivity and environmental costs and benefits.

Beyond market-based assessments

A recent survey of 1100 agricultural research impact assessment studies found that only 11 included environmental impacts (Alston *et al.*, 1998). In the face of rising expenditures on NRM projects, the fact that 99% of past impact studies relied on measures of economic efficiency alone highlights the need for better assessment of impacts related to sustainability and environmental quality. Because many NRM problems involve economic externalities and/or public goods, neither the problems nor the impacts of NRM technologies designed to solve them are readily measured in markets. It is now widely accepted that impact assessment of NRM interventions should look beyond conventional market-based techniques. Non-market valuation methods, now widely in use in the developed countries for assessment of environmental impacts, can be tested and adapted for evaluating the non-marketed impacts of agricultural and NRM practices. One major challenge is how to measure or find indicators for the dynamic and multidimensional impacts of NRM technologies in agriculture.

Measurement problems

In order to assign economic values to changes in the flow of ecosystem goods and services, the essential first step is to understand how the new interventions affect the quality or quantity of the resource in question and how that translates into changes in goods and services that people value. Changes in ecological functions and processes may be very gradual and take a long time to manifest. Moreover, the dynamic, interdependent nature of ecosystems makes it hard to measure a clear cause–effect relationship from an NRM technology intervention. A basic hurdle in measurement and quantification of biophysical changes therefore has been the incomplete understanding of how NRM practices affect ecosystem health and sustainability.

However incomplete, human knowledge about ecosystems is growing. Long-term experimentation in selected systems has provided useful information about system dynamics and how crop and resource management interventions affect agricultural productivity and resource conditions. The need for such experimentation is even stronger in locations where variability of production conditions is high and data from a short time-series will fail to capture the underlying variation. Simulation models that emulate soil, water, nutrient and crop interactions are now widely used. If properly validated using site-specific biophysical and climatic conditions, such models can be very helpful in evaluating the impacts of multiple changes. This is particularly the case for the integrated interventions of what has now come to be known as integrated natural resource management (INRM). Satellite imagery and geographic information systems are becoming useful tools for monitoring the spatial and temporal dynamics of changes in patterns of land use, vegetation cover, drought stress, surface water, water logging and land degradation. These tools are also gaining importance in yield forecasting and assessment of production risks.

A combination of these scientific advances is making it possible for biophysical scientists and agro-ecologists to estimate physical, chemical and biological changes in agro-ecosystems associated with NRM interventions. When such changes can be understood or predicted, certain measurable indicators can be developed to quantify the magnitude of change associated with a given intervention. Indicators may be developed through experimentation and proper monitoring of changes over a sufficient period of time or through the application of exploratory and predictive simulation models. Chapters 3, 4 and 5 in this volume examined specialised indicators of soil quality, water quantity and quality, and changes in other agro-ecosystem services.

For impact assessment purposes, the most useful indicators of ecosystem functions and services show impacts within 3–5 years of an NRM intervention. For soils, Pathak *et al.* (Chapter 3, this volume) find that biological indicators like soil respiration, microbial biomass, and C and N mineralisation are useful, as are physical indicators such as nutrient runoff and soil loss. Relative to changes in soil quality indicators, changes in surface and groundwater quantity and quality can be observed in a relatively short period of time (Sahrawat *et al.*, Chapter 4, this volume). In order to measure NRM impacts on agro-biodiversity, Wani *et al.* (Chapter 5, this volume) suggest the following indicators for observation within 3–5 years: the index of surface percentage of crops, crop agro-biodiversity factor, and surface variability factors. Changes in biodiversity indicators related to genetic variability, species diversity or richness require longer periods to become visible, indicating the need for long-term follow up and monitoring. Changes in the level of carbon sequestered in soils and vegetation may require even long periods, making simulation modelling a promising approach for predictive purposes.

Valuation problems

When public funds have been invested in developing environmental services and measurable indicators of those services have been identified, a natural question is how to value changes in their status (as a step toward measuring return on investment). The value of a given resource or environmental service is measured in terms of trade-offs that consumers face with or without the change. The techniques for eliciting this information depend on the kind of markets at hand. For marketable goods and services, observed market behaviour can be used. Two such methods, applied by Drechsel *et al.* to valuing soil fertility changes, are the calculation of replacement cost and the value of a productivity change (Chapter 9, this volume). Even when a natural resource service is not traded, so long as there exist marketed substitutes, the behaviour observed in markets for the substitute can be used for valuation of changes in quality or quantity. However, markets for factor inputs (e.g. land and labour) in developing countries are often imperfect, limiting the usefulness of market prices in valuation studies. Even when markets function well, NRM technologies may not generate goods and services that are traded in markets. As Shiferaw *et al.* (Chapter 2) show, the social benefits associated with changes in NRM are typically non-marketed, ruling out the use of actual markets to measure the economic values of changes in natural resource service flows due to NRM technologies. However, techniques exist for estimation of non-use values and indirect use values that are not traded in markets. In particular, contingent valuation and similar non-market valuation techniques need to be tested and developed for application to NRM impacts in agriculture. Although benefit transfer methods have been proposed to reduce the cost of estimating non-market values, they are of limited relevance when economic and ecological conditions differ markedly between the original location and the one where the values would be applied.

Attribution Problems

Establishing a cause–effect relationship between NRM programme interventions, intermediate outcomes and developmental or environmental impacts can be challenging. First, *ex post* impact assessments often rely on scanty cross-sectional adoption data, making it difficult for the impact evaluator to see the full picture of technology dissemination. Second, crop and resource management research often is not embodied in an observable physical entity that farmers can adopt or reject. The improved management practices are knowledge-intensive techniques transmitted as a recommendation or as a cognitive framework regarding such topics as pest management or soil conservation. Among the multiple sources of such information, it may be difficult to attribute changes in management practices to any given source (Traxler and Byerlee, 1992). Third, in contrast to crop improvement research, NRM research frequently involves multiple interactions, multiple stakeholders, and participatory processes. These characteristics pose formidable complications to the attribution of project impacts to a given research or development intervention (Freeman *et al.*, Chapter 1, this volume). Douthwaite *et al.* (Chapter 14, this volume) discuss the rationale for qualitative, step-wise and adaptive monitoring and evaluation methods for understanding the innovation process and how adoption begets outcomes that in turn beget impacts.

Beyond attribution, NRM impact analyses must measure impacts against the counterfactual case of what would have occurred in the absence of the NRM intervention – whereas scientific experiments typically include a control treatment as a baseline against which to judge other intervention effects. Although social programmes are often practically (or ethically) constrained from including a true control treatment, impact assessments must still characterise and try to measure the counterfactual case.

Various quasi-experimental approaches are suitable alternatives (Cook and Campbell, 1979). Baseline data are essential for reliable estimates of the changes attributable to the NRM intervention. One practical approach is the double-difference comparison. This method involves comparing relative changes in performance indicators before and after the NRM intervention between participants and non-participants. Careful research design and statistical analysis can help control for selection bias and other attribution problems (Pender, Chapter 6, this volume).

Integration of Resource and Environmental Impacts into Economic Impact Assessment

The economic surplus (ES) framework is the most desirable approach for summarising the economic welfare impacts of agricultural research investments. The classic ES approach measures the shift in a product supply curve resulting from technological change. The supply shift triggers changes in consumer surplus and producer surplus. Although benefit–cost analysis has been applied to a number of NRM projects, there have been scarcely any attempts to apply the ES approach (Alston *et al.*, 1995; Swinton, Chapter 7, this volume). The ease with which resource and environmental impacts can be integrated using this framework depends on the type of NRM intervention. Because non-market environmental or health effects often are not directly tied to agricultural output, productivity and environmental impacts must be calculated separately. Estimating ES for environmental impacts will require a simulated or surrogate market in which the marginal willingness to pay (WTP) (demand) curve can be estimated separately. Further research is needed to define the conditions under which the total ES may be measured as a sum of the economic surplus from productivity changes in the marketed commodity plus the estimated economic surplus from the simulated markets for environmental and health services. NRM may also change the quality of the products, which may induce a shift in consumer demand as well as in producer supply. Impact evaluation in this case will require measurement of the supply as well as the demand shifts (Swinton, Chapter 7, this volume).

At present, however, the suggested methods for integrating quantitative estimates of both marketed productivity impacts and non-marketed environmental impacts are untested. The current state of the art is exemplified by Bantilan *et al.* (Chapter 11, this volume), which combines an estimate of economic surplus based on marketed productivity changes with an inventory of environmental benefits and costs. The authors conduct a qualitative assessment of the environmental benefits vs. costs, concluding that environmental net benefits are positive. Based on this result, they infer that the market-based net benefits estimated from productivity enhancement alone are a lower bound for the true combined net benefits from both productivity and environmental dimensions.

Two broad classes of empirical methods are used to estimate changes in ES. When past data are available about the performance of NRM interventions, econometric regression methods can be used for several important purposes. First, econometrics is widely used to test the potential effects of NRM changes on productivity (Pender, Chapter 6, this volume). When data from a sufficiently large sample is available, econometric methods are useful in testing whether investments in specific crop and resource management practices had significant effects on productivity or on the quality of the resource base (Pender, Chapter 6; Kerr and Chung, Chapter 10, this volume). Careful econometric analysis can substantially reduce the problems of attribution. Second, econometric inverse demand models are used to estimate the price elasticity of demand for marketed (and non-marketed) products. Third, econometric models can identify the factors determining both: the likelihood of adoption of an NRM innovation; and the degree of NRM used by those who have adopted.

When sufficient data are not available for econometric estimation, an alternative useful approach for estimating the magnitude and form of production and environmental effects is bioeconomic modelling (Kruseman and Bade, 1998; Barbier and Bergeron, 2001; Okumu *et al.*, 2002; Holden and Shiferaw, 2004). Using mathematical relationships, bioeconomic models link economic behavioural objectives with key ecological and production processes that determine biophysical outcomes (Oriade and Dillon, 1997). As discussed by Holden (Chapter 8, this volume) such integration allows the analysis of efficiency, distributional and sustainability impacts of proposed technology and/or policy interventions (Ruben *et al.*, 2001). The approach can also be used to measure the impact of these interventions *ex post*. A household-scale example is the impact analysis of soil and water conservation technologies (Shiferaw and Holden, Chapter 12, this volume). At the regional scale, computable general equilibrium (CGE) models become very useful to capture the economy-wide impacts of technology and policy interventions (Holden and Lofgren, Chapter 13, this volume). CGE models are particularly suited for assessing price effects and distributional issues associated with technical and policy interventions.

What We Need to Know – Areas for Future Research

Despite recent progress in developing methods for evaluating the impacts of productivity enhancing technologies on the one hand and for measuring natural resource service flows and their value on the other, these advances have not been unified in NRM impact studies. With very few exceptions, NRM impact evaluations have failed to incorporate the non-productivity related impacts (resource and environmental service flows) into economic impact assessments. This volume has brought together some of the methodological tools that can be used to integrate the sustainability impacts with the productivity impacts of agricultural NRM interventions. But the state of the art does not yet permit us to advocate 'best practices' for comprehensive evaluation of NRM impacts. Several knowledge gaps first beg the attention of researchers.

How does NRM affect ecosystem functions and services?

Our understanding of the impacts of human interventions on ecosystem functions and services at different scales and how this affects productivity, sustainability and environmental outcomes is still inadequate. The concept of 'natural resource management' itself is very broad, ranging from crop and livestock management practices to strategies for managing natural resources such as soils, water, biodiversity, fish and forests. Agricultural activities may have important externalities, such as global warming. Improved NRM enhances the provision of essential ecosystem services that reduce such negative environmental externalities. How different types of NRM interventions affect the flow of ecosystem services at different spatial and temporal scales is, however, not clearly understood. While there are several reports on the environmental impacts of intensive agricultural activities (e.g. the Green Revolution), there are few empirical examples for crops other than wheat and rice (Maredia and Pingali, 2001). The limited evidence and insufficient understanding of the key links between agricultural activities and how NRM would regulate this link, prevent quantification and measurement of key outcomes and potential impacts on human welfare. As Altieri has argued, 'what is lacking … is the explicit description of the scientific basis of NRM and of methods to increase our understanding of the structure and dynamics of agricultural and natural resource ecosystems and providing guidelines to their productive and sustainable management' (Altieri, 2002, p. 7). Such understanding is a key first step in enhancing attribution of certain environmental outcomes to NRM interventions. Progress toward better definition of agro-ecosystem functions and services is urgently needed. Simulation modelling offers an increasingly valid and cost-effective tool for understanding the biophysical dynamics of NRM interventions.

Indicators of ecosystem performance

To the extent that agricultural natural resource functions *are* understood, the measurement of their status and service flows remains too costly for practical impact assessment purposes. Inexpensive but reliable indicators continue to be needed. A core set of environmental and sustainability indicators would allow researchers to check for deviation from trend by gathering time-series data for regular ecosystem monitoring, not to mention establishing the counterfactual to NRM interventions. Some preliminary steps have been taken by the Heinz Foundation (http://www.heinzctr.org/ecosystems/index.htm), which started to monitor the state of US ecosystems in 1999. Their efforts offer useful criteria for consideration, although their indicators obviously need adaptation to the developing country settings of most NRM projects.

How to enhance attribution of impacts?

More systematic thinking is needed about how to measure the dissemination of knowledge-based technologies that are not embodied in improved tools or germplasm. Knowledge-based innovations appear less well suited to the reduced form input demand approaches that economists have used for embodied technologies like improved seeds. Better indicators for ecosystem performance measurement can help. So too can direct approaches to measuring farmers' knowledge and attitudes and how they affect the choice of management practices. The knowledge–attitudes–practices (KAP) model from epidemiology may be a start, as the explicit measurement of changes in knowledge and attitudes of a treatment group compared with a control can confirm attribution to project interventions. Indeed, explicit attribution becomes doubly important – albeit doubly complicated – when NRM technologies are introduced in tandem with genetic technologies or a newly supportive public policy. Careful adherence to sound impact assessment methods (especially the double-difference method) and strict adherence to avoid or measure selection bias among beneficiaries will have to be joined to closer scrutiny of knowledge and attitudes.

Can we properly value non-market ecosystem services?

Even when we can understand and measure cost-effectively the resource and environmental service flows from NRM interventions, shortcomings in our ability to measure the welfare impacts of these changes can impede accurate assessments. The reviews in this volume have identified several techniques used for valuation of non-market outcomes in the developed world (Shiferaw *et al.*, Chapter 2). Valuation methods for non-marketed ecosystem services (e.g. carbon sequestration in soil or biodiversity preservation) need to be tested and refined.

Many methods for measuring WTP for environmental services presuppose that consumers directly demand the service in question. Yet many agricultural NRM services do not fit that description. Few consumers would pay for the presence of *Rhizobium* bacteria in soil, yet the nitrogen-fixing services that they perform provide plant nutrition and, if carefully timed, may reduce nitrate leaching into drinking water supplies. In short, the demand for the services of *Rhizobium* bacteria is indirect, not direct. As such, it is analogous to the demand for other agricultural inputs. Two key factors differ, however. First, whereas conventional derived input demand arises solely from market prices and factor endowments, part of the derived demand for NRM services originates in a direct demand for health that affects the non-market valuation of exposure to reduced drinking water quality. Second, that same health component involves externalities to neighbours of the producer, a stakeholder group whose members' utility is not included in an indirect demand function based upon the marketed agricultural product. Moving from theory to practice in measuring indirect WTP for environmental and health services will be complex. A major desirable innovation is to find lower cost – yet accurate – ways to estimate downward-sloping inverse demand curves as a basis for estimating elasticities of demand for non-market environmental and health services.

For policy purposes, a simpler approach than measuring WTP is to measure farmers' willingness to *accept* compensation for the non-marketed health and environmental services that they provide. This will be a compensating surplus measure for farmers to provide essential services to society. Such measures will require more bioeconomic modelling in order to estimate the opportunity costs implicit in providing cost-increasing health and environmental services.

Can the economic surplus approach be extended for integrated assessment?

Despite its strengths, the economic surplus approach has been criticised on several counts. For purposes of NRM impact assessment, the most serious of these is its failure to account for environmental impacts that are external to functioning markets. One of the goals of NRM is to reduce the undesirable on-site and off-site externalities associated with agricultural production. Two chapters in this volume (Swinton, Chapter 7 and Bantilan *et al.*, Chapter 11) have discussed the ways to extend the economic surplus approach towards comprehensive evaluation of productivity and environmental impacts. However, progress in this area has been hampered by measurement problems, the high cost of WTP estimation, and the difficulties of mixing values assessed from different market settings (e.g. real markets and hypothetical ones). Initial efforts to integrate productivity and environmental impacts in a comprehensive assessment should focus on simple cases where price elasticities of demand can readily be estimated. Serious thinking is needed on how to combine productivity and environmental effects in computing a single, comprehensive measure of impact from NRM interventions.

Alternatively, the economic surplus approach to productivity impact assessment may be supplemented by qualitative information. Some audiences uncomfortable with the demanding assumptions required for many WTP estimation studies may consider these methods more valid. The participatory methods for interdisciplinary analysis of adoption pathways, processes and outcomes may also contribute to participant empowerment that can enhance impacts, whether or not they enhance impact assessment *per se*.

What do we know about economy-wide impacts?

In addition to direct effects associated with supply shifts, agricultural productivity interventions also generate indirect economic effects through product and factor market linkages. The overall effect of technical change from research and development (R&D) interventions hence depends on system-wide growth and multiplier effects induced through input use, factor markets and production linkages (Maredia *et al.*, 2000). For comprehensive evaluation of large-scale NRM impacts, it would be useful to include these general equilibrium or economy-wide effects. While this can be done using a CGE model (Holden and Lofgren, Chapter 13, this volume), there is limited experience in developing CGE models that incorporate environmental and sustainability impacts. In situations where substantial impacts occur from both general equilibrium market effects and sustainability effects, it could be very rewarding to develop and employ such methods. Standard CGE models (Lofgren *et al.*, 2002) are now being developed for many developing countries, and these models deserve research into possibilities for adaptation to evaluate NRM technology and policy impacts.

Simple steps toward better impact assessments

Advance planning can greatly improve the quality of NRM impact assessments. The classic principles of quasi-experimentation remain relevant: to compare affected and unaffected groups before and after the program intervention, taking care not to bias results due to non-random selection of participants (Cook and Campbell, 1979). The few NRM impact assessments available have had adequate baseline data only on the productivity dimension, not on environmental and health dimensions. In some instances, this is because intended environmental and health outcomes had not been clearly specified at project outset.

For effective impact assessments, baseline data on all intended outcomes dimensions is necessary. Acquiring such data calls for projects *before implementation begins:* 1. to specify clearly the intended outcomes; 2. to choose acceptable indicators of important outcome dimensions; 3. to identify comparable, paired groups inside and outside the intervention area; and 4. to budget for and to conduct baseline studies on the intended outcomes and related variables for the paired groups within and without the NRM intervention zone. It goes without saying that planning and budget are also needed for one or more follow-up studies to measure progress toward the intended outcomes – again, among comparable households both affected and unaffected by the NRM programme. Ensuring that appropriate baseline and follow-up data are collected is not only possible; it will also greatly facilitate advances in the methodological areas listed above.

Institutionalising NRM impact assessment

If it is to affect institutional decision making, NRM impact evaluation needs to be integrated into programme planning in research and development institutions. An institutional learning cycle from programme planning to implementation to impact analysis and back to programme planning can help to ensure that lessons are learned and plans modified systematically. Such a process is more likely to prevent uncorrected flaws from turning well-conceived programmes into vaunted 'failures'.

NRM impact assessment can be conducted internally within R&D institutions or subcontracted to external evaluators. But there are compelling reasons for R&D institutions to institutionalise IA capacity 'in-house' if they are to be effective in influencing internal programme planning. Some R&D institutions have institutionalised impact assessment within an independent economics or social science programme. However, given the trend toward replacing disciplinary research areas with thematic research areas, a promising approach is to institutionalise IA capacity into a specialised impact assessment unit that reports directly to senior management. Staff in this unit should be drawn from both the social and the biophysical sciences, in order to provide comprehensive analysis of the multi-dimensional and non-monetary impacts of NRM interventions. Such a unit can provide intellectual leadership for all IA studies and can provide a platform for integrating the results from impact assessment studies into institutional learning and research planning for the purpose of enhancing future impacts.

An independent IA unit can be effective at forging strategic alliances between research institutes, development partners, and advanced research institutes. The chapters in this volume demonstrate how comprehensive assessment of NRM impacts can emerge from strategic partnerships between university-based researchers (with a comparative advantage in development of theories and methods for assessing NRM impacts) combined with researchers in R&D institutions (with comparative advantages in empirical applications of these methods, synthesis of experience, and scaling-up results).

Research managers also need to think carefully about how much to invest in impact assessment. A standing IA unit can be expensive, and R&D institutions exist primarily to generate impacts, not to measure them. Yet in a world where many institutions claim to generate impacts and compete for funds to sustain their efforts, a competitive advantage can be built from the institutional capabilities to perform high-quality impact assessments and to adapt programme planning systematically based upon the lessons learned. Building such capabilities will require a modest proportion of core funds on a continuing basis, with the understanding that the IA unit will help to attract competitive funds through collaboration with thematic units on project design.

Conclusions

Reducing poverty and ensuring livelihood security for the millions of impoverished people whose subsistence depends on agriculture will not be possible without judicious management of the productive resource base. But failure to demonstrate desired impacts could undermine current R&D efforts in developing and disseminating new innovations that provide dual productivity and sustainability benefits. Although many NRM interventions do not provide direct short-term net economic benefits to producers, they do generate non-marketed ecosystem goods and services that are essential for sustaining agricultural productivity and environmental quality. Failure to incorporate the value of environmental and health outcomes of agricultural NRM investments will lead to bias and likely underestimation of their social net benefits, followed by underinvestment from the standpoint of social welfare. The mirror image of such misallocation of R&D resources is equally troubling, for it entails overinvestment in agricultural programmes that may cause environmental and health damage.

Methods for comprehensive economic impact assessment that would integrate productivity, environmental and sustainability impacts are only just beginning to emerge. This volume has assembled recent methodological advances from this nascent area. It has critiqued the methodological *status quo*, and sought to define new horizons for experimentation to refine current practices and to develop second-generation methods that address existing and emerging challenges. The key challenges relate to measurement, indicators, valuation and attribution of impacts.

As we look into the future, NRM is entering a new era. With the emerging recognition that participatory NRM projects can empower individuals and communities, empowerment is shifting from being an unintended to an intended benefit. As it becomes an explicitly intended outcome of integrated NRM projects, empowerment begs the same needs for measurement, attribution and valuation that have challenged assessments of environmental and health dimensions of NRM interventions. Likewise, INRM projects typically prioritise poverty alleviation, making measures of income distribution effects another newly important dimension of NRM impact assessment.

In addition to application of new methods from environmental and resource economics, future NRM impact assessments have much to gain from employing a mix of quantitative and qualitative approaches. This can enrich interpretation and communication of outcomes and assist in their attribution. Qualitative methods can be especially helpful at elucidating *how* outcomes came to be. Such process understanding has particular value for unanticipated outcomes, with an eye to ensuring that desirable ones can be replicated and undesirable ones avoided in future.

Participatory impact assessments by NRM project beneficiaries may also enhance the empowerment outcome and associated impacts. However, the role of such participatory assessments should be recognised as a self-monitoring activity that is part of the project effort, not a true impact assessment of what would have occurred without the project. Accurate impact assessment, even of environmental and empowerment dimensions, must adhere to the basic principles of: 1. freedom from participant selection bias; 2. before vs. after comparisons aided by baseline information; and 3. 'with vs. without' measures of what the intervention accomplished.

The needs for improved methods for economic and social impact assessment are matched by needs for improved understanding of ecosystem performance. Following Altieri's call for increased 'understanding of the structure and dynamics of agricultural and natural resource ecosystems and providing guidelines to their productive and sustainable management' (Altieri, 2002, p. 7), the need for close future interdisciplinary collaboration is clear. Forging strong linkages and effective dialogue among ecologists, economists, and other social scientists is a *sine qua non* for future advances in scientifically sound natural resource management interventions and for thorough and balanced evaluations of their impacts.

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