IMPACT ASSESSMENT OF WATERSHED DEVELOPMENT

Issues, Methods and Experiences

Editors
K. Palanisami and D. Suresh Kumar

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Watershed development has been conceived as one of the important rural development programmes in India where the rainfed agriculture is characterised by low productivity, degraded natural resources and widespread poverty. Recognising the importance of watershed development for their perceived ability to promote agriculture and rural development, both Central and State governments make huge investments on watershed development and management. As million of rupees are invested on watershed development it is essential that the programme have positive impact. It is in this context the Impact Assessment of Watershed Development and Management assumes importance.

Difficulties in the assessment of Watershed Development impact stem from its own multi-faceted nature. Watershed development changes (at different spatial and temporal scales) and is assessed in terms of multiple objectives, such as poverty alleviation, resilience, and soil and water resources conservation, and human social development, that reflect the needs and expectations of different stakeholders.

Impact assessment estimates the effect of an intervention on its physical surroundings, the people involved, and/or the organizational context, and refers to short term outputs of products, medium term result, and longer term consequences or outcomes, and is a crucial component of the watershed development and management policy making. Impact assessment facilitates informed decision-making processes of different stakeholders. The stakeholders range from local farmers, community organizations, governments to external development oriented organizations, and researchers and policy makers. Assessment of the progress and changes effected by watershed development activities over a specified time period requires a systematic and continuous monitoring process. In addition, an evaluation process permits identifications of broader positive and negative outcomes of watershed treatment activities and to reach conclusion about their overall effectiveness in reaching projected objectives. Impact assessment of watershed development can feedback into planning and prioritization process.

The present volume is the outcome of a national seminar on "Impact Assessment of Watershed Development : Conceptual and Methodological Issues" organised by Water Technology Centre, Tamil Nadu Agricultural University, Coimbatore. It contains valuable papers on different themes such as Theoretical and Empirical Issues in Watershed Impact Assessment, Decision Support System for Watershed Development Planning and Evaluation and Experiences and case studies in relation to impact assessment of watershed develoopment. The book attempts to provide a forum for discussing various issues in relation to impact assessment of watershed development. It is hoped that this effort of ours will enhance understanding of the issues involved and aid in proper policy framework. Also the book would be very much useful to the researchers, scholars, NGOs, PIAs, government officials,donor agencies and policy makers and planners.

ISBN : 81-85211-62-0 : 2006

Price : Rs. 695.00

Associated Publishing Company

Post Box No. 2679, Karol Bagh, New Delhi - 110 005

E-mail : apc@apcbooks.com

Website : www.apcbooks.com

ISBN 81-85211-62-0

9788185211626

Assessing Impact of Integrated Natural Resource Management Technologies in Watersheds

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Introduction

Natural resource management (NRM) is an important issue to be addressed carefully, more so in the semi-arid tropics where majority of the rural livelihoods are dependent on agriculture. Water scarcity and land degradation are the major constraints along with poor socio-economic conditions and lack of infrastructure for increasing agricultural productivity of rainfed systems. The sustainability of production, soil quality and other environment resources are the major impact factors of INRM. With watershed as an entry point, assessing the impact of integrated natural resource management (INRM) interventions offer useful information on the performance of agricultural watersheds. Pathways of impact in watersheds are multi-pronged (Fig. 1) and complex demanding critical analytical framework to assess the impact of watersheds. Agricultural interventions typically involve opening closed natural systems that may have attained 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 (INRM) options have been developed and implemented (Wani *et al.*, 2004).

Economists and natural resource experts have long struggled to assess the broader economic and environmental impacts of INRM technologies. This has been a difficult task because such technologies are not separately developed and marketed as divisible component inputs like seeds. Typically INRM 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. Moreover, several interventions leading to impacts such as availability of drinking water in the village, increased self dependence, happiness, satisfaction of empowerment, development of institutions which enable the community to cope with adverse conditions are such which cannot be measured in economic terms. However, the full social impact of INRM cannot be measured directly using conventional methods of economic evaluation (Shiferaw *et al.*, 2004). Therefore, appropriate qualitative and quantitative indicators of biophysical, social and institutional impacts 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 INRM impacts will need new methods, tools and multidisciplinary teams of experts to understand and accurately quantify the benefits. Some non-marketed agro-ecosystem services and environmental benefits are especially difficult to recognise and quantify. Such tools as simulation modelling, geographic information system (GIS), and satellite imaging used in conjunction with traditional productivity-based techniques can be used to estimate some impacts. Productivity-based indicators (e.g. biomass and crop yields) at micro levels need to be complemented by indicators like the vegetation index at eco-regional 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.

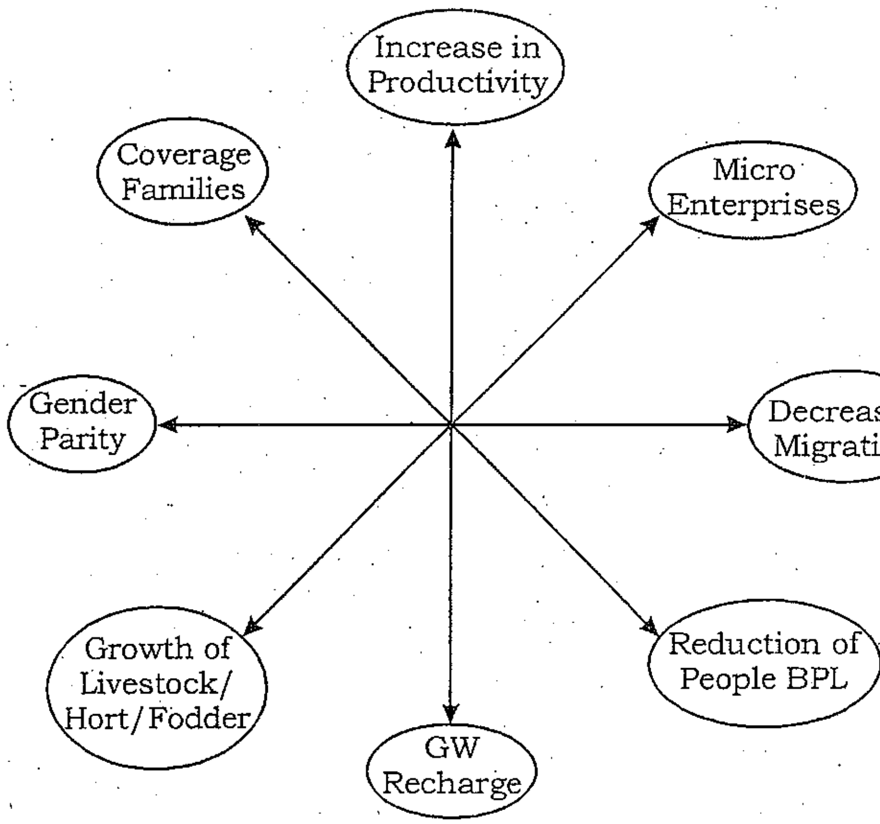

Fig. 1. Eight arms of holistic development

The criteria and indicators for monitoring INRM impacts related to various ecological functions and ecosystem services are described. Indicators used for assessing soil quality, water availability and quality, monitoring changes in the flow of such other ecosystem services as biodiversity conservation, carbon sequestration and ecosystem regulation, and tools and methods available to monitor and estimate the impacts associated with adoption of INRM technologies on various scales (use of simulation models to estimate biophysical changes, how remote sensing and GIS tools can be used to monitor spatial and temporal changes) are presented subsequently. The key issues and areas for future research are highlighted.

Indicators of INRM Impact

Indicators need to reflect the biological, chemical or physical attributes of ecological conditions to characterise current status and predict significant change. With a foundation of diagnostic research, an ecological indicator may also be used to identify major ecosystem stress. 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 INRM 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 like environmental statistics that measure or reflect environmental status or changes in resource conditions. 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 INRM 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 1 presents commonly used and potential indicators for monitoring INRM impacts.

Biodiversity Indicators

Natural resource management affects biodiversity on various scales. Indicators are required to assess the impacts of INRM interventions on natural and managed ecosystems. Biodiversity 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. Biodiversity provides many benefits. Its reduction influences the structure, stability and functions 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.*, 2004; Shiferaw *et al.*, 2004). These benefits include such ecosystem services as air and water purification, climate regulation, soil formation, and the generation of moisture and oxygen.

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 changes in biodiversity for species richness, diversity, and risk index.

Table 1: Indicators for monitoring biophysical and sustainability impacts of INRM 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

Source: Wani *et al.*, 2004.

Agro-biodiversity Indicators

Agricultural biodiversity or agro-biodiversity includes cultural and spiritual dimensions of biodiversity together with the practical and economic values of gaining sustainable rural livelihoods for poor people (Altieri, 1999). Agrobiodiversity includes 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 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 agro-ecologically adapted to the prevailing management systems. Genetic variability or diversity refers to variation in the genetic composition within or among species. From traditional Mendelian methods to DNA-based molecular techniques provide 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 aerial 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 INRM 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. The examples given for watershed Kothapally (India) show how such quantitative indicators as CAF, and surface variability of main crops have changed as a result of integrated watershed management interventions. 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/pigeon pea intercrop (40 ha in 1998 to 180 ha in 2002); thereby changing the CAF from 0.41 in 1998 to 0.73 in 2002 (Wani *et al.*, 2003b).

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 INRM 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 INRM 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 post rainy-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 INRM options were tested using time series yield data during 1977-2003 (Fig. 2) along with soil quality parameters. Crop yields increased under both management practices, but the annual productivity growth under improved management (78 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.

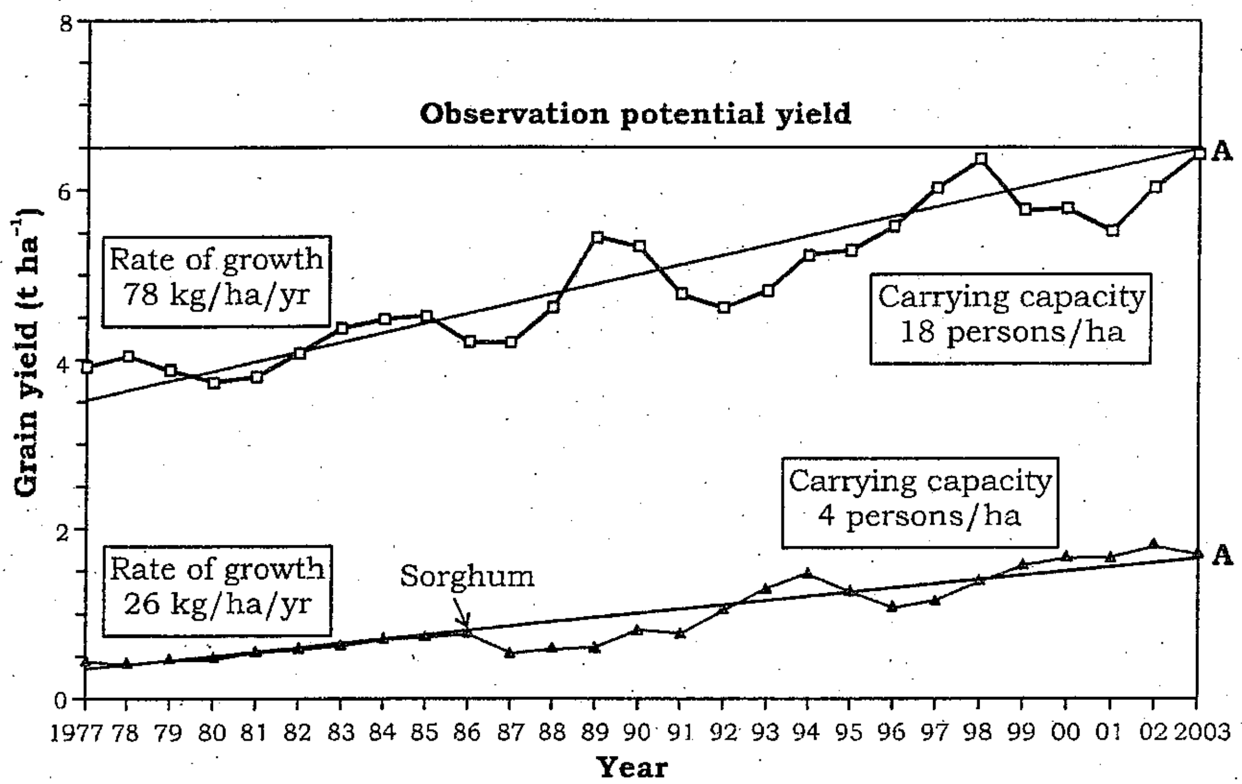

Fig. 2. Average grain yields under improved (A) and traditional (B) technologies on a Vertisol watershed at ICRISAT (1977-2002)

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, INRM impact can also be estimated in terms of the extent to which improved INRM 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 INRM interventions to increase soybean productivity at the district level (Table 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 2. Simulated soybean yields and yield gap for the selected locations in India

Location date	*Mean sowing*	*Harvest date*	*Simulated yields (kg/ha)*		*Mean observed Yield[a] (kg/ha)*	*yield gap (kg/ha)*
			Mean	*SD*		
Raisen	22 Jun	11 Oct	2882	1269	—	—
Betul	19 Jun	8 Oct	2141	603	858	1283
Guna	30 Jun	14 Oct	1633	907	840	793
Bhopal	16 Jun	8 Oct	2310	615	1000	1310
Indore	22 Jun	10 Oct	2273	939	1122	1151
Kota	3 Jul	16 Oct	1165	936	1014	151
Wardha	17 Jun	6 Oct	3040	640	1042	1998
Jabalpur	23 Jun	11 Oct	2079	382	896	1183
Amaravathi	18 Jun	8 Oct	1552	713	942	610
Belgaum	17 Jun	30 Sep	1844	629	570	1274

[a]Mean of reported yields during 1990-1995.

Related to the productivity measure, various economic efficiency indicators like the benefit-cost ratio can also be computed to evaluate the efficiency of agro-ecosystems. Such indicators can be used to evaluate the economic feasibility of various cropping systems and sustainability enhancing INRM options (Tisdell, 1996). Simple economic productivity indicators 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.

Bio-physico-chemical Indicators

To diagnose and quantify the impacts of various INRM interventions, reliable and measurable soil quality indicators are important as they reflect on the inherent capacity of the soil to perform its production and environment related functions (Pathak *et al.*, 2004). An example of use of some of the indicators for monitoring soil quality in crop production systems is shown in Table 3. The long-term effects of improved and traditional management on soil chemical and biological properties of Vertisols show that 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.*, 2003a).

Table 3. 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 center, Patancheru, India

Properties	*System*	*Soil depth (cm)*		*SE±*
		0-60	60-120	
Soil respiration	Improved	723	342	7.8
(kg C ha^{-1})	Traditional	260	98	
Microbial biomass C	Improved	2676	2137	48.2
(kg C/ha)	Traditional	1462	1088	
Organic carbon	Improved	27.4	19.4	0.89
(t C/ha)	Traditional	21.4	18.1	
Mineral N	Improved	28.2	10.3	2.88
(kg N/ha)	Traditional	15.4	26.0	
Net N mineralization	Improved	–3.3	–6.3	4.22
(kg N/ha)	Traditional	32.6	15.4	
Microbial biomass N	Improved	86.4	39.2	2.3
(kg N/ha)	Traditional	42.1	25.8	
Non-microbial organic N	Improved	2569	1879	156.9
(kg N/ha)	Traditional	2218	1832	
Total N	Improved	2684	1928	156.6
(kg N/ha)	Traditional	2276	1884	
Olsen P	Improved	6.1	1.6	0.36
(kg P/ha)	Traditional	1.5	1.0	

SE = Standard error of mean
Source: Wani *et al.*, 2003a

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 INRM 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 INRM 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 INRM 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 INRM 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 INRM, 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 INRM practices that sequester more carbon from the atmosphere. In February 2005, the Kyoto Protocol is operationalized under which the principle that polluters pay is established. 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 (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 4). 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 agri-horticultural systems can be quantified using suitable biochemical methods based on data collected from long-term experiments. The amount of C sequestered 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 INRM systems, once quantified in C units, can now be valued in economic terms.

Table 4. Carbon sequestration potential of various land management practices under dryland conditions

Management practice	*C sequestration potential (t C/ha/yr)*
Conservation tillage	0.10-0.20
Mulch Farming (4-6 t/ha/yr)	0.05-0.10
Compost application (20 t/ha/yr)	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	00.05-0.10

Source: Lal (1999)

Using this approach, Bruce *et al.* (1999) recorded an annual soil C gain of 0.2 t/ha on pasture and rangelands in the United States 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 per year attributable to INRM. Under improved soil fertility (60 kg N and 20 kg P/ha per 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.

Based on the work conducted by Wani and associated through the NATP project (NATP, 2004), the investigation on 28 benchmark spots covering 52 pedons across 5 bio-climatic zones in the semi-arid tropics, India through systematic studies and documentation of soil parameters vis-à-vis the management interventions helped to identify 22 systems comprising of forest (2 nos.), fallow (1 no.), horticulture (2 nos.) and agriculture (17 nos.). Highest concentration of organic carbon (1.44%) was under forest system, followed by permanent fallow (1.42%), horticultural system (0.80%), agricultural system (0.70%) and wasteland (0.47%). The sequestration of inorganic carbon on the other hand was found to be highest in horticultural system and agricultural system (0.80%), followed by wasteland (0.70%) and forest system (0.16%) (NATP-terminal report, 2004).

Growing knowledge on the C-sequestration benefits of INRM 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 (d' Silva *et al.*, 2004). 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 representation 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. 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 INRM 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. 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 provides a summary of different types of simulation models.

Table 5. 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 v 3.5
DSSAT v 3.5	Decision support systems for agro-technology 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 agroforestry	C and N changes in soils in response to land clearing and agronomic management of agroforestry 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

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 INRM 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 INRM 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.

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.*, 2004, and Sahrawat *et al.* 2004). 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 assess 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.

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. 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.

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 INRM 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 INRM investments. Alternatively, soil C simulation models can also be used to simulate the impact of INRM 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 v 3.5 (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. *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 NR with the ancillary information and to develop a decision-support system.

Impact assessment of INRM 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 INRM 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 (geostationary 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 INRM impacts. How spaceborne multispectral data could be used to monitor the spatial and temporal dynamics of agro-ecosystems is discussed here.

Vegetation Cover

Amongst various biophysical parameters relevant to INRM 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), 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. NDVI can be used as an indicator of change in relative biomass and greenness.

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. 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 INRM interventions has been studied in the Ghod catchment. 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).

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), saltaffected 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).

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 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 shows that 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.

Social and Institutional Impacts and Learnings

Watershed management programs are moving more towards holistic community participatory, empowering community and impact oriented (Wani *et al* 2003-ADB) one. Building institutions and social capital through empowerment is one of the important pathways of impact in the watershed programs. These benefits cannot be assessed through simple econometric impact assessment methods generally adopted. There is an urgent need to develop, adopt and include qualitative and semi-quantitative indicators for assessing social, institutional and pshycological impacts on people due to watershed development. This re-emphasises the need for multidisciplinary team of scientists to assess the impact of watersheds programs holistically as against the conventional compartmental impact assessment methods.

A key source of evidence needed for impact assessment is the monitoring and evaluation carried out within the project, which also provides real-time information necessary to facilitate the adaptive management of all stakeholders necessary for successful INRM. Effective monitoring and evaluation should be based on a shared view amongst the stakeholders of the outcomes they expect the project to contribute, and how these outcomes contribute to larger-scale developmental impact through knowledge generation and diffusion, emergence and evolution of innovation networks and creation of organisational capabilities (Douthwaite *et al.*, 2004)

Conclusion

Assessing the multi-dimensional impacts of INRM interventions– especially in non-tangible environmental services, institutional and social impacts– 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 INRM 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.

Agro-biodiversity and agro-ecosystem efficiency indicators can be applied on different spatial scales. The impacts of INRM 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. 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 INRM 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 56m from Advanced Wide Field Sensor (AWFS) to 23m from LISS-III to 5.8m from LISS-IV offer unique opportunities to monitor biophysical impact indicators on different spatial and temporal 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 INRM impacts on ecoregional scales. Such indicators will complement and enhance economic approaches for evaluating the impacts of INRM interventions, especially on larger spatial scales. Certain benefits such as aesthetic, satisfaction derived by the farming community due to the positive impact of INRM interventions in watershed need to be accounted. 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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