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

S.P. Wani[1], Piara Singh[1], R.S. Dwivedi[2], R.R. Navalgund[2] and A. Ramakrishna[1]

[1] *International Crops Research Institute for the Semi-Arid Tropics (ICRISAT) Patancheru, India*
[2] *National Remote Sensing Agency (NRSA), Hyderabad, India*

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

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

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):

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

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:

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

where $0 \leq SI \leq 1$, 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:

$$EI = H/ln(S) \tag{3}$$

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.

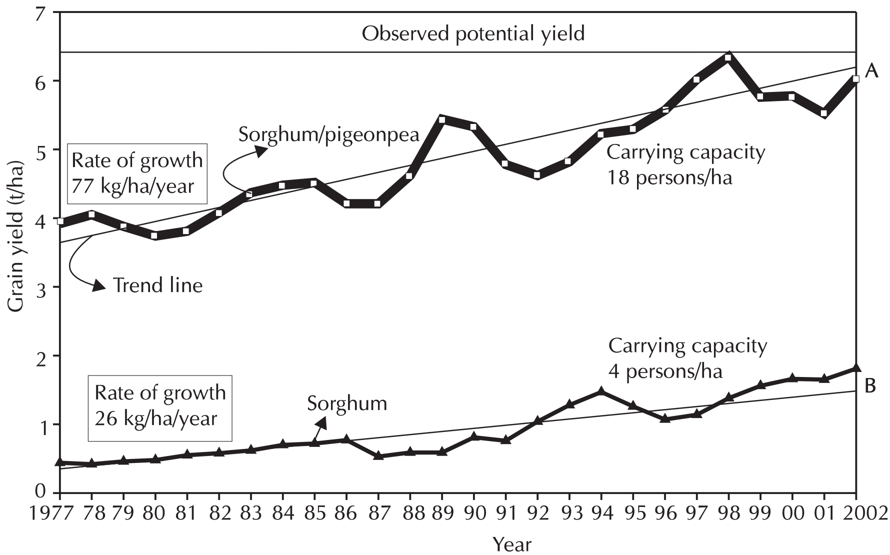

Fig. 5.1. Average grain yields under improved (A) and traditional (B) technologies on a Vertisol watershed at ICRISAT (1977–2002).

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 observed yield[a] (kg/ha)	Yield gap (kg/ha)
			Mean	SD		
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:

$$\text{NDVI} = \frac{\text{NIR} - \text{R}}{\text{NIR} + \text{R}} \qquad (4)$$

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).

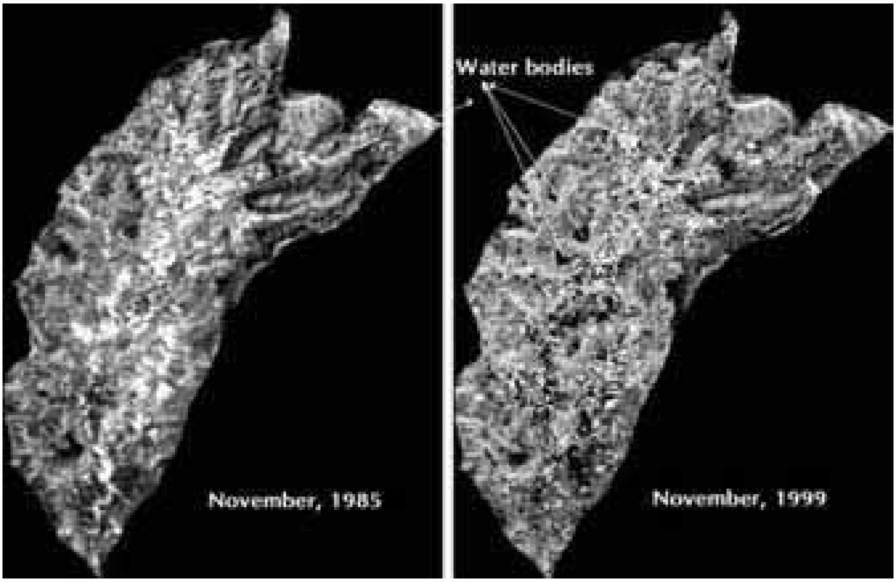

Fig. 5.2. Changes in surface water resources in gc1f micro-watershed, Ghod catchment, Maharashtra, India.

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

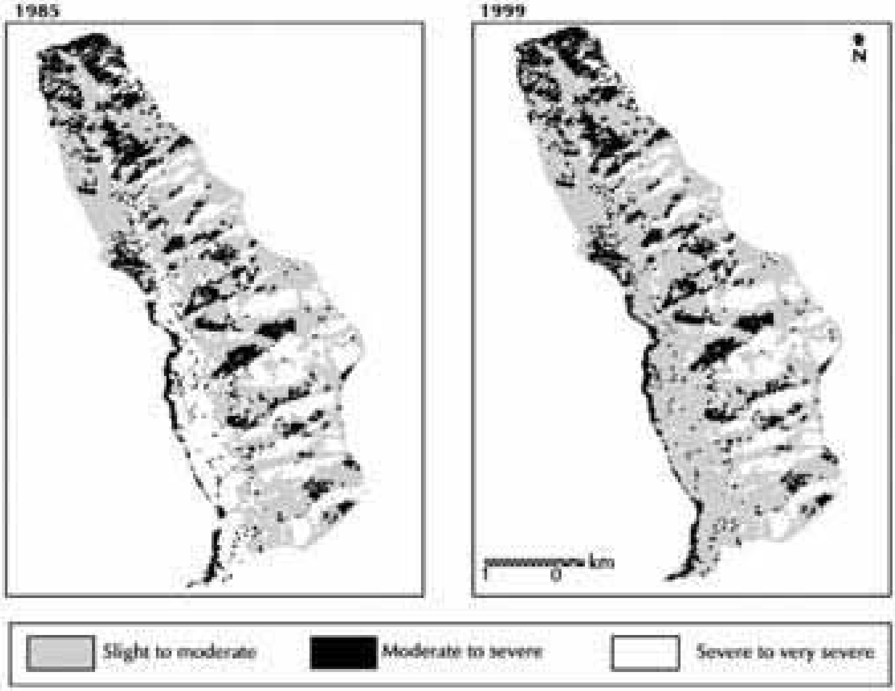

Fig. 5.3. Change in spatial extent and distribution of eroded lands in rg2h micro-watershed, Ramganga catchment, Uttaranchal, northern India.

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.

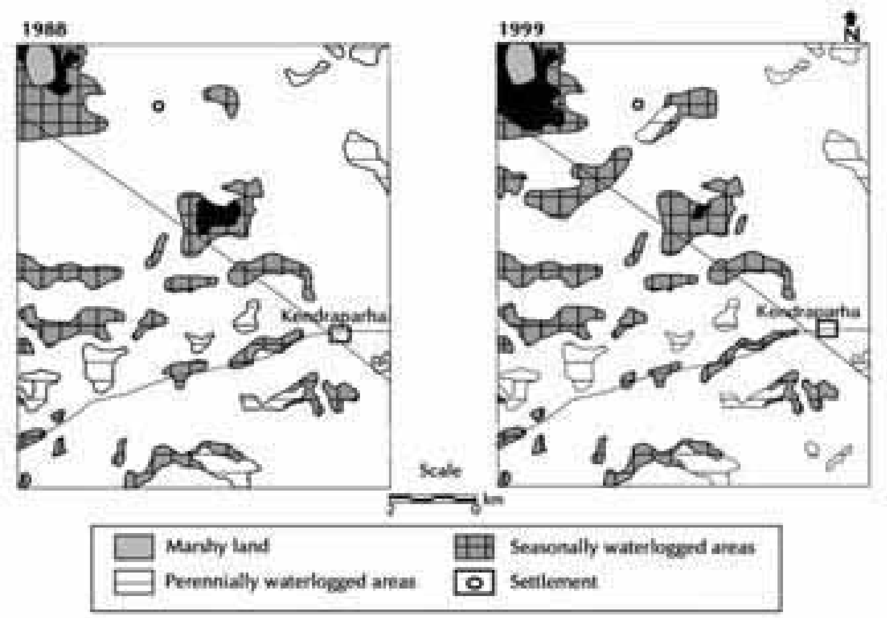

Fig. 5.4. Dynamics of waterlogging in part of Kendrapartha district, Mahanadi Stage-I command, Orissa, India.

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.

References

Adam, S., Wiebe, J., Collins, M. and Pietroniro, A. (1998) RADARSAT flood mapping in the Peace-Athabasca Delta, Canada. *Canadian Journal of Remote Sensing* 24, 69–79.

Aldrich, R. (1971) Space photos for land use and forestry. *Photogrammetric Engineering and Remote Sensing* 37, 389–401.

Alocilja, E.C. and Ritchie, J.T. (1993) Multicriteria optimization for a sustainable agriculture. In: Penning de Vries, F.W.T., Teng, P. and Metselaar, K. (eds) *Systems Approaches for Agricultural Development.* Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 381–396.

Altieri, M.A. (1999) The ecological role of biodiversity in agroecosystems. *Agriculture, Ecosystems and Environment* 74, 19–31.