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

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Introduction

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

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

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

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

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

Agricultural Practices and Water Quantity

Water availability indicators

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

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

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

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

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

Indicators for available surface water

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

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

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

Indicators for surface water outflow (runoff)

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

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

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

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

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

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

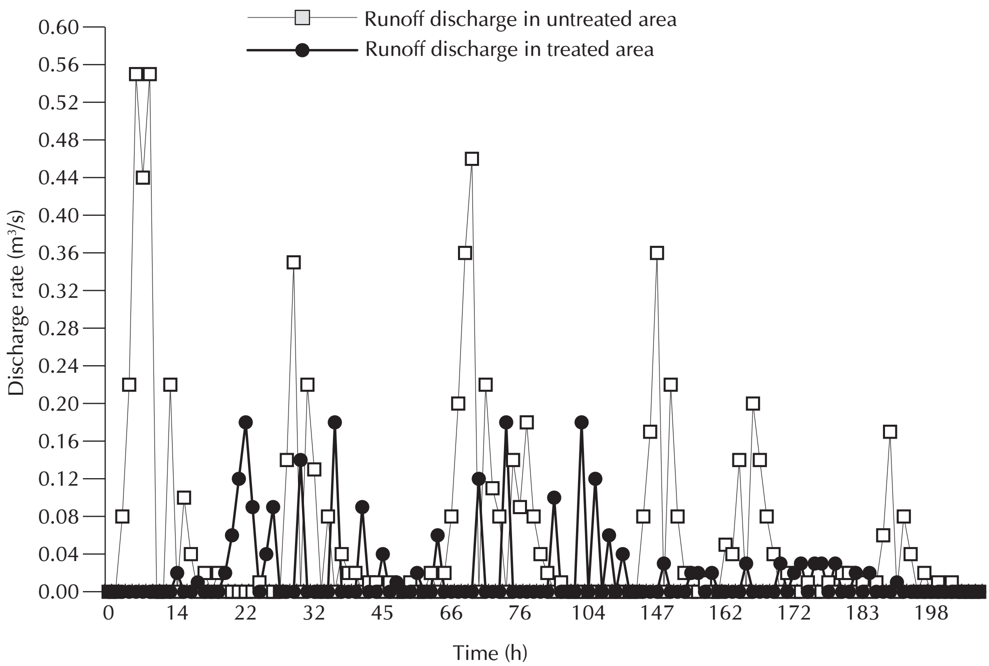

Fig. 4.1. The impact of integrated watershed management interventions on runoff as measured by a runoff hydrograph from untreated and treated sub-watersheds at Lalatora watershed, India, during 5–14 September 1999 (Pathak *et al.,* 2002).

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

Indicators for upstream and downstream temporary flood frequency and area affected

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

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

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

Indicators for groundwater availability

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

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

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

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

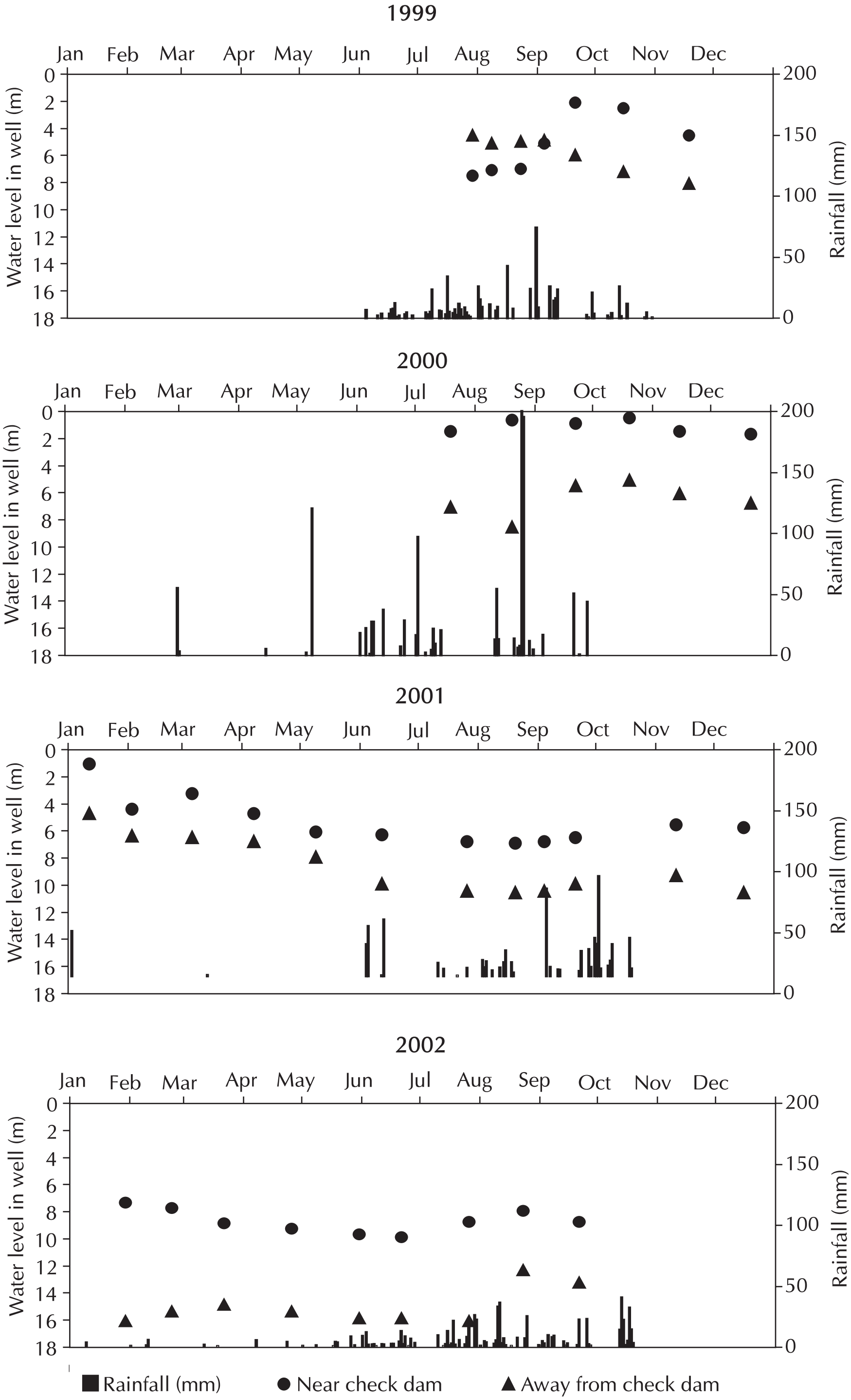

Fig. 4.2. The impact of check dam construction and soil and water conservation practices on groundwater levels at Adarsha watershed, Kothapally, India, 1999–2002 (ICRISAT, unpublished data).

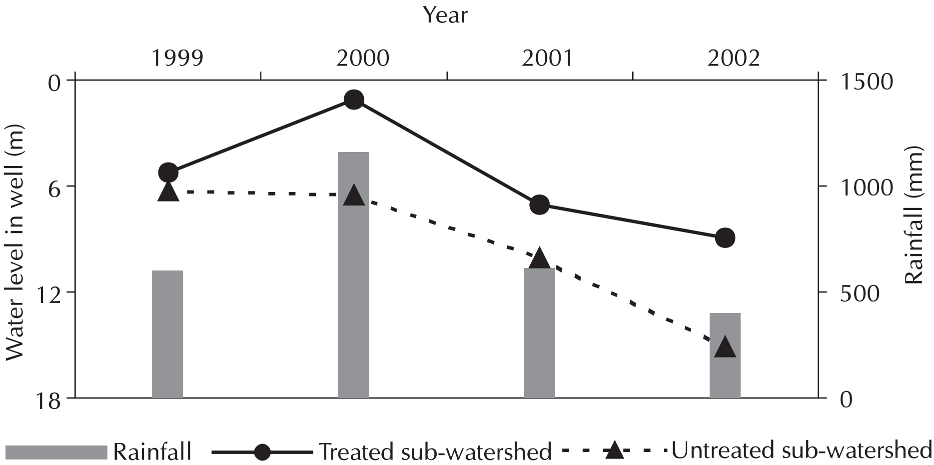

Fig. 4.3. The impact of integrated watershed management on groundwater levels at Adarsha watershed, India, 1999–2002 (ICRISAT, unpublished data).

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

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

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

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

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

Indicators for rainfall use efficiency

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

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

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

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

Water Quality Indicators

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

[a]NA = not available.

Application of Simulation Modelling

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

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

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

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

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

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

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

Summary and Conclusions

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

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

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

Commonly used water quality indicators include:

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

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

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

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

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

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

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